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Articles

Article

Evaluation of The Relationship between Abundance of Pollinators and Landscape Structure in Hyuganatsu (*Citrus tamurana*) Orchards in Aya Town, Miyazaki Prefecture

Takahiro Yumura[*1], Yasushi Mitsuda[*1], Mari Iwamoto[*1], Ryoko Hirata[*1] and Satoshi Ito[*1]

ABSTRACT

The study aimed to examine the relationship between pollination service (as an ecosystem service) and the landscape structure of Aya Town, Miyazaki Prefecture, Japan. Our target agricultural crop for the evaluation of pollination service was hyuganatsu (*Citrus tamurana*), and native (*Apis cerana*) and managed honey bees (*Apis mellifera*) were considered the key species of ecosystem service provider. We selected 15 hyuganatsu trees in 5 orchards and counted the number of honey bee visits. We tested local and macro scale landscape indices as explanatory variables to clarify the number of honey bee visits. A model that used the number of flowers, area of adjacent natural forests, and area of agricultural fields within a 1-km radius was selected as the best model. Our results suggest that landscape structure affected the number of honey bee visits to a hyuganatsu tree, which represents the quantity of ecosystem service for the tree, and should be considered in the evaluation of ecosystem services.

keyword: ecosystem service, honey bee, hyuganatsu, landscape structure, Aya Biosphere Reserve

INTRODUCTION

Evaluation of ecosystem services, which is defined as "the benefits people obtain from ecosystems (Millennium Ecosystem Assessment, 2005)", is one of the most urgent topics not only for ecological research, but also for ecosystem management. Understanding the mechanisms underlying the provision of ecosystem services by ecosystems is essential for the quantification of ecosystem services, and it also contributes as an important factor in ecological studies (Chee, 2004). Evaluation of ecosystem services, on the other hand, is essential for ecosystem management (Kremen, 2005), because the goal of ecosystem management is sustainability of ecosystem services (Millennium Ecosystem Assessment, 2005).

Several studies have been conducted on ecosystem services to map the supply and demand for ecosystem services, to assess threats to them, and to estimate the economic value of ecosystem services (e.g. Daily, 2000; Kremen, 2005). Pollination service, which is classified as a regulating service in the Millennium Ecosystem Assessment (2005), has been well studied, because it is strongly related to food production (Klein et al., 2007; Gallai et al., 2009). Gallai et al. (2009) estimated that the total economic value of pollination worldwide amounted to 153 billion Euro. Pollination by bees and other animals increases both yield quantity and quality of approximately 75% of the leading global crops (Klein et al., 2007; Ricketts et al., 2008).

Abundance of pollinators, such as insects, birds, and mammals, has been used as an indicator of the quantity of ecosystem services (Ricketts, 2004; Taki et al., 2010). Some direct measures of pollination service have been developed (Thomson and Goodell, 2001; Kremen et al., 2002; Klein et al., 2003), however, we focused on the abundance of pollinators as an indirect measure of ecosystem service because of the ease of measurement in the field. Additionally, some studies have shown a strong relationship between abundance of pollinators and efficiency of food production (Cane, 2005; Bosch et al., 2006; Isaacs and Kirk, 2010). The abundance of native honey bee (*Apis cerana*), as a key pollinator, was significantly correlated to crop production of common buckwheat (*Fagopyrum esculentum*) (Taki et al., 2010). Further, Taki et al. (2010) revealed that the visitation rate of native honey bee increased with the increase in surrounding forest areas.

As reviewed by Ricketts et al. (2008), recent studies have revealed that the quantity of pollination service was affected by the surrounding landscape structure. For instance, Carvalheiro et al. (2010; 2011) revealed that the production of mango (*Mangifera indica*) and sunflower (*Helianthus annuus*) was remarkably affected by landscape structure, i.e. the distance from natural habitats (forests and grass lands)

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represent proximity of a target tree to favourable habitats of honey bees, and is regarded as an index of landscape structure. Tscharntke et al. (2005) claimed that it is important to understand the effects of agricultural land-use on ecosystem service from the landscape perspective, and that the relationship between ecosystem service and landscape structure contributes to the success of ecosystem management.

Our target crop for the evaluation of pollination service is hyuganatsu (*Citrus tamurana*), which is a citrus fruit plant grown in Japan. Hyuganatsu is an important agricultural product of Aya Town, Miyazaki Prefecture, where we conducted this study. This plant exhibits strong self-incompatibility (a genetic characteristic of flowering plants to prevent self-pollination and to promote cross-pollination), and requires pollinators for fruiting (Honsho et al., 2009). We defined native and managed honey bees (*Apis mellifera*) as the key species of ecosystem service provider for the open-field culture of hyuganatsu.

The objective of this study was to examine the relationship between abundance of honey bee visits, as an indicator of pollination service, and landscape structure defined by the spatial arrangement of patches of various land-use types. We developed regression models to explain the abundance of honey bees using indices of landscape structure and examined whether a particular landscape structure affected the pollination service or not. Our findings will contribute to a better understanding of the relationship between pollination service and landscape structure, and provide its implications on landscape management, which is a component of ecosystem management. Further the results of this study will give us some suggestions for forest planning with considerations in ecosystem services.

MATERIALS AND METHODS

Study Area

The present study was conducted in Aya Town, Miyazaki Prefecture (Fig. 1), located in the south-western part of Japan, which belongs to the warm temperate zone. The annual mean temperature and precipitation are 17°C and 2,300 mm, respectively. Aya Town has been designated as a UNESCO Biosphere Reserve in 2012 because it comprises large areas of lucidophyllous forests. Aya Town is also known for its long history in promoting organic agriculture by the town government. Hyuganatsu is an important agricultural product of Aya Town, which contributes to 17% of the total hyuganatsu production in Miyazaki Prefecture, which is the largest hyuganatsu producing area of Japan (Nagatomo et al., 2001).

Field Measurements

We counted the number of honey bee visits as the indicator of pollination service by field visual observation. We selected 5 hyuganatsu orchards in Aya Town (Fig. 1), and selected 2 to 5 target trees in each orchard; a total of 15 target trees were selected for counting honey bee visits. The geographical position of target trees was recorded using differential GPS (DGPS, Trimble, Pathfinder ProXH) and position data was corrected using Global Navigation Satellite Systems post-processing software (Trimble, Pathfinder Office) with GEONET data provided by the Geospatial Information Authority of Japan. For each target tree, we placed a 50 × 50−cm plastic frame and counted the number of honey bee visits to the frame by field visual observation. If a honey bee came to the frame and flew away, and the same honey bee revisited the same frame, we counted it as 2 visits (for the first visit and the revisit). We conducted observations on honey bee visits four times a day for 1 h each (10:00 to 11:00, 11:20 to 12:20, 15:00 to 16:00, and 16:20 to 17:20) from 4th to 22nd May 2014. Each target tree was subjected to this observation at least 5 times during the four observation times. We did not distinguish native honey bee from managed honey bee in order to eliminate the effects of the difference in the identification ability among observers.

It is known that the number of flowers of a plant represent the attractiveness for pollinators (index of

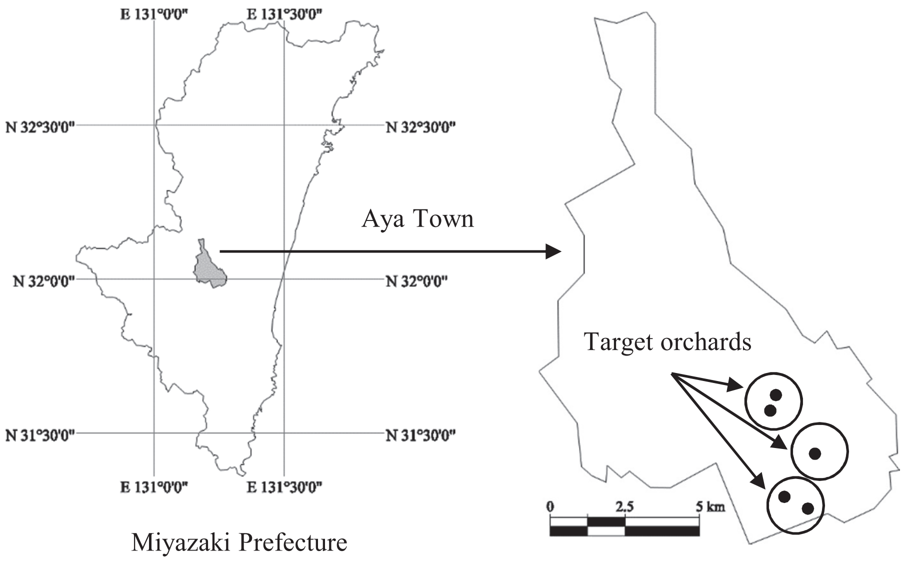

Fig. 1 Location of Aya Town and target hyuganatsu orchards.

individual attractiveness) (Kobayashi, 1981; Eckhart, 1991); therefore, we also counted the number of flowers on each target tree. In order to establish the effects of landscape structure on the number of honey bee visits, we treated the maximum number of honey bee visits of each target tree as the objective variable and the number of followers of each target tree when the maximum number was observed as an explanatory variable in the following statistical analysis.

Landscape Indices

For calculating landscape indices, a land-use vector map of the study area was developed by delineating each patch of land-use type on the orthophoto taken in 2013 provided by the Miyazaki Prefectural Government. We defined 15 land-use types: natural forest, planted forest, bamboo forest, newly planted forest, newly harvested forest, agricultural field, orchard, grass land, bare land, river, river bed, artificial green area, golf field, buildings, and roads.

We used two types of indices representing landscape structure, which were local-scale and macro-scale landscape indices, because the spatial scale of landscape structure was an important factor to be considered (Taki et al., 2010). According to previous studies, we selected totally 12 landscape indices (Table 1). Natural forests and agricultural fields were considered as suitable land-use types for providing favourable habitats for honey bees and were used as key land-use elements to calculate landscape indices, based on previous studies, which indicate that these two land-use types are important factors that affect the abundance of pollinators (Taki et al., 2007; 2010; Ricketts et al., 2008). Local scale landscape indices used in this study include shape index of target orchard polygon, area of adjacent natural forests (area of natural forest polygons neighbouring to each target orchard polygon), area of adjacent agricultural fields, boundary length between adjacent natural forests, and boundary length between adjacent agricultural fields. Macro scale landscape indices used in this study were: area, area proportion, and total edge length of natural forest patches existing within a 1-km radius, and those of agricultural field patches, and Satoyama index (defined by Kadoya and Washitani, 2011, see. Table 1). We used the 1-km radius for calculating macro scale indices, because according to Taki et al. (2010), a 1-km radius is a considerable foraging range for honey bee.

Statistical Analysis

We established the relationship between the number of honey bee visits to each target tree and the landscape structure surrounding each target tree by using the generalized linear regression model (GLM). As described above, we used the maximum number of honey bee visits ([visits/hour], hereafter NBV) as the indicator representing the quantity of pollination service and then the objective variables for GLMs. We used Poisson distribution for the statistical distribution of NBV and Log function for the link function in GLMs developed in the present study.

First, we developed a GLM using the number of followers of each target tree when the maximum number was observed (NF) as an explanatory variable for the reference (Model 1). Second, we added one of each local scale landscape index to Model 1, then developed GLMs with two explanatory variables. A total of 5 models were developed and compared by the goodness-of-prediction using Akaike's information criteria (AIC), we then selected the best model using NF and a local scale landscape index as explanatory variables (Model 2). Further, we determined the best model using NF, a local-scale landscape index, and a macro-scale landscape index as explanatory variables (Model 3). We assumed that neighbouring natural forests may be a crucial factor from the field observation, while previous studies showed that macro-scale landscape structure was important to predict abundance of honey bee visits. Therefore we applied this nested modelling approach.

RESULTS

The summary of field visual observation of honey bee visits are shown in Table 2. There were large variations in honey bee visits within each target tree, thus we used the maximum number of bee visits of each target tree as an indicator of abundance of pollinator.

Standardized coefficients of regression (SCR) and AICs of Model 1, 2, and 3 are shown in Table 3. Among local scale

Table 1 Description of variables used to develop regression models.

Name	Scale	Bibliography
Number of flowers (NF)	tree scale	Kobayashi, 1981
Shape index (*1)	local scale	Concepción et al., 2007
Area of the adjacent natural forests (AAN)	local scale	Taki et al., 2010
Area of the adjacent agricultural fields	local scale	Taki et al., 2010
Boundary length between adjacent natural forests	local scale	Rand et al., 2006
Boundary length between adjacent agricultural fields	local scale	Rand et al., 2006
Satoyama index (*2)	macro scale	Yoshioka et al., 2013
Area of natural forests	macro scale	Taki et al., 2010
Area of agricultural fields (AA)	macro scale	Taki et al., 2010
Proportion of natural forests	macro scale	Taki et al., 2010
Proportion of agricultural fields	macro scale	Taki et al., 2010
Edge length of agricultural field	macro scale	Rand et al., 2006
Edge length of natural forest	macro scale	Rand et al., 2006

*1 An index representing shape complexity using perimeter-to-area ratio

*2 A biodiversity index for agricultural landscapes

Table 2 Summary of the filed observation of honey bee visits.

Tree No.	Orcherd No.	Honey bee visits [visits/hour] Min.	Mean	Max.	Number of observatio
1	1	1	10.3	21	8
2	1	3	15.9	33	8
3	1	3	15.4	34	9
4	1	2	16.4	68	9
5	2	0	0.1	1	9
6	2	0	1.7	5	9
7	2	0	5.7	18	9
8	2	0	7.3	12	9
9	3	0	0.6	2	5
10	3	0	0.2	1	6
11	4	0	0.5	2	8
12	4	0	1.8	4	8
13	4	0	1.8	5	8
14	5	0	3.4	10	7
15	5	0	2.6	7	7

Table 3 Standardized regression coefficients and AIC of regression models.

Model	Formula	AIC
Model 1	2.357 + 0.912NF	191.94
Model 2	2.325 + 0.835NF + 0.245ANF	184.86
Model 3	2.205 + 0.243NF + 0.415ANF + 0.674AA	118.73

landscape indices, the area of adjacent natural forests (AAN) was selected as the best predictor variables. SCRs of NF and AAN in Model 2 were 0.835 and 0.245, respectively. AIC of Model 2 was 184.86 and lower than that of Model 1 (191.94). Area of agricultural fields within a 1-km radius (AA) was selected as the best predictor variable among macro-scale landscape indices. SRCs of NF, AAN, and AA in Model 3 were 0.243, 0.415, and 0.674, respectively. AIC of Model 3 was 118.73 and lower than that of Model 2. We tested the difference in AIC between Model 1 and Model 2 using parametric bootstrap, then the statistically significant difference ($p = 0.003$) was determined. The statistically significant difference was also determined between Model 2 and Model 3 ($p < 0.001$). The relationship between the predicted and observed NBV is shown in Fig. 2.

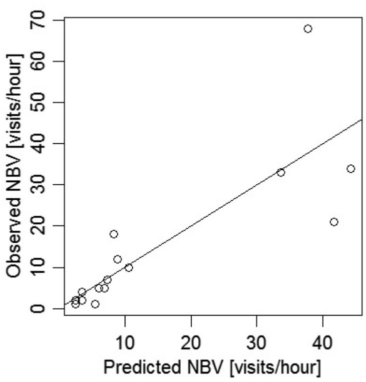

Fig. 2 Comparison between observed and predicted values of the number of honey bee visits.

DISCUSSION

Model 3 was the best model among the three GLMs, and estimated NBVs using this best model were remarkable over or under estimation for a few target trees; however, this best model could represent the trend in NBV of 15 target trees (Table 3, Fig. 2). These results suggest that the number of honey bee visits to hyuganatsu trees in the orchards in Aya Town are affected by surrounding landscape structure. Similar to previous studies (Kremen et al., 2002; Isaacs and Kirk, 2010; Taki et al., 2010), our study also reflects the effects of landscape structure on pollination service. In Model 3, SCR of AA was largest, followed by ANF, and then NF. Both landscape indices had greater effect on the number of honey bee visits than the number of flowers as the index of individual attractiveness. As we assumed, ANF was selected as the best predictor variable among local-scale landscape indices in Model 2. The result of the nested approach, however, suggested that the flowering level and both local and macro scale landscape structure should be considered all together, because Model 2 was superior to Model 1 and Model 3 was superior to Model 2. The flowering level represents the favourability for honey bees among trees in an orchard, and landscape structure represents the favourability among the orchards; therefore, landscape structure should be taken into consideration in the evaluation of pollination services.

The values of SCR of NF and AA were positive, which indicated that an orchard located near large natural forests and surrounded by large areas of agricultural fields can receive larger pollination service (Fig. 3). Natural forests are regarded as suitable nesting habitats for honey bees (Sasaki, 1999; Taki et al., 2010), and hyuganatsu orchards are suitable foraging habitats for honey bees in the flowering season. Thus, the landscape comprising neighbouring natural forests and hyuganatsu orchards is desirable both for honey bees and humans, because honey bees can find suitable nesting and foraging habitats and humans can receive large pollination service from the honey bees. Agricultural fields in Aya Town are also suitable foraging habitats, because major agricultural crops here include Japanese radish (*Raphanus sativus*), cucumber (*Cucumis sativus*), and common buckwheat, which provide food for honey bees. Agricultural fields and hyuganatsu orchards are suitable habitats for foraging and natural forests are suitable habitats for nesting; thus, a combination of these three land-use types can be a key factor to conserve honey bee communities and to maintain the pollination service from honey bees in Aya Town.

Previous studies have shown that intensive agriculture with much use of pesticides contributes to large negative impacts on the local population of honey bees (Ricketts et al., 2008; Chauzat et al., 2009; Henry et al., 2012). In this study, however, agricultural fields offered suitable habitats; this could be related to the long history of promoting organic agriculture by the town government since 1973. Aya Town government issued an ordinance that promote agriculture harmonized with ecosystem conservation in 1988 and approximately 90% of agricultural fields in Aya Town follow

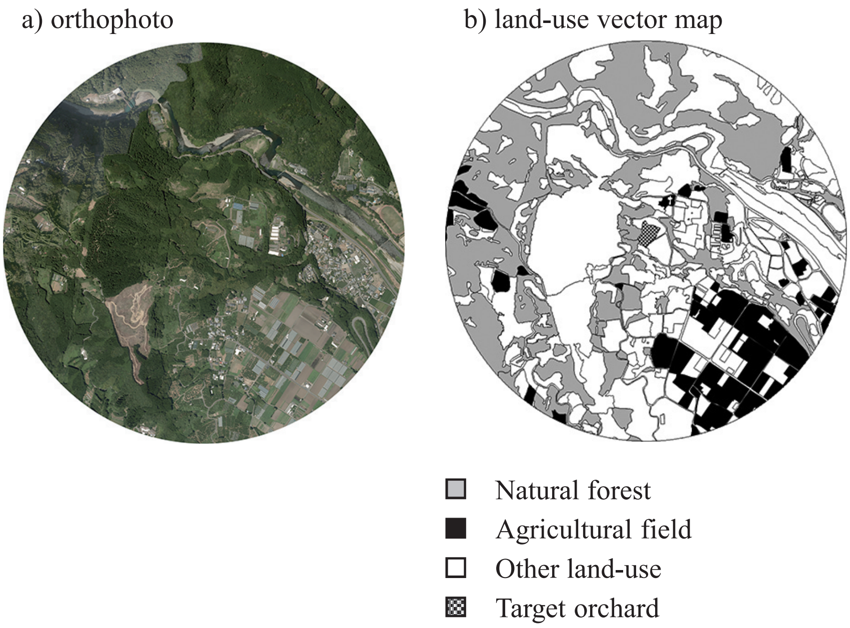

Fig. 3 An example of a suitable landscape structure for honey bees.

organic farming or reduced-chemical farming practices. Therefore the large part of agricultural fields in Aya Town can be safe foraging habitats without chemicals damaging honey bees. In hyuganatsu orchards also, reduced-chemical farming practices are followed. Thus, the results of the present study show a possibility of the positive effects of the long-term organic farming efforts of Aya Town.

In this study, we revealed the relationship between the quantity of ecosystem service, indicated by number of pollinator visit, and landscape structure. Our model described that hyuganatsu orchards located near natural forests and surrounded by agricultural fields within a 1-km radius can receive better pollination services. However, further studies are required to validate this conclusion. Remarkable estimation errors were observed at target trees in the orchard No.1, this suggested that positions of target trees in each target orchard may affect suitability for honey bee. The number of target orchards was only 5 and each target orchard was small; therefore, variations in local and macro scale landscape indices were relative low. Our results were derived from observations for only one flowering season. Thus, extensive studies involving more orchards and continuous field monitoring studies are required to further elucidate these aspects.

Farmers in hyuganatsu orchards of Aya Town manually conduct artificial pollination by hands, which is time- and labor-intensive and involves high costs. We believe our findings can contribute to reducing the efforts and cost of artificial pollination, and to design a future land-use and forest allocation plan taking ecosystem services into consideration. Continuous studies are therefore required not only for interests of ecological research but also for better hyuganatsu production harmonizing with the ecosystem management practices in the Aya Biosphere Reserves.

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Article

Analysis of Forest Cover Changes Using Landsat Satellite Imagery: A Case Study of the Madhupur Sal Forest in Bangladesh

Mohammad Abdullah Al Faruq[*1], Sourovi Zaman[*2] and Masato Katoh[*1]

ABSTRACT

Bangladesh is a poor, partially forested nation located in South Asia. The forests cover an estimated 17.1% of the land surface area of the nation. Rapid human population expansion has increased wood consumption and resource overexploitation, leading to the degradation of forest reserves. We mapped and analyzed forest cover change for the period 1972-2014 using Landsat satellite images of the Madhupur Sal forest captured in 1972, 1991, 2010, and 2014. This forest is a tropical deciduous stand within the Bangladeshi Tangail Forest Division. Forest cover changes were identified and approximately delineated on remotely sensed images. We applied a supervised classification approach to the satellite images using ERDAS IMAGINE ver.10 software. The mapping and analyses of five land-use classes were performed with ArcGIS ver.10 software. Thus, we analyzed the trends in forest cover changes over 42 years. The area under natural forest cover was progressively reduced by 7079.4 ha through anthropogenic activities during the period 1972-2010. However, the natural forest area increased by 202.4 ha between 2010 and 2014 due to the implementation of a revegetation program involving local community groups that was initiated by the national forest department. Our maps are very relevant to forest conservation initiatives and will enable a long-term, integrated approach to forest revegetation operated by the forest department in association with local communities.

keyword: forest cover, Madhupur, remote sensing, rubber plantation, Sal tree

INTRODUCTION

Among natural resources, forests are undoubtedly the most important for sustainable development across the globe. They directly influence local atmospheric cycles and contribute enormously to the diverse needs of forest-dependent and forest-independent peoples as well as to socioeconomic development and environmental stability. Unsustainable management of forest resources coupled with increasing population has resulted in dramatic change of forest cover in tropical countries. Here, change is defined as "an alteration in the surface components of the vegetation cover" (Milne, 1999). Detecting forest conditions and monitoring changes in forest structure lead to improved understanding of forest ecosystem services. Forest change detection is important for updating forest-cover maps and for monitoring and managing forest resources because it provides a quantitative analysis of the spatial distribution of forest density. For a comprehensive understanding of changes in tropical forest conditions, change analyses should span at least several decades (Lambin, 1999). Long-term datasets allow managers to draw meaningful inferences from the information at hand.

Temporal remote sensing data which offers an effective way to monitor gradual forest cover and ecosystem changes are used in many parts of the world. Remotely sensed data relating to land use and its changes over time provide crucial information for a range of diverse applications, such as environmental protection, forestry, hydrology, agriculture, and geology, among others. They present current conditions and provide digital data acquisition characteristics. Remote sensing is among the most accurate means of measuring the

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extent and pattern of changes in vegetation cover over a period of time (Miller et al., 1998). It has the potential to provide comprehensive information on diverse criteria for forest management. Data on forest/land-cover changes are required for updating land-cover maps, managing resources effectively, and planning for sustainable development (Alphan, 2003; Muttitanon and Tripathi, 2005). Satellite image quality and availability have greatly improved in recent years. These improvements have enabled progress in image analysis procedures. Although most developed countries have detailed updated information on land use and land cover, the dearth of geospatial databases for many developing countries hampers appropriate development planning.

Bangladesh is a densely populated, South Asian nation that is still in the development stage. Poverty and unemployment rates are very high, and disadvantaged people naturally place great pressure on the country's natural resources as they forage for fuel and food. Deforestation is a major issue in a nation where forest resources are of ecological and economic importance. The estimated per capita consumption of timber and fuel wood is only 0.01 m^3 and 0.07 m^3, and the demand for timber and fuel wood is calculated to be 3.2 and 8.7 million m^3, thus giving an estimated deficit of 62 and 60 percent, respectively (IMF, 2005). The forests contribute significantly to the agricultural income of Bangladesh. It contributed about 1.76% of the country's GDP and 16.77% of the agriculture income in 2012-13 (BBS, 2014). A total forest cover of 25% is required for the country to maintain ecological balance and environmental stability. Bangladesh has approximately 2.52×10^6 ha of forested land, accounting for *ca.* 17.1% of the national land area (BFD, 2010). In the past 30 years, the area of forest has rapidly decreased. The average annual rate of deforestation is 3-4% (Rasheed, 2008), which can be explained by the high rate of human population growth, commercial land use, the high demand for fuel wood, and extremely heavy use of natural resources (Alam et al. 2008). However, the poor budgetary practices of the national government do not provide extensive information on national forest resources. The scarcity of essential data delays programs aimed at mitigating the effects of deforestation. The development of strategies for estimating forest resources is therefore among the urgent issues required for sustainable forestry in Bangladesh.

Among the wooded tracts in Bangladesh, the Sal forest is crucial to the development of appropriate management methods for the country. This vegetation, which is dominated by the Sal tree (*Shorea robusta*), is the third largest forest ecosystem in Bangladesh (BFD, 2011); the stands are classified as tropical, moist, deciduous forest (UNEP, 2002). Sal forests occupy *ca.* 0.12 million ha, representing 4.7% of the total wooded area of Bangladesh (GOB, 2010). Most of the Sal forest is located in the greater Mymensingh and Tangail districts, also known as Madhupur Grath (Rahman, 2003). The Madhupur Sal forest is considered precious; it is the only forest located in the flood-free central part of the country. The Madhupur Sal forest has functioned for centuries as the homeland for ethnic communities such as the Garo and Koch (Ahmed, 2008). Thousands of people have become directly and indirectly dependent on the forest, placing it under severe pressure in recent decades through illegal logging, clearing for agriculture and industrialization, and the provision of livelihoods for the poor living around the forest (BFD, 2004). These pressures have caused significant changes to the forest and its associated resources. Encroachment and tree removal have significantly degraded forest ecological functions (Muhammed et. al., 2008). Such overexploitation, combined with inappropriate management, has made forest resource use unsustainable (Iftekhar, 2006).

The goal of sustainable forest management is to provide a steady flow of resources and income while preserving vegetation cover, biodiversity, and ecosystem integrity (Webb and Sah, 2003). In the early 1990s, the Bangladesh Forest Department and many NGOs participated in several social forestry programs to mitigate the deforestation (Salam and Noguchi, 2006; Alam et al., 2010). The programs contributed importantly by revegetating occupied forest lands with fast-growing plant species, but much less attention was given to sustainable conservation of the Sal forest. Sustainable forest management will require an understanding of the natural characteristics, distribution, quality, suitability, and limitations of the stands in relation to provision of resources to the human population. Most importantly, information on past and present land cover in the area is crucial for sustainable management (Chaurasia et al., 1996). Few literature sources describe the status of the Sal Forest or changes in its canopy cover over time, although a range of reports provide information on soil quality, biodiversity, social forestry, and agro-forestry. In the face of rapid deforestation and the dearth of information on changes in vegetation cover, we focused on calculations of forest-cover change by using remote sensing technology to classify digital images.

We attempted to map land use in the Madhupur Sal forest over 42 years (1972-2014), focusing primarily on estimation of the rate of forest-cover change. The data that emerged should facilitate dynamic forest management planning. Our work also contributed to the development of remote sensing technology for developing countries, such as Bangladesh, where data gathered by satellite imagery will be especially valuable, as information gathering at ground level is inherently difficult.

MATERIALS AND METHODS

Study Area

Forest is a complex ecosystem consisting mainly of trees that buffer the earth and support a myriad of life forms. In Bangladesh, forest land spanning more than 0.5 hectares with trees higher than 5 meters and a canopy cover of more than 10 percent, or trees able to reach these thresholds in situ. It does not include land that is predominantly under agricultural or urban land use (FRA, 2015). The Madhupur deciduous Sal forest (24°32'-24°47'N, 89°59'-90°11'E) covers an area of *ca.* 31,222 ha. It is located in the northeastern section of the Tangail Forest Division; a small segment runs along the boundary of the Mymensingh Forest Division (Fig. 1).

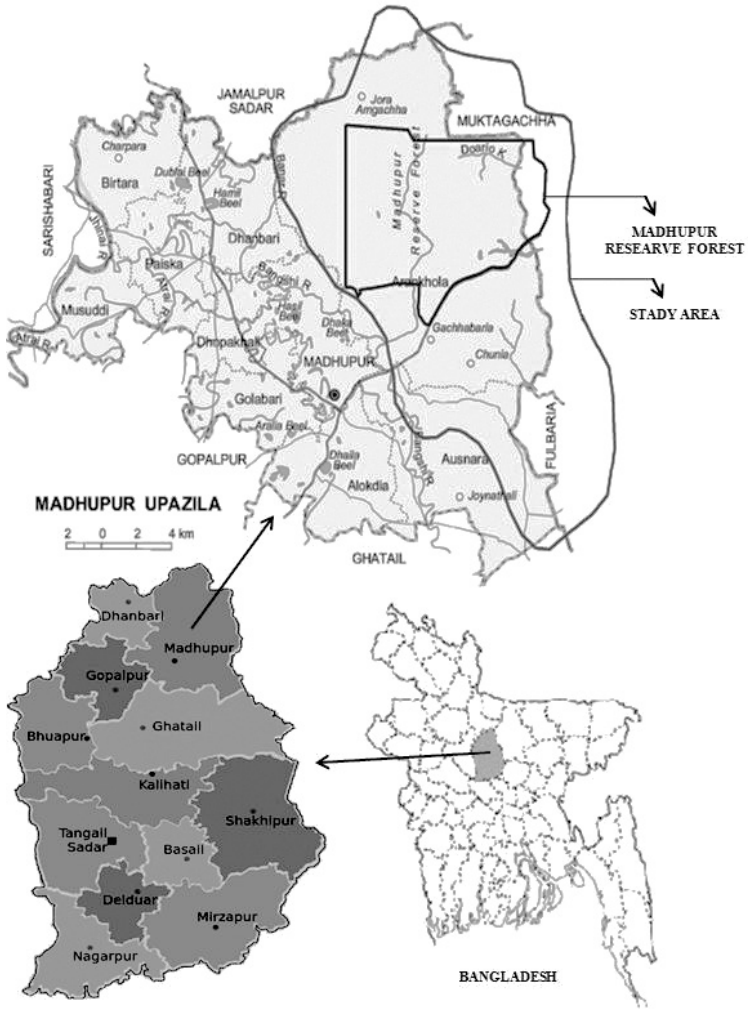

Fig. 1 Location of the study area in Madhupur, Bangladesh

The study site was selected because of the anthropogenic threats facing this stand. Land use in Bangladesh is changing rapidly, and new developments affect the area significantly. The Madhupur Sal stand is the only forest in central Bangladesh. Thousands of ethnic and non-ethnic people depend on its products. The forest is under strong pressure from illegal logging and land encroachment, both of which degrade forest resources. A major portion of the forested area (*ca.* 8,499 ha) has been given "reserve forest" status. The designation was announced in a gazette in 1982, when the reserve was renamed the "Madhupur National Park." Reducing and perhaps reversing the rapid degradation of the stand requires acquisition of robust data on the rates and trends of degradation. Conservation and preservation programs will be enabled by such reliable information.

The forest is located *ca.* 20 m above mean sea level. The mean annual rainfall is 2,000-2,300 mm, and annual mean temperature is 26.3°C (Rahman, 2003). The forest is divided into four administrative ranges: Jatyo Uddyan, Dokhola, Aronkhola, and Madhupur. The Madhupur Sal forest area, commonly known as the Madhupur Garh, is on a tract of land *ca.* 1-2 m higher than the surrounding plains. The ridges, known locally as Chala, are covered with forest formations and are not continuous. The forest is dense in some parts and sparse in others; scrub jungle also occurs on the ridges. The forest canopy height varies between 10 m and 30 m. The dominant species (80-100% of trees) is the commercially profitable Sal tree (*Shorea robusta*), which dominates the upper canopy. It is associated with Ajuli (*Dillenia jpentagyna*), Amlaki (*Phyllanthus emblica*), Koroi (*Albizia procera*), *Terminalia* sp, and Sonalu (*Cassia fistula*), among other taxa. The understory includes *Bambusa* sp., *Alsophila* sp., and several ferns and epiphytes (Feeroz and Islam, 2000).

Datasets

We used four multi-temporal satellite images (Landsat MSS 1972, Landsat TM 1991, Landsat TM 2010, and Landsat OLI 2014) to classify land use and to evaluate forest-cover changes in the Madhupur Sal forest area. The images selected spanned temporal changes at intervals of *ca.* two decades prior to the start of a revegetation program in 2009. The images were collected from the Global Land Cover Facility (GLCF) and USGS Global Visualization Viewer (GLCF, 2014) (Table 1). The Landsat Multispectral Scanner (MSS) had four spectral bands with a spatial resolution of 60 m. The Landsat Thematic Mapper (TM) had seven spectral bands with a spatial resolution of 30 m (except band 6, which had 120-m resolution). The Landsat Operational Land Imager (OLI) had nine spectral bands with a spatial resolution of 30 m (except for band 8, which had a 15-m

Table 1 List of satellite images selected.

Satellite data	Date	Spatial resolution	Bands
MSS (WRS−1, Path 148, Row 43)	23/11/1972	60 m	4,3,2 (NIR, R, G)
TM (WRS−2, Path 137, Row 43)	26/11/1991	30 m	4,3,2 (NIR, R, G)
TM (WRS−2, Path 137, Row 43)	30/ 1/2010	30 m	4,3,2 (NIR, R, G)
OLI (WRS−2, Path 137, Row 43)	30/ 3/2014	30 m	5,4,3 (NIR, R, G)

NIR, near infrared: R, red; G, green. MSS, Landsat Multispectral Scanner; TM, Landsat Thematic Mapper; OLI, Landsat Operational Land Imager

Table 2 Major specifications of MSS, TM, and OLI

MSS		TM		OLI	
Observed band	Wave range (μm)	Observed band	Wave range (μm)	Observed band	Wave range (μm)
Band 4	0.5 to 0.6	Band 2	0.52 to 0.60	Band 3	0.53 to 0.59
Band 5	0.6 to 0.7	Band 3	0.63 to 0.69	Band 4	0.64 to 0.67
Band 6	0.7 to 0.8	Band 4	0.75 to 0.90	Band 5	0.85 to 0.88

MSS, Landsat Multispectral Scanner; TM, Landsat Thematic Mapper; OLI, Landsat Operational Land Imager

spatial resolution). Table 2 lists the band specifications of sensors used in the study (USGS, 2014). All of the images were acquired between November and March, during the dry season. The images were clear and cloud free, and had moderate color contrast for land-use mapping and change detection. Our application of satellite images to land-cover mapping and change detection was supported by a number of national and international studies: Naithani (1990), Rosenholm (1993), Quadir et al. (1998), Islam et al. (2006), Islam (2006), Zaman and Katoh (2011), Nath (2014).

Data processing and analysis

The flow chart in Fig. 2 depicts the sequence used in our research. Multi-temporal sets of remote sensing data were used to categorize land-use classes. The image processing software package ERDAS IMAGINE ver.10 (ERDAS, Inc., Atlanta, GA, USA) and the vector data manipulation software ArcGIS ver.10 (ESRI, Redlands, CA, USA) were used to process, analyze, and integrate spatial data. To make the images comparable, the digital image data were first transformed to a uniform ground coordinate system. Geometric correction was required to avoid geometric distortions; thus, we established an image coordinate system relationship. We first performed image rectification on the Landsat TM 2010 image using the World Geodetic System (WGS) 1984 datum, zone 46 north, which is derived from the Universal Transverse Mercator (UTM) coordinate system. We used 15 well-distributed ground control points (GCPs). Finally, Landsat TM 2010 was rectified to 0.25 pixels (7.5 m) using the nearest-neighborhood method with root square mean errors. Landsat MSS 1972, Landsat TM 1991, and Landsat OLI 2014 images were subsequently rectified to Landsat TM 2010 using image-to-image rectification, and resampled to 30-m pixels using the nearest-neighborhood method. We subsequently generated a false color composite combination of infrared, red, and green to facilitate vegetation recognition; chlorophyll in plants strongly reflects the near infrared. We then generated subset, and selected the areas of interest at the study site.

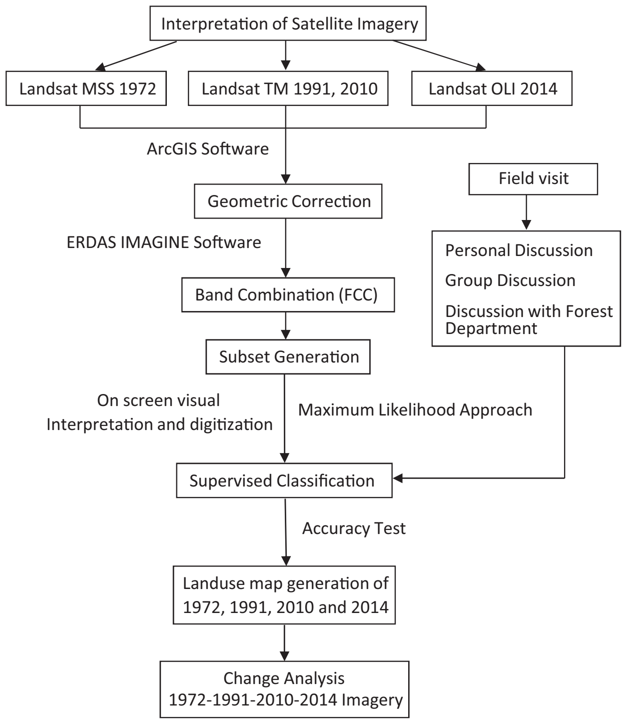

Fig. 2 Flow chart depicting the sequence of procedures used in the study

Finally, we categorized the images using the maximum likelihood classification (MLC) technique of supervised classification approaches; five land-use classes were identified: natural forest, rubber plantation, woodlot plantation, water bodies, and settlements/croplands/other uses. The supervised classification procedure requires training areas for each of the classes identified. These training areas were used to define spectral reflectance patterns/signatures of each land-use category. The signatures were then used by classifiers to group the pixels into a selected category consistent with the spectral patterns.

Table 3 Interpretation key for MSS, TM, and OLI images.

Landsat imagery	Feature	Physical characteristics	Feature color in the images
MSS	Natural forest Rubber plantation Woodlot plantation Water bodies Settlements/croplands/other uses	Tropical moist deciduous forest Rubber tree plantation Native/exotic tree species River, pond, canal Crop/bare land, rural houses with homestead gardens, roads etc.	Dark red with smooth texture – – Dark green with smooth texture Light green, whitish, cyan with medium texture
TM	Natural forest Rubber plantation Woodlot plantation Water bodies Settlements/croplands/other uses	Tropical moist deciduous forest Rubber tree plantation Native/exotic tree species River, pond, canal Crop/bare land, rural houses with homestead gardens, roads etc.	Dark to pinkish red with smooth texture Light red with smooth texture Dark red with medium texture Dark blue with smooth texture Light green, whitish, cyan, red with medium texture
OLI	Natural forest Rubber plantation Woodlot plantation Water bodies Settlements/croplands/other uses	Tropical Moist deciduous forest Rubber tree plantation Native/exotic tree species River, pond, cannels Crop/bare land, rural houses with homestead gardens, roads etc.	Pinkish red with smooth texture Light red with smooth texture Dark red with medium texture Dark blue with smooth texture Whitish, cyan, red with medium texture

MSS, Landsat Multispectral Scanner; TM, Landsat Thematic Mapper; OLI, Landsat Operational Land Imager

We selected 15 homogeneous training areas for each land-use class and used an interpretation key for classification. The interpretation key was created with the aid of visual analysis of the images using displays of red/green/blue (RGB) combinations. The classification was also supported by reference data and ancillary information. The reference data, which were collected from digital forest-cover maps for 1963, 1977, 1991, 2003, and 2008, together with visual interpretations of the images, were used to validate the classified images. The ancillary information was obtained from the digital land-use map, forest-cover maps, Google Earth (Google Inc., Googleplex, Mountain view, California, USA), and the author's prior knowledge of the study area obtained by visiting the site. A 3 × 3 majority filter was finally applied to smooth the classified images. The land use descriptions were as follows:

* Natural forest: Most of the trees in natural forests were indigenous and had recruited naturally. The stands were capable of producing timber or other wood products.
* Rubber plantation: Areas covered with rubber trees (*Hevea brasilensis*).
* Woodlot plantation: Areas covered with native and exotic timber-producing tree species.
* Water bodies: Rivers, lakes, ponds, irrigation lines, and seasonal standing water.
* Settlements/croplands/other uses: Agricultural lands used to produce human food, and bare land devoid of vegetation, such as sand dunes and exposed soil. Settlements were mostly rural housing with homestead vegetation. Other uses included roads, and an air force live bombing ground.

We produced land-use maps for 1972, 1991, 2010, and 2014. The general land use of an area conveyed information on the overall area-based utilization of both natural and cultural resources.

Accuracy assessment was the next step in image classification. It is very important for interpreting and application of the results. We used this procedure to evaluate the quality of each thematic map from a satellite image. The software we used contained an accuracy assessment tool; overall accuracy and kappa statistics were computed. Finally, using the ERDAS IMAGINE attribute table, we calculated the area statistics (in ha) of each land-use category and then compared different classes of land use among years. Thus, we examined changes in forest cover and other land use within the Madhupur Sal forest at decadal intervals over 42 years (1972-1991-2010-2014). Statistically significant differences were detected using Microsoft Excel 2003 software (Microsoft Corporation, Redmond, WA, USA).

Interpretation key

The criterion set by interpretation elements that we used for identification of an object was termed an interpretation key. The key is a set of guidelines used to assist interpreters in rapid identification of features on a remote sensing image. The six primary elements of visual interpretation were tone (or color), size, shape, texture, shadow, and pattern. We developed an interpretation key for the study years by cross-comparing existing maps of the forest cover and the tonal characteristics of false color composite (FCC) imagery. Tone refers to the relative brightness or color of objects captured in an image. Texture is the frequency of tonal change on the image; it determines the overall visual smoothness or coarseness of an image's features. We interpreted images using ERDAS IMAGINE ver. 10 software, with additional information provided by (i) maps made available by Google Earth, (ii) a land-cover map of the Madhupur Reserve Sal forest, and (ii) site visits (site visits included discussions with Forest Department and local people, as well as the authors' personal examination of the forest). Table 3 lists the land-cover characteristics and interpretative remarks.

RESULTS

Land-use classification

The study area was categorized into five land-use classes: natural forest, rubber plantation, woodlot plantation, water bodies, and the "settlements/croplands/other uses." The land use maps for 1972, 1991, 2010, and 2014 are depicted in Fig. 3. Table 4 lists the areas occupied by each land-use category. The settlements/croplands/other uses category

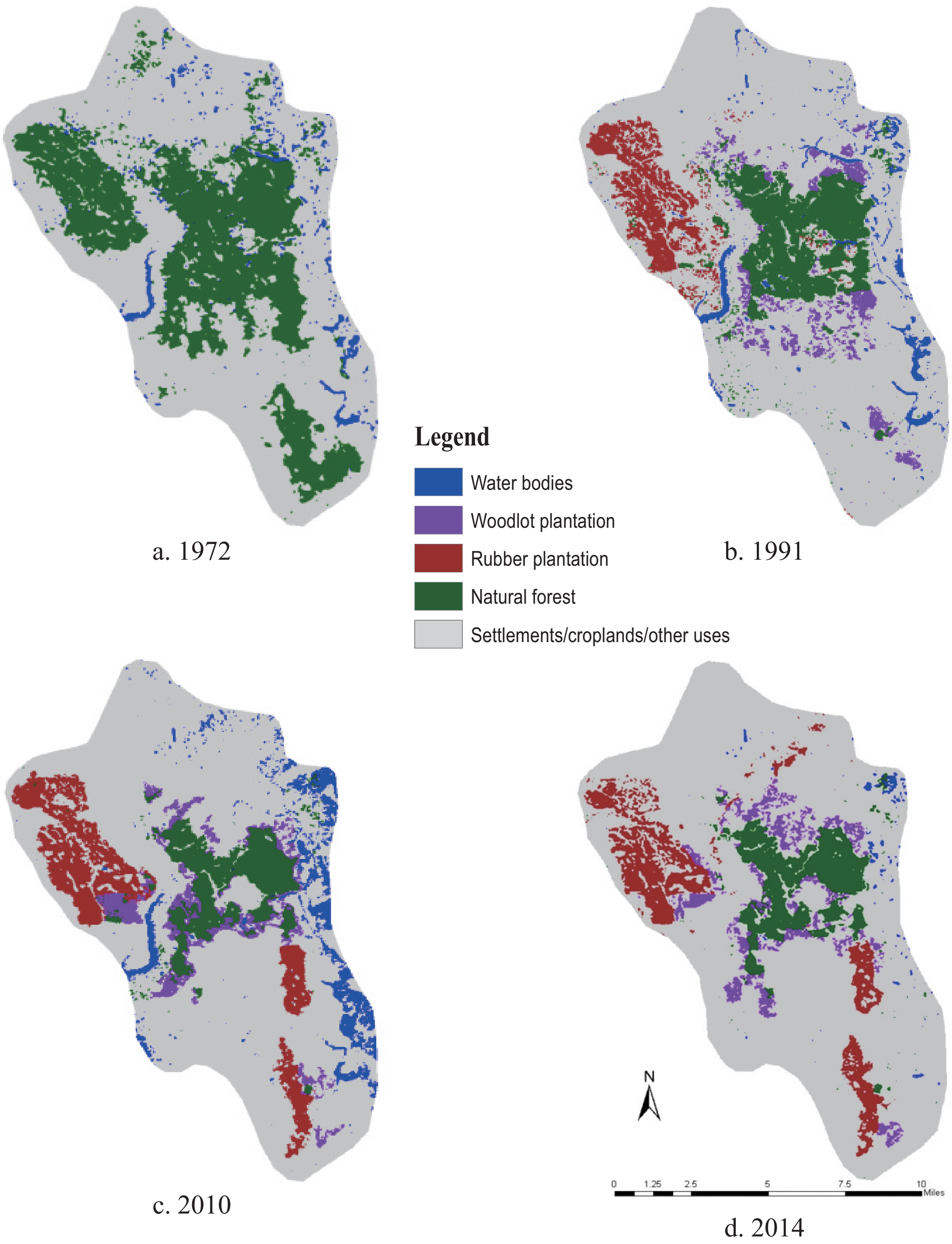

Fig. 3 Classified image maps of the Madhupur Sal Forest area

Table 4 Land areas (ha) and their proportions (%) of the total calculated from maps of different land uses in the Madhupur Sal forest area

Year	Natural forest		Rubber plantation		Woodlot plantation		Water bodies		Settlements/croplands/other uses		Total area
	(ha)	%	(ha)	%	(ha)	%	(ha)	%	(ha)	%	(ha)
1972	9840.3	31.5	–	–	–	–	1114.8	3.4	20267.5	64.9	31222.6
1991	4168.4	13.4	2201.0	7.1	1435.8	4.6	1211.9	3.9	22205.6	71.1	31222.6
2010	2760.9	8.8	2924.4	9.4	1519.0	4.9	2146.9	6.9	21871.4	70.1	31222.6
2014	2963.3	9.5	2778.3	8.9	1537.3	4.9	259.1	0.8	23684.6	75.9	31222.6

occupied the largest proportion of the area throughout the study period, reaching a maximum of 23,684.6 ha in 2014. Natural forest was the second largest category. It occupied 9,840.3 ha in 1972, declining to 2,760.9 ha in 2010. Rubber and woodlot plantations first appeared in 1991 and gradually increased thereafter. The area of the water bodies peaked (at 2,145.9 ha) in 2010, and fell dramatically in 2014.

Accuracy assessment of the classification

Overall classification accuracies and overall kappa statistics obtained for the classified images are listed in Table 5. Kappa coefficient values were aggregated into three groups: (i) values >80%, representing strong agreement; (ii) values of 40-80%, representing moderate agreement; and (iii) values <40%, representing poor agreement (Rahman et. al., 2004). The kappa statistics for the year 1972, for example, indicated that our classification system produced a map in which ≥ 97% of pixels were correctly classified (more than would be expected by random assignment).

Table 5 Accuracy assessment of the classified images

Reference Year	Classified image	Overall classification accuracy (%)	Overall kappa statistics
1972	Landsat MSS	98.3	0.97
1991	Landsat TM	86.4	0.76
2010	Landsat TM	93.2	0.89
2014	Landsat OLI	91.9	0.89

MSS, Multispectral Scanner; TM, Thematic Mapper; OLI, Operational Land Imager

Forest-cover change

The land-use change statistics for the study area are depicted in Fig. 4. Natural forest cover decreased by 18.2% from 1972 to 1991 and by 4.5% from 1991 to 2010, but increased by 0.7% from 2010 to 2014. The absolute and relative changes in natural forest area are summarized in Table 4. In total, 6,877 ha of natural forest cover were lost to other land uses through the study period.

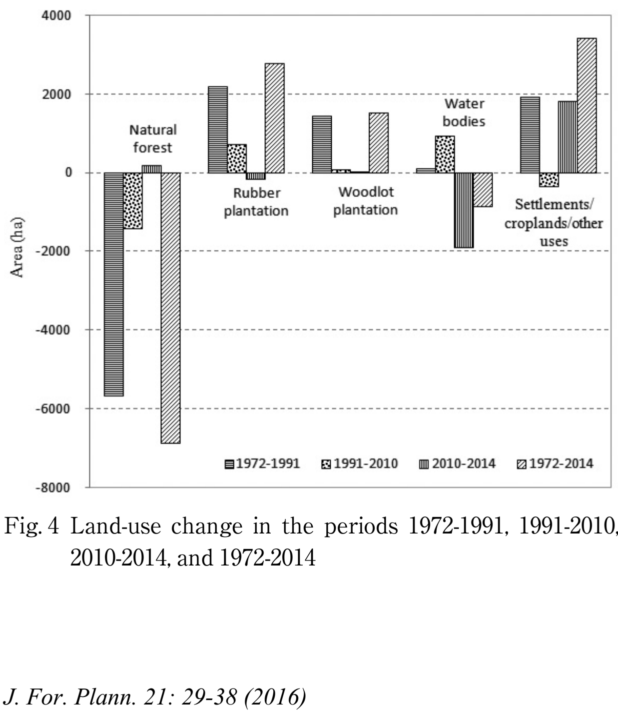

Fig. 4 Land-use change in the periods 1972-1991, 1991-2010, 2010-2014, and 1972-2014

Rubber and woodlot plantation areal changes

The trends in areal occupancy by rubber and woodlot plantations were the reverse of those for natural forest cover. Rubber plantations occurred in the northwestern and southern sectors of the natural forest. Woodlots were located around and within the natural forest. Table 4 summarizes absolute changes in the area occupied by rubber plantations and woodlots through the study period.

Water bodies and settlements/croplands/other uses categories

Most of the water bodies were located in the eastern and western sectors of the natural forest. Their areal extent increased slowly from 1972 to 1991 and more rapidly between 1991 and 2010. Water body area decreased dramatically in 2014 as the areas of natural forest, woodlot plantation, and settlements/croplands/other uses increased. The areas occupied by the settlements/croplands/other uses category were distributed around and within natural forest. Details of areal expansion of this land use category are listed in Table 4.

DISCUSSION

Trend in forest cover changes

The destruction of natural Sal forest began at the periphery of the stand and moved inward over time. The spatial distribution of damage was variable, with most occurring in northern, northwestern, and southern sectors of the forest. The northwestern sector was clear-cut, and most of the logged tract was converted to a rubber plantation in the period 1972-1991 (Fig. 3b). A heavily deforested area in the southern sector was used as woodlot plantation and possibly for other purposes in the period 1972-1991. Between 1991 and 2010, this area was logged over for rubber tree planting. Commercial monoculture rubber planting was introduced in the Madhupur Sal forest area in 1985 (Gain, 1998). Additional extensive damage was caused in and around the natural forest, likely through human population growth, establishment of settlements, firewood shortages, domestic consumption, poor forest management policies, local-level corruption, land encroachment, illegal felling, conversion of forest land for industrialization, and commercial crop cultivation. Islam and Sato (2012) showed that illegal logging and forest land conversion were the ultimate causes of losses in the Sal forests of Bangladesh. They also indicated that illegal logging is a complex phenomenon promoted by local syndicates; they demonstrated that land conversion to diverse commercial activities directly influenced national policies and prevailing attitudes in the country. Settlements, road networks, and other infrastructure for the burgeoning population in the area had negative impacts on the forest and its wildlife, while continually degrading forest resources. Over the past 20 years, both the human population in the area adjacent to the park and the number of illegal houses inside the forest have doubled; most of the people in this increasing population depend on the forest for their livelihoods (Islam and Sato, 2012). The Forest Department has identified at least 5000 households in the Madhupur Sal forest area (GOB, 2010). Some of the department's personnel and a few members of the local elite are indirectly responsible for illegal logging in the Sal forests. Corruption in the department has strongly promoted forest conversion (Gain, 2002). Ineffectual and bureaucratic forest management approaches have had immense negative impacts on most state forests in Bangladesh (Ahmed, 2008). The market demand for timber in the overpopulated nation is among the causes contributing to the depletion of forest resources, including losses to illegal logging. People with inadequate incomes covet the wooden furniture seen in television images or in furniture company advertisements (Ahmed, 2008). Additionally, the live firing range of the Bangladesh Air Force (BFA) occupied a significant area (405 ha) after clear cutting began from the southern perimeter of natural forest, causing losses of canopy cover (Gain, 2005).

To reduce the pressure exerted by local forest-dependent people on the natural forests and to improve the general understanding of forest resource conservation, Bangladesh has introduced a number of forestry management approaches, including agro-forestry plantations, social or participatory forestry, woodlot plantations, Sal coppice management, and buffer zone management. These approaches may have reduced the rate of deforestation in this area after 1991. Participatory, people-oriented forestry activities have operated in the Tangail forest area since 1987 (Muhammed et al., 2008). The woodlot plantation and agro-forestry programs have operated satisfactorily, but the Sal coppice and buffer zone management efforts were total failures (GOB, 2010), and the areas set aside for these projects were occupied by members of the local elite for commercial agriculture enterprises.

Forest cover increased markedly from 2010 through 2014. This is the most optimistic finding of our study. Zaman and Katoh (2011) reported significant increases in forest cover in areas protected by local governments and private owners. They also observed conversion of some croplands into closed forest and open forest tracts in the Thakurgaon forest, located in northern Bangladesh.

A co-management procedure was introduced after 2009 for reforestation by rehabilitation of forest-dependent local and ethnic communities. Conservation measures included the provision of guards, a participatory plantation program in the encroached forest land, and a program to encourage forest-dependent people to seek alternative sources of income. The reductions in external pressures probably promoted natural regeneration on the forest floor, which increased canopy cover. Bangladesh government is committed to increase overall forest cover under many forestry management programs which funded by developed countries. In Madhupur Sal forest, the participatory plantation of adjacent and inside encroached land may be added to the natural forest canopy, which also increased overall tree cover. These are the most likely explanations for increases in forest coverage between 2010 and 2014, postulates that are supported by current literature. Revegetation activities have proceeded apace, even as authorities have ignored many of the illegal activities of forest-dependent people and protected their encroachments; despite this inconsistency, the net effect has been a significant improvement in the vegetation (Islam et al., 2013).

Trends in rubber and woodlot plantations

No rubber or woodlot plantations operated in the Madhupur Sal forest area in 1972. These enterprises began in the 1980s (Fig. 3). Rubber production is an important land-use issue because large tracts of land have been converted from their original natural forest state. At the initiative of the Bangladesh Forest Industries Development Corporation (BFICD), the area under rubber trees increased between 1991 and 2010. Between 2010 and 2014, this area decreased (Table 4), perhaps through encroachment by local people who converted plantations to cropland, mismanagement by authorities charged with plantation protection, or natural calamities. BFIDC (2014) reported that rubber plantations were created in the Madhupur Sal forest in the periods 1987-1989 (2,138.4 ha) and 2001-2002 (1,072.4 ha). Woodlot plantations were established within and around degraded and denuded forest land under the auspices of a reforestation program. This program, which represented an important participatory activity involving local people, was initiated by the Bangladesh Forest Department in 1989 to reduce the anthropogenic pressure on natural forests exerted by the collection of fuel wood, timber, and other resources. GOB (2010) reported that the woodlot plantation enterprise is a dominant operating program among the different participatory management approaches in the Madhupur Sal forest area. Plantation production is a continuous process involving logging and replanting. Spatial and temporal variation in these activities causes variability in the area of land under woodlot cover.

Trends in water body and settlements/croplands/other use categories

The area under water bodies increased slowly through 1991 and then more rapidly until 2010. These changes resulted from continued encroachment into the natural forest. A number of brick kilns built through this period were fired with illegally cut trees. Clay from nearby land was consumed in brick making, and the excavation process may have reduced the topographic elevation. The low-lying tracts contained large volumes of seasonal water in the late rainy and early winter seasons of 2010. On the other hand, as agriculture based country, rice is the major crop in Bangladesh. The only season of high yielding rice production is winter and it starts in January-February. Lot of water as flood irrigation is needed for rice field preparation. Probably, the irrigation water increased the total water volume in 2010.

Over the entire study period, lands in the settlements/croplands/other use category occupied the largest area within the park and continued growing through the time span of the study as the natural forest decreased. The area under seasonal water decreased after 2010, leaving open land, whose area peaked in 2014.

The settlements/croplands/other uses and woodlot plantation categories may have been confused in the analysis due to the plantation patterns. A typical rural settlement consisted of native and exotic tree plantations known as homestead gardens. In some cases, the homestead garden may have been classified as woodlot plantations. The areas of water bodies may have been overestimated due to mixed pixel effects. The extent of marshland varied among seasons depending on the water regime in the area. Furthermore, the minimum mapping unit size that we used precluded the detection of small changes.

CONCLUSION

Forest-cover change through deforestation is a major environmental problem in the Bangladeshi Madhupur Sal forest. In this study, we tracked the recent history of forest cover change in the region. Five classes of land use were extracted from satellite images of the park. The extent of forest cover changed markedly between 1972 and 2014, with radical decreases from 1972 to 2010. The areas of forested lands declined by 5,672.0 ha in the period 1972-1991; they fell by 1,407.4 ha in the period 1991-2010. We observed expansions of rubber plantations, woodlot plantations, agricultural lands, and areas in the settlements/croplands/other use category as the forest cover declined. Across the whole area, 9,840.3 ha of land were covered with forest in 1972, but this value declined to 2,760.9 ha in 2010. This decrease resulted from anthropogenic changes in the study area, but we did not investigate the activities that led to these changes. Forest coverage dramatically increased from 2010 to 2014, reaching 2,963.3 ha. This increase resulted from co-management practice procedures operated jointly by the Bangladesh Forest Department and local forest-dependent ethnic and non-ethnic peoples.

Therefore, to prevent further destruction of forest resources:

* Local people should be encouraged to plant fast-growing trees within their farm boundaries, on their homesteads, and on degraded lands instead of cutting trees in the existing forest.
* Forest-cover monitoring is required to alert authorities to forest changes; forest maps should be well organized and accurate in location and extent.
* Awareness of the forest conservation system should be promoted among local people by the authorities to support the regeneration of the Sal forest.

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Article

Performance of a photogrammetric digital elevation model in a tropical montane forest environment

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ABSTRACT

Digital photogrammetry has advanced to the point where digital elevation models (DEMs) can be derived in full automation from stereo images, offering new opportunities in various fields including forestry. However, the performance and limitations of digital photogrammetry need to be carefully investigated in forest environments where both scientific studies and forest management depend on accurate information. We evaluated the performance of a photogrammetric digital surface model (photo-DSM) derived from small-format aerial photographs over approximately 2000 ha of tropical montane forest in northern Borneo, Malaysia. The accuracy of the photo-DSM was evaluated by using a reference dataset derived from airborne laser scanning (ALS) with an approximate density of 15 pulses/m^2. The vertical accuracy over the total area (18,349,288 pixels) was represented by a mean error of 0.006 m and RMSE of 3.003 m, with 61.1% of all measured heights accurate to within ±1 m, 81.9% accurate to within ±2 m, and 88.7% accurate to within ±3 m. More detailed local accuracy evaluation was conducted at block level: 31 1-ha blocks and one 0.25-ha block located over different forest types and characterized by the mean canopy height (range=8.4−41.1 m) and standard deviation (range=2.0−9.8 m) of the ALS-canopy height model (ALS-CHM). RMSE of the forest blocks ranged from 1.01 to 4.19 m, and this variance in RMSE could be explained by 78.6% of standard deviation of the ALS-CHM. Canopy slope and dark areas also had an effect on the RMSE: in areas of higher canopy slope and in darker areas within the forest blocks, the RMSE increased by up to 8.6 and 5.8 m, respectively. No-data areas accounted for 3.24% in the forest blocks and were also influenced by canopy slope and darker areas. RMSE of non-forest areas was 0.39 m (n=5243 pixels). Research and development on image-matching algorithms (which achieved 86.1% successful alignment of the aerial photographs in our study), cameras, unmanned aerial vehicles, and flight parameters are ongoing; as a result, digital photogrammetry and its capacity for use in various forestry applications are also continuing to improve.

keyword: tropical montane forest, digital surface model (DSM), digital photogrammetry, small-format aerial photographs

INTRODUCTION

The forest canopy is an important subcomponent of the forest ecosystem and plays a major role in many forest processes and functions. For example, it is the interface between the forest and the atmosphere (e.g. Parker et al., 1992), an area of major photosynthetic activity (e.g. Carswell et al., 2000), a biotic habitat (e.g. Erwin, 1983; Kays and Allison, 2001), and a site of ecological interaction within the forest ecosystem (e.g. Nadkarni, 1994). The complexity of forest canopy lies not only its structure but in the various ways to conceptualize and describe it (e.g. Bongers, 2001; Parker and Brown, 2000). One definition of the canopy is the outermost representation of the canopy surface (Bongers, 2001). In the past, three-dimensional descriptions of the forest canopy were the main challenge, especially in terms of physical access (Barker and Pinard, 2001; Lowman and Wittman, 1996) and resource intensiveness of reconstructing a three-dimensional representation (Bongers, 2001). The advancement of remote sensing technologies (i.e. LiDAR, InSAR [interferometric synthetic aperture radar], and photogrammetry) has enabled fully automated

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reconstruction of the forest canopy surface over large spatial areas with high accuracy and highly detailed information, depending on the sensor type and flight parameters (e.g. acquisition height and speed). Among the three techniques available for forest canopy reconstruction, photogrammetry has several advantages, especially in terms of its low cost and the presence of reflectance information (Leberl et al., 2010).

In analytical photogrammetry, the height information is derived manually by using an analytical stereo-plotter to measure the elevation in each cell of an operational grid-typically 2.5 m (e.g. Fujita et al., 2003a; Okuda et al. 2004) or 5 m (e.g. Nakashizuka et al., 1995)—constructed over the image. This technique is resource intensive where study areas are typically limited to 100 ha (e.g.1600 points/ha are required for 2.5 m grid), and this drawback has impeded the use of photogrammetry for large-scale operations in forest areas. The most recent innovation in photogrammetry technology is digital photogrammetry, in which height elevation can be derived fully automatically. Digital photogrammetry technology is made possible by the development of image-matching algorithms, integrated GNSS/IMU navigation system, graphics processing units (GPUs), and digital photography (e.g. Leberl et al., 2010). In addition to the ability to derive height information from stereo-pair images, aerial photographs also provide a range of unique fundamental characteristics such as color, tone, and texture from the reflectance information (Morgan et al., 2010) with high geometric resolution (Kardoš, 2013) and at relatively low cost. Aerial photography also has the ability to collect the same set of data three times as efficiently in terms of person-hours than the conventional field approach (Brown et al., 2005). This leads to more precise analysis of forest structure (Bongers, 2001) and aerial photography is thus used for various forestry applications such as key data sources for forest inventory and land cover classification of accuracy assessment (Imai et al., 2009; Phua et al., 2008; White et al., 2013), wildlife surveys (Vermeulen et al., 2013), and stand parameter estimation (Awaya et al., 2000), including manual stand delineation and visual interpretation of species (e.g. Garzon-Lopez et al., 2013; Valérie and Marie-Pierre, 2006) or semi-automatically when using multi-spectral imagery (e.g. Hirata et al., 2014). It is currently not possible to conduct these types of analyses by InSAR or airborne laser scanning (ALS).

The three-dimensional information derived from aerial photographs offers huge potential, especially when a high-quality digital terrain model derived from ALS is available. Numerous studies have taken advantage of digital photogrammetry in forestry applications, especially in the estimation of forest variables such as diameter at breast height, tree height, basal area, and volume per hectare using photogrammetric height metrics in plot-based (e.g. Bohlin et al., 2012; Järnstedt et al., 2012; Gobakken et al., 2015; Nurminen et al., 2013; Straub et al., 2013; Vastaranta et al., 2013; Wong et al., 2015), and its use is reported to rank second after ALS and better than InSAR and radargrammetry for timber volume estimation (Rahlf et al., 2014). However, the success of using height metrics in plot-based for those purposes does not represent the performance and accuracy of photogrammetric DEMs at pixel resolution level, where this type of information would be important for other detailed studies such as monitoring forest dynamics or forest gap study (e.g. Bongers, 2001; Fujita et al., 2003b). Although several studies have attempted to evaluate photogrammetric DEMs in different environments such as in mountain environments (e.g. Müller et al., 2014), our understanding of the performance of photogrammetric DEMs in complex forest environments, especially in tropical rainforests, remains poor (Miller et al., 2000; White et al., 2013) and more studies are needed to contribute to the robustness of digital photogrammetry technology for forestry applications. Slope (e.g. Müller et al., 2014) and shade (Baltsavias et al., 2008) are reported to affect the performance of digital photogrammetry, and those characteristics often exist in forest environments, especially in the heterogeneous forest structures of primary rainforest.

In this study, we investigated the performance of a photogrammetric DSM (photo-DSM) in the forest environment of a tropical montane forest by using canopy height metrics (i.e. mean and standard deviation [SD] of canopy height of the ALS-derived canopy height model [ALS-CHM]), by using canopy slope and dark areas and by evaluating the relationship between no-data areas of the photo-DSM and both slope and dark-area class. We also discuss the advantages, limitations, and issues of digital photogrammetry in forestry applications.

MATERIALS

Study area

The study area is located in the Ulu Padas forest area (approx. 4°26'N, 115°45'E; Fig. 1) of northern Borneo, Malaysia, inside the *Heart of Borneo* initiative area that, together with the Pulong Tau National park in Sarawak and the Kayan Mentarang National Park in Kalimantan, Indonesia, forms part of an important mountain eco-regional representation of Borneo. The Ulu Padas forest area covers approximately 155,000 ha in the southwestern tip of Sabah bordering North Kalimantan and Sarawak. The area is covered by rugged terrain ranging in altitude from about 1000 to 1900 m. The vegetation of this region consists of several types (i.e. dominant montane oak/chestnut forest with *Agathis* spp., hill dipterocarp forest, stunted montane mossy forest, and high-level swamp forest; SBCP, 1998). Land use consists of both small- and large-scale logging activities by timber companies and local people as well as small-scale farming activities by the local people with some portion remaining as old growth forest with the canopy height reaching up to 60 m. The area covered by the current study is Site 1 (approx. 2000 ha) of the LiDAR and aerial photograph data acquisition flight mission conducted in October 2012. In Site 1, the elevation ranges from 961 to 1895 m, with average slope of 18.6°, and is generally covered by vegetation to a mean canopy height of 22.5 m with SD of 9.0 m (information is derived from ALS data).

Aerial photograph data

The aerial photographs in the October 2012 flight

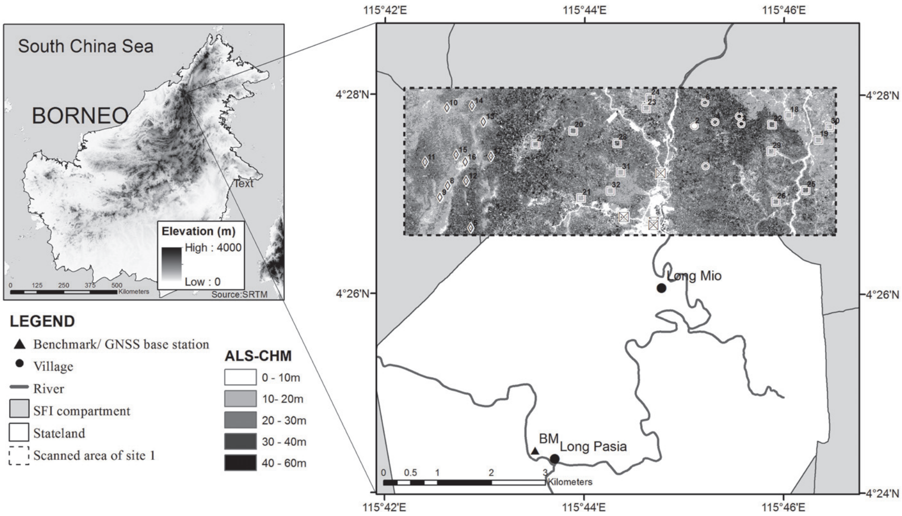

Fig. 1 Location of the study area. Site 1 (large dashed rectangle) represents the site of this study (source of elevation map: Shuttle Radar Topography Mission). Site 1 is enclosed with ALS-CHM and distribution of the forest blocks of upper montane forest (lozenge), lower montane forest (circle), and secondary forest (square), and non-forest blocks (square with cross).

mission were acquired with a small-format digital single-lens reflex camera (Canon EOS-1D Mark III) fitted with a lens with focal length of 28 mm and cross-track field-of-view of 52.9° and along-track field-of-view of 36.6°. The camera was attached to a platform on a Bell 206B3 Jet Ranger helicopter, and the flight mission was conducted over 4 days at an altitude of approximately 400 m above ground level. However, due to the ruggedness of the terrain and also to the strong wind on the data acquisition days, the pilot had difficulty in maintaining the pre-planned flying altitude of 400 m above ground level during the entire flight mission. The flying altitude was calculated to have ranged from as low as 73 m to as high as 791 m with an average of 390 m. Flying at an average speed of 100 km/h and with an exposure interval of 3.5 s resulted in 55−70% of forward overlap and about 45% of side overlap. The camera was set with exposure time of 1/2500 s, ISO-speed of 1250, and aperture range from f/1.8 to f/10. A total of 2400 aerial photographs for Site 1 were delivered in 24-bit sRGB in large-size format of JPEG (3888×2592 pixels) with average GSD (ground sampling distance) of approximately 10 cm, together with GNSS/IMU data information.

ALS data

The LiDAR data were acquired simultaneously with the aerial photographs. The LiDAR system (Riegl LMS-Q560; Riegl LMS GmbH, Horn, Austria) was operated with a 45° of field of view and 240 kHz pulse repetition rate with beam divergence of less than 0.5 mrad. The side overlap was within 30−50%. The processed data were delivered in the coordinate system of the WGS84 UTM Zone 50N / WGS84 ellipsoid in the LAS 1.2 format (for processing information, see Ioki et al., 2014), and the point cloud, comprising 832 million points with an approximate density of 14.9 pulses/m^2, was classified into ground and non-ground class using TerraScan software (Soininen, 2010). The vertical accuracy (RMSE) of the points in the ALS point cloud was estimated to be within 25 cm.

ALS is the best available reference dataset for forest ecosystem evaluation due to its non-clustering effect (Müller et al., 2014) unlike when using ground reference data from DGPS (differential global positioning system) and surveying instrument of total station (e.g. Tonkin et al., 2014). ALS is very accurate to within tens of centimeters; 15 cm for flat forest areas and increasing with terrain slope to a value about 40 cm at a slope of 40% for DTM (Hyyppa et al., 2000).

METHODS

Image-matching process

The aerial photographs were processed with a digital photogrammetry software package employing the structure from motion (SfM) technique (Agisoft Photoscan Pro 1.0.3; Agisoft, St. Petersburg, Russia). SfM technique aims to simultaneously reconstruct three-dimensional scene structure, camera positions and orientations from a set of overlapping photographs (Snavely et al., 2008). The workflow required to produce a dense photogrammetric point cloud consists of two stages (Agisoft, 2014). The first stage is camera alignment, in which image matching is executed to create a sparse point cloud model by searching for common points on the photographs as well as the position of each photograph, and using this information to refine camera calibration parameters. In this process, we used 2400 aerial photographs and the GNSS/IMU data to generate a sparse point cloud of 3,172,874 points covering an area of about 2300 ha. The GNSS dataset collected with 1 hertz was post-processed using differential GNSS technique (Waypoint GrafNet version 8.4; NovAtel Inc., Alberta, Canada). Muller et al. (2014) performed co-registration assessment and demonstrated that offset is <25cm, which is smaller than the

GSD. Of the 2400 photographs, 2067 (86.1%) were successfully aligned. The reported average camera location errors in the *x*, *y*, and *z* directions were 1.82, 0.69, and 0.55 m, respectively, and the total error was 2.03 m. The second stage is building a dense point cloud, where the Photoscan software calculates depth information for each photographs and, based on the estimated camera positions, combines all of the point cloud into a single dense point cloud. In our study, the entire area needed to be divided into two blocks (each approx. 1000 ha) because of the limits of the workstation's processing capability. A dense point cloud with a total of 935 million points was generated, and then converted to LAS format for further processing. The total processing time was approximately 13 hours by using a workstation with the following specifications: Intel Core i5-4670 CPU at 3.40 GHz, 16.0 GB installed memory (RAM), 64-bit OS, and NVIDIA Quadro K2000 GPU.

The whole process was performed in full automation, with us setting only the initial software parameters. For the camera alignment stage, we used the following parameters: high image matching, point limit of 40,000, and with ground control (i.e. GNSS/IMU data), together with optimization of fit aspect, skew transformation coefficient, focal length, principal point coordinates (*cx, cy*), radial distortion coefficient (*k1, k2, k3*), and tangential distortion coefficient (*p1, p2*). For the point cloud densification stage, we used the following parameters: quality of 'high' with advanced option of 'mild'. The settings for these built-in parameters were decided based on evaluation analysis employing forward sequential selection (e.g. Järnstedt et al. 2012) to determine the best settings in a 100 ha test area. This is important; otherwise even a robust matching method would produce an unsatisfactory three-dimensional reconstruction (Remondino et al., 2014).

DSM, digital terrain model, and CHM generation

A photogrammetric digital surface model (photo-DSM) of 1-m pixel resolution was derived by using LAStool in the ArcGIS 10.1 software package (ESRI Inc., Redlands, CA, USA). The maximum value in the point cloud in each 1 m × 1 m pixel grid was used to compute the photo-DSM height (Fig. 2a). The same procedure was applied to derive the digital surface model of the reference dataset (ALS-DSM). ALS points of ground class were used to generate the digital terrain model (ALS-DTM) by using triangulation with natural neighbor interpolation. Then, canopy height models (ALS-CHM and photo-CHM) were derived by subtracting the ALS-DTM from the respective ALS-DSM and photo-DSM. A spatial resolution of 1 m was used because it has been tested and shown to produce high-performance results in the same study area (Wong et al., 2014) and also because it has been used in several other studies (e.g. Bühler et al., 2012; Hese and Lehmann, 2000; Hobi and Ginzler, 2012). We manually masked out problematic areas (264.29 ha) where the aerial photographs did not successfully align during image matching.

Canopy height characterization

Forest canopies are extremely complex, and forest canopy descriptions are difficult to conceptualize (e.g. Parker and Brown, 2000). In this study, we used mean and SD of canopy height (e.g. Hawbaker et al., 2009; Pascual et al., 2010) from the ALS-CHM to characterize the forest canopy structure. Vegetation zones are also complex: inconsistencies in designating zones can be found even in the same mountain (Kitayama, 1992), notwithstanding the *Massenerhebung* effect (Grubb, 1971) in which altitudinal limits can vary with the type of mountain in similar regions, and patchiness can be

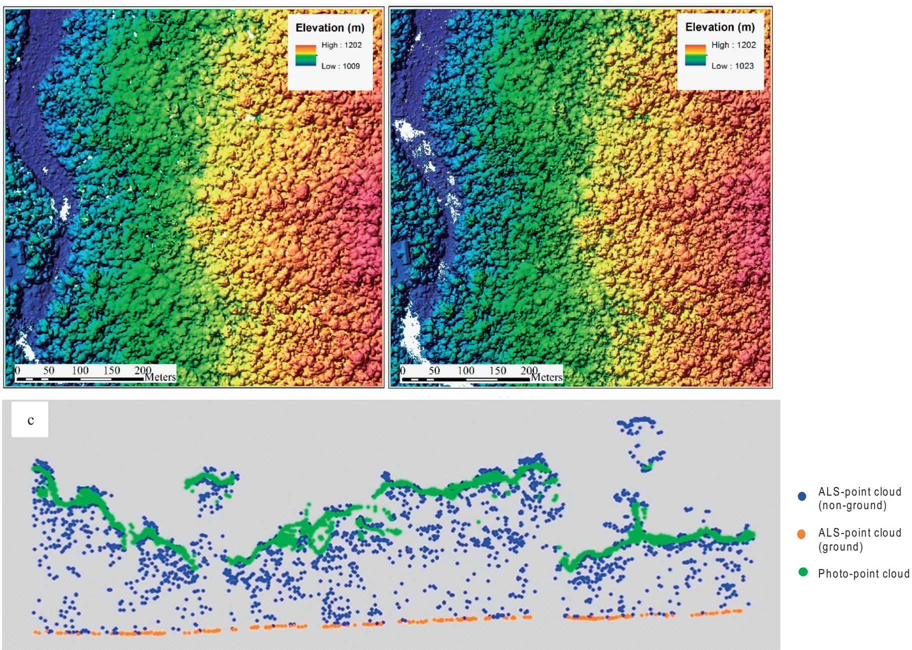

Fig. 2 (a) Photogrammetric digital surface model with shaded relief (1-m pixel resolution); (b) ALS digital surface model with shaded relief (1-m pixel resolution); (c) Cross sectional profile (1 m × 100 m) illustrating the photogrammetric and ALS point cloud.

found in transitional zones (e.g. Pearce, 2006). Pearce (2006) found patches of lower montane forest occurring at altitudes of 950 to 1750 m, while patches of upper montane forest could occur at low altitudes of 1300 m up to summits in similar ecoregional areas. The description of forest type or vegetation zone can be attributed to species composition (e.g. Pearce, 2006) as well as to soil type (Kitayama, 1992). Therefore, due to the limited information of species composition and soil type in categorizing forest type in this area, we arbitrarily defined 1600 m as the altitudinal line separating upper montane primary forest (PU) and lower montane primary forest (PL), and we designated regenerating logged areas and areas of abandoned shifting cultivation as secondary forest (SF) (Fig. 1). By visual observation of the ALS-CHM, we created 32 forest blocks (Fig. 1) of 1 ha each (except for one block which was 0.25 ha) each representing a different mean and SD for ALS-derived canopy height. Upper montane forest in the study site typically consisted of vegetation with a lower mean canopy height (average=21.2 m) and lower SD (average=4.1 m) in the ALS-CHM, whereas lower montane forest had a higher mean canopy height (average=32.3 m) and higher SD (average=6.9 m) in the ALS-CHM (Fig. 3). We identified secondary forests caused by shifting cultivation and logging activities based on fieldwork observations in combination with visual interpretation of the geometrically corrected aerial photo (ortho-photo) and the ALS-CHM. The mean canopy height in each block ranged from 8.4 to 41.1 m and SD ranged from 2.0 to 9.8 m. We also generated evaluation blocks (n=4) from the observation of the ALS-CHM and/or orthophoto for non-forest areas consisting of roads and bare areas of paddy field (square with cross in Fig. 1), so that comparable evaluation with other studies in non-forest areas could be performed. These non-forested evaluation blocks were smaller and varied in size (0.05 to 0.54 ha) because the study area is dominated by vegetation cover.

Canopy slope and dark areas

We derived canopy slope for each pixel from the ALS-DSM, ALS-CHM, and photo-DSM, and classified them in 10° bins (Fig. 4 and Fig. 5b). Because the slope was derived from the canopy top surface instead of using a digital terrain model, we termed it 'canopy slope'. Canopy slope derived from an ALS-CHM (also referred as a normalized DSM [nDSM]) can be influenced by the steepness of the terrain and the crown shape (Khosravipour et al., 2015); therefore, it is recommended to use an ALS-DSM or photo-DSM to derive canopy slope. We generated a RGB color orthophoto mosaic from the aerial photographs with a spatial resolution of 25 cm using Agisoft Photoscan Pro 1.0.3. (Fig. 5a). We then performed PCA transformation of the orthophoto using ArcGIS 10.1., and manually determined the threshold digital number (DN) of PCA component 1 (hereafter PCA1) to differentiate bright areas (PCA1 DN> 205) and dark areas (PCA1 DN ≤ 205). Dark areas were categorized into 7 classes (Fig. 5c). The cumulative contribution of PCA1 was 94.19% and the coefficient for R, G and B were 0.673, 0.596 and 0.439, respectively.

Height accuracy assessment

Height accuracy was assessed by calculating the difference in z-value (Δh_i) between the photo-DSM and corresponding reference data of the ALS-DSM, and then calculating the following statistics at global (overall) and local (block) level:

$$Absolute\ Mean\ Error\ (AME) = \frac{1}{n}\sum_{i=1}^{n}|\Delta h_i| \quad (1)$$

$$Mean\ Error\ (ME) = \frac{1}{n}\sum_{i=1}^{n}\Delta h_i \quad (2)$$

$$Root\ Mean\ Square\ Error\ (RMSE) = \sqrt{\frac{1}{n}\sum_{i=1}^{n}{\Delta h_i}^2} \quad (3)$$

$$Relative\ Root\ Mean\ Square\ Error\ (RMSE\%) = 100 \times \frac{RMSE}{\bar{y}} \quad (4)$$

$$Standard\ Deviation\ (SD) = \sqrt{\frac{1}{(n-1)}\sum_{i=1}^{n}(\Delta h_i - ME)^2)} \quad (5)$$

where n is the number of samples; and $\bar{y}$ is the mean of canopy height in the ALS-CHM.

We further analyzed the relationship between RMSE values and both canopy slope class derived from the ALS-DSM (and also photo-DSM) and dark-area class (PCA1 DN value) in the region of the 32 blocks by using formulae similar to those described above.

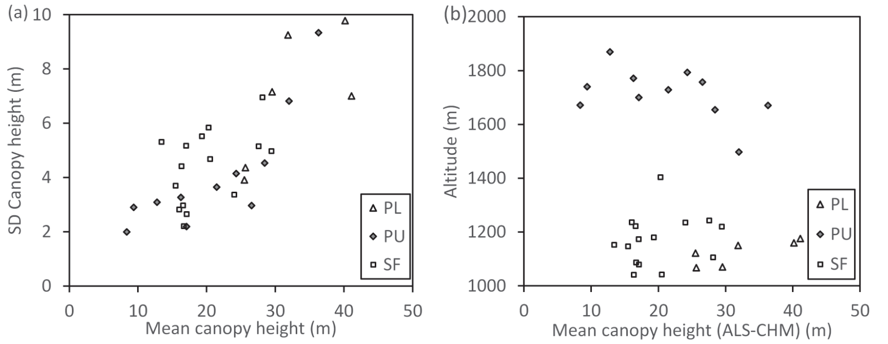

Fig. 3 (a) Forest canopy characterization using SD of canopy height against mean canopy height (ALS-CHM). (b) Altitudinal location of the blocks based on mean ALS-CHM and forest type category. PU, upper montane primary forest; PL, lower montane primary forest; SF, secondary forest.

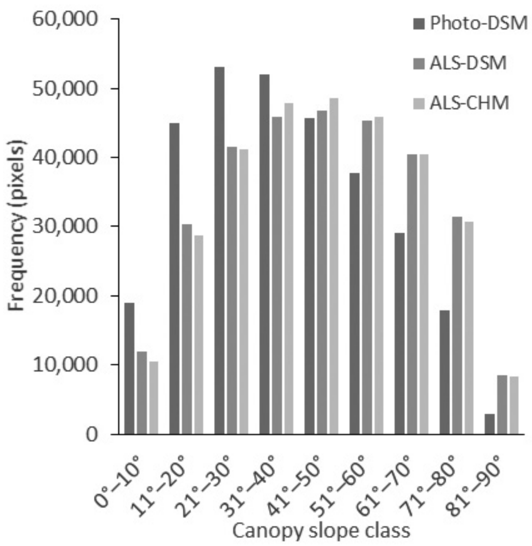

Fig. 4 Frequency of canopy slope derived from photo-DSM, ALS-DSM, and ALS-CHM.

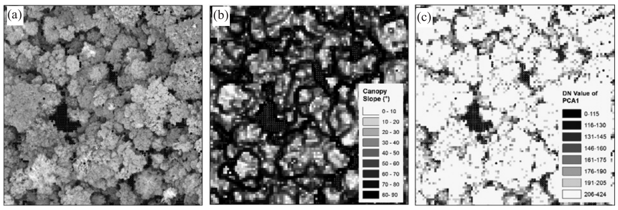

Fig. 5 (a) RGB color orthophoto of Block 4, (b) canopy slope derived from the ALS-DSM, and (c) the same area categorized by brightness class (PCA1 DN value) derived from the orthophoto. Dotted green areas represent the no-data area of the photo-DSM.

No-data areas

We examined the no-data areas in each of the 32 blocks. A no-data area is where no photogrammetric point cloud occur in a 1 m × 1 m pixel of the photo-DSM. We analyzed the relationship between the no-data areas and both the canopy slope class derived from the ALS-DSM and the dark-area class (PCA1 DN value) derived from the orthophoto.

Table 1 Accuracy evaluation statistics for the photo−DSM

	Global	Forest blocks	Non-forest blocks
Mean Error	0.0058 m	−0.0891 m	−0.3152 m
RMSE	3.0032 m	2.5473 m	0.3928 m
AME	1.5160 m	1.3291 m	0.3207 m
Number of pixels (1 m resolution)	18,349,288	302,184	5243
Δh_i			
±1 m	61.09%	63.19%	98.89%
±2 m	81.93%	84.44%	99.71%
±3 m	88.73%	90.77%	99.85%
<−3 m	4.56%	3.97%	0.15%
>3 m	6.71%	5.26%	0%

RMSE, root mean square error; AME, absolute mean error; Δh_i, difference in z-value between the photo-DSM and corresponding reference data of the ALS-DSM

RESULTS

Overall performance of photogrammetric DSM in forest and non-forest areas

The global performance of the photo-DSM over the whole study site revealed that 61.1% of the height values fell within ±1 m of the corresponding reference data of the ALS-DSM, 81.9% fell within ±2 m, and 88.7% within ±3 m (Table 1). Overestimation errors of greater than +3.0 m were at 6.7%, whereas underestimation errors less than -3.0 m were at 4.6%. The mean error, RMSE, and AME was 0.0058 m, 3.003 m, and 1.516 m, respectively. Evaluation of local accuracy in the forest blocks revealed the mean error, RMSE, and AME to be -0.089 m, 2.547 m, and 1.329 m, respectively. The overall

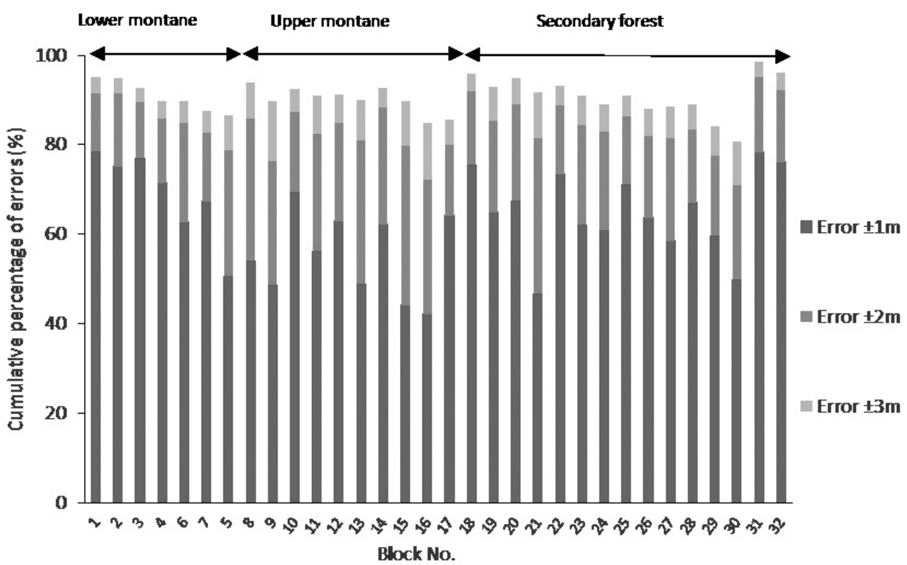

Fig. 6 Errors within ±1 m, ±2 m, and ±3 m at block level over different forest categories. Error ±1 m, errors within -1 to 1 m; error ±2 m, errors within -2 to 2 m excluding error ±1 m; error ±3 m, errors within −3 to 3 m excluding error ±2 m.

RMSE in the four non-forest blocks (i.e. bare land of roads and paddy fields) was found to be 0.393 m with mean error of -0.315 m.

Performance of photogrammetric DSM in forest blocks

The average percentage of error within ±1 m, ±2 m, and ±3 m was 72.0%, 87.6%, and 91.6% respectively for lower montane forest blocks; 54.9%, 81.5%, and 89.8% respectively for upper montane forest blocks; and 65.0%, 84.8%, and 91.0% respectively for secondary forest blocks (Fig. 6). Overestimation greater than 3 m was observed to be highest in lower montane forest blocks (average=5.9%; max=10.1%), whereas underestimation lower than −3 m was observed to be highest in upper montane forest blocks (average=5.2%; max=12.9%). RMSE, AME, ME, and SD of the photo-DSM accuracy in each of the blocks ranged from 1.01 to 4.19 m, 0.71 to 2.09 m, −1.11 to 1.01 m, and 0.97 to 4.08 m, respectively.

Effect of ALS-CHM metrics on RMSE

Our results revealed a linear relationship between RMSE for a block and both the ALS-CHM mean (R^2=0.295) and the ALS-CHM SD (R^2=0.786) for the corresponding block (Fig. 7a & 7b). Higher RMSE of > 3 m was typically observed in blocks with a complex canopy height structure, where ALS-CHM mean and SD of canopy height were greater than 30 and 5 m, respectively.

Effect of canopy slope and dark areas on RMSE

We tested the relationship between RMSE and the canopy slope classes derived from both the ALS-DSM and the photo-DSM. The result demonstrated that the RMSE exhibited an exponential relationship with slope class, ranging from 1.14 m on the gentlest canopy slopes (0−10°) up to 8.63 m on the steepest canopy slopes (81°−90°). The result on RMSE when using canopy slope derived from the photo-DSM was very similar (Fig. 8a). A significant positive relationship between RMSE and canopy slope was observed (Table 2), with the strongest correlation occurring at the > 70° threshold (r= 0.924, P <0.001).

The correlation between dark-area pixels and RMSE for those pixels showed that RMSE was highest for the darkest class (RMSE=5.8 m) and decreased linearly with the brightness class category (RMSE=2 m for the brightest class) (Fig. 8b).

Underestimation and overestimation

Overestimation and underestimation in the photo-DSM were observed to be influenced by the ALS-CHM metrics. Underestimation in the photo-DSM tended to occur in the upper montane forest (where mean and SD of the ALS-CHM

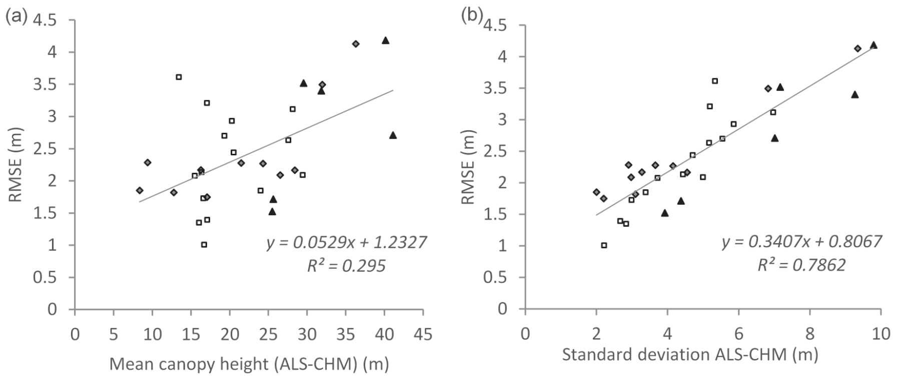

Fig. 7 (a) RMSE vs. mean of ALS-derived canopy height (m); (b) RMSE vs. SD of ALS-derived canopy height (m); (c) RMSE% vs. mean of ALS-derived canopy height; (d) RMSE% vs. SD of ALS-derived canopy height.

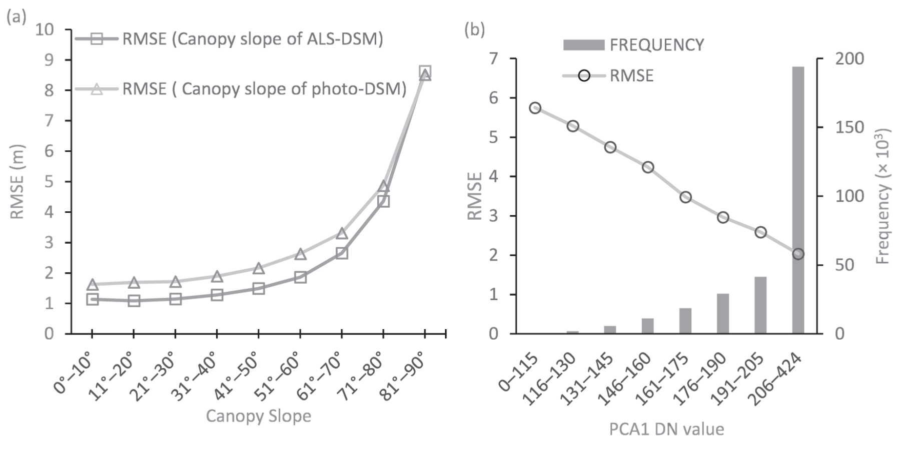

Fig. 8 Relationship between RMSE and (a) canopy slope class and (b) brightness class (PCA1 DN values).

Table 2 Pearson correlation between RMSE and canopy slope class at different canopy slope thresholds.

Canopy slope threshold	Pearson correlation coefficient ALS-DSM	Photo-DSM
>10°	0.6596**	0.4694*
>20°	0.7101**	0.5772**
>30°	0.7750**	0.6419**
>40°	0.8338**	0.7137**
>50°	0.8713**	0.7690**
>60°	0.9028**	0.7875**
>70°	0.9244**	0.7862**
>80°	0.8948**	0.7240**
n	32	

*P-value<0.01; **P-value<0.001.

were lower), whereas overestimation tended to occur in the lower montane forest (where mean and SD of the ALS-CHM were higher) (Fig. 9). Overestimation was observed to occur with higher prevalence in lower montane forest as shown in the scatter plot and cross-sectional profile of a representative lower montane block (Fig. 10c). This was largely contributed to by the limitation of the photo-DSM in identifying forest gaps. Underestimation was observed with higher prevalence in upper montane forest (Fig. 10a), where trees could be missed by the photo-DSM.

Further detailed analysis of the relationship between canopy slope and mean error in each block demonstrated that underestimation of <−0.5 m tended to occur in blocks with a higher proportion of gentler canopy slopes, whereas overestimation of >0.5 m tended to occur in blocks with a higher proportion of steeper canopy slopes (Fig. 11a). Averages of the mean errors were underestimated in gentle canopy slopes as compared to steeper canopy slopes (Fig. 11b). Standard deviation and range of mean error were higher in steeper canopy slopes, particularly those above 60°.

No-data areas

No-data pixels accounted for 3.24% (n=10,136) of the total area of the 32 blocks. At a block level, the percentage of no-data pixels ranged from 0.01% to 9.18%, except for one block (i.e. block no. 17) in which the percentage was exceptionally high (20.91%). We found a moderately strong linear relationship between the percentage of no-data pixels (%) and both ALS-CHM mean (R^2=0.197) and ALS-CHM SD (R^2= 0.359) (Fig. 12).

Effect of canopy slope and brightness values on no-data pixels

The proportion of no-data pixels was plotted against ALS-DSM canopy slope class (Fig. 13a) and against brightness class of the PCA1 component derived from the orthophoto (Fig. 13b). The steepest canopy slope class and the darkest class (0−115 DN) had the largest proportion of no data at 20.17% and 23.58%, respectively. No-data pixels in the photo-DSM constituted 40.79% of the no-data pixels in the ALS-DSM. Fig. 14 shows the cross tabulation of no-data pixels (n=10,136) between canopy slope and dark area. Of the no-data pixels in the gentlest and steepest canopy slope classes, 26.67% and 82.70% were of the dark class category(≤ 205 DN), respectively (Fig. 14a). Of the no-data pixels in the darkest and brightest classes of PCA1 DN values, 69.6% and 38.7% were on high canopy slopes (>70°), respectively (Fig. 14b).

DISCUSSION

RMSEs for individual forest blocks in this study varied from 1.01 to 4.19 m and 78.6% of the variation could be explained by the SD of canopy height, while the mean error ranged from−1.11 to 1.01 m. Hobi and Ginzler (2012) found that RMSE was as high as 7.06 m for a forested area in eastern Switzerland, whereas Næsset (2002) reported that mean tree heights were seriously underestimated by −5.2 to −5.7 m compared to the true mean tree heights of forest stands assessed using ground measurements as a reference dataset over different forest types characterized by species type (i.e. spruce, pine, mixed). Hese and Lehmann (2000) reported that a photo-DSM was found to perform better in areas of beech species (ME=1.45 m; R^2=0.974) than in areas of spruce species (ME=3.29 m; R^2=0.756). Photo-DSMs have also been found to be successfully derived from satellite imagery in forest environments, where larger errors would be normally observed mainly due to lower spatial resolution as compared to aerial photographs, as demonstrated by Baltsavias et al. (2008) using IKONOS images (RMSE=6.61 m) and Hobi and Ginzler (2012) using WorldView2 (RMSE=8.02 m) (Table 3).

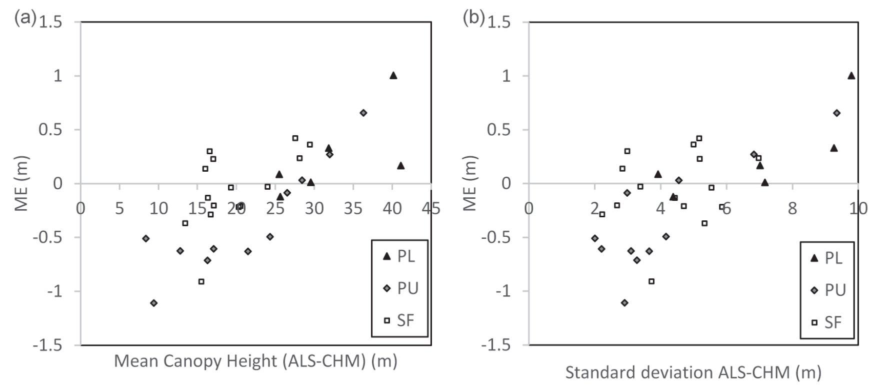

Fig. 9 Mean error (ME) plotted against (a) mean canopy height and (b) SD of the ALS-CHM.

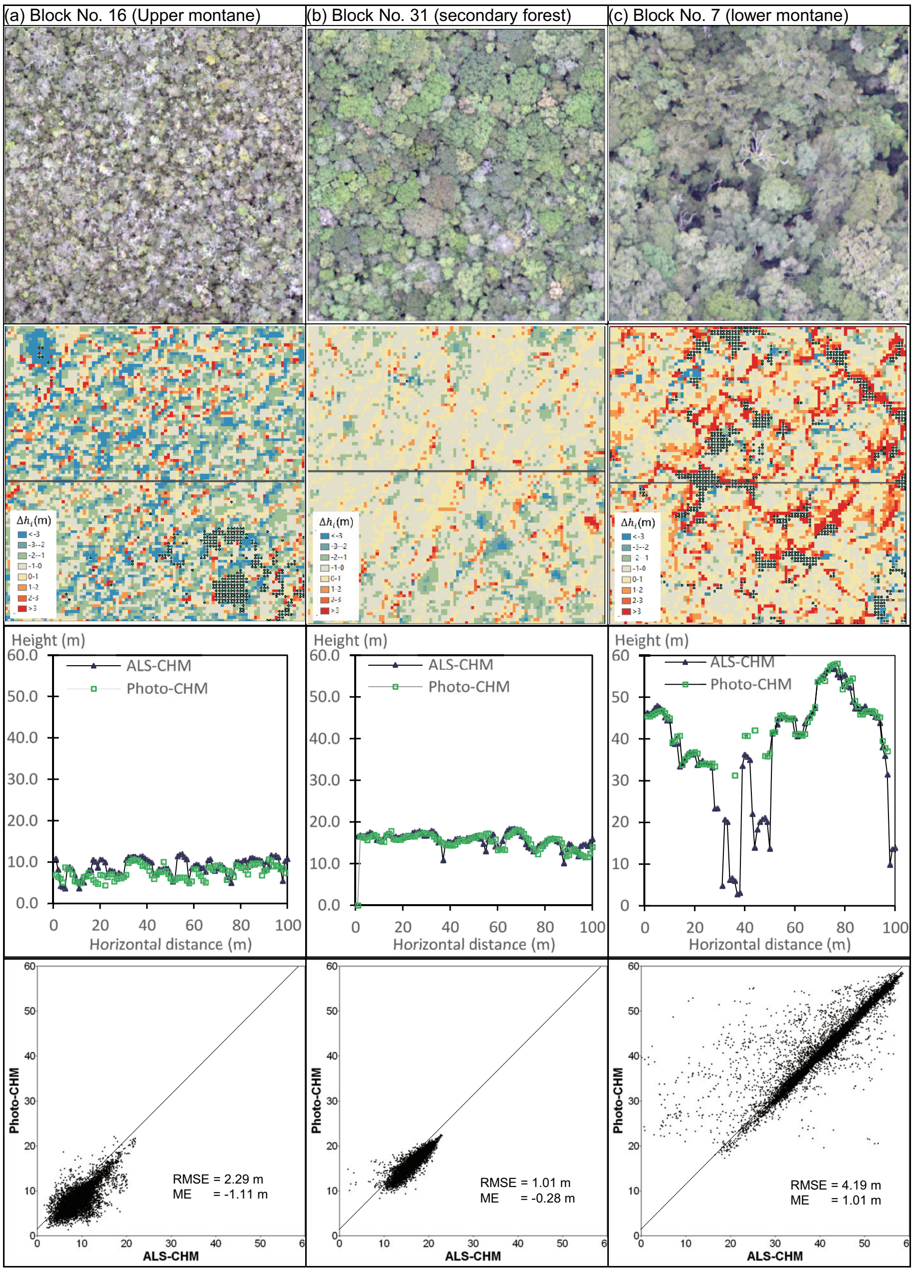

Fig. 10 Color RGB orthophoto (row 1), difference between photo-DSM and ALS-DSM (row 2), cross-sectional profile along the line shown in row 2 (row 3), and scatter plot (row 4) in three selected blocks. In the cross-sectional profile, a tree (A) was missed in the upper montane forest while a forest gap (B) was missed in the lower montane forest. Black dots in row 2 are no-data areas.

Our results demonstrated that SD of ALS-CHM, canopy slope, and dark area can all affect the performance of photogrammetric DSMs. Canopy slope was found to influence the RMSE in steeper areas where a small horizontal distance between two points can result in a very high vertical difference. Our RMSE of less than 2 m for canopy slopes below approximately 50° was similar to the result reported by Müller et al. (2014). RMSE on the steepest slopes could increase to 4 times (Müller et al., 2014) or 4.5 times (Bühler et al., 2012) that of the RMSE on flat areas. Our RMSE was found to increase by almost 8 times from flattest to the steepest canopy slope. Our study revealed that underestimation tended to occur in gentle canopy slopes, where systematic errors could occur during image-matching process even after co-registration (Müller et al., 2014). The variation of the mean error for the photogrammetric DSM can either be observed when using same software package on different datasets (Müller et al., 2014) or using different software packages on same dataset (Remondino et al., 2014; Sona et al., 2014). For non-forest areas (i.e. bare land and paddy fields), the RMSE did not exceed the value of 0.3928 m and the accuracies were comparable to those found in several other studies (Table 3), indicating the consistency of the performance of the photogrammetric point cloud. All the RMSEs in those studies (i.e. Hobi and Ginzler, 2012; Müller et al., 2014; Tonkin et al., 2014) did not exceed 1 m for flat and

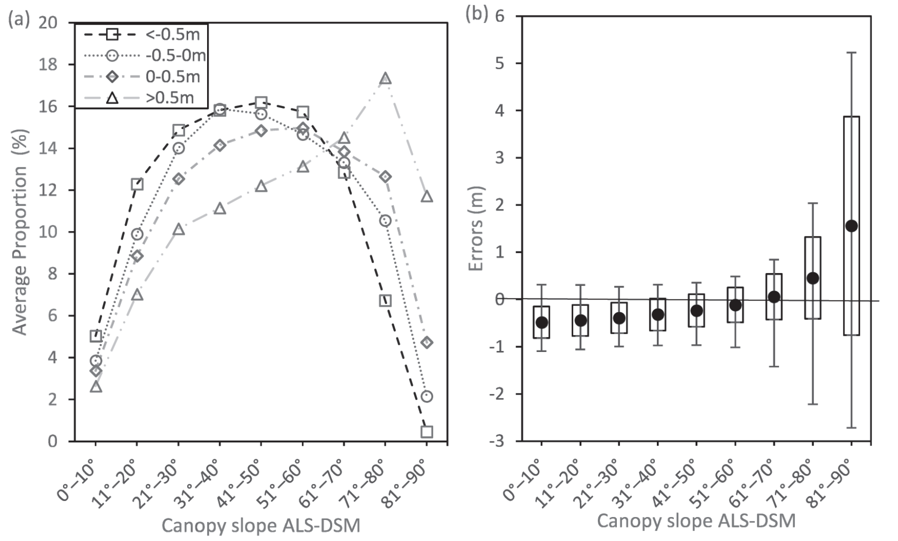

Fig. 11 (a) Average proportion of canopy slope classes (%) in blocks categorized by four mean error classes (<−0.5 m, n=7; −0.5 to 0 m, n=11; >0 to 0.5 m, n=12; >0.5 m, n=2). (b) Averages of mean error are plotted as circles, boxes are SDs about the mean error, and range of mean error is defined by the line indicating the minimum value at the bottom and the maximum value at top (for each canopy slope class, n=32 except for 81°−90°, n=30).

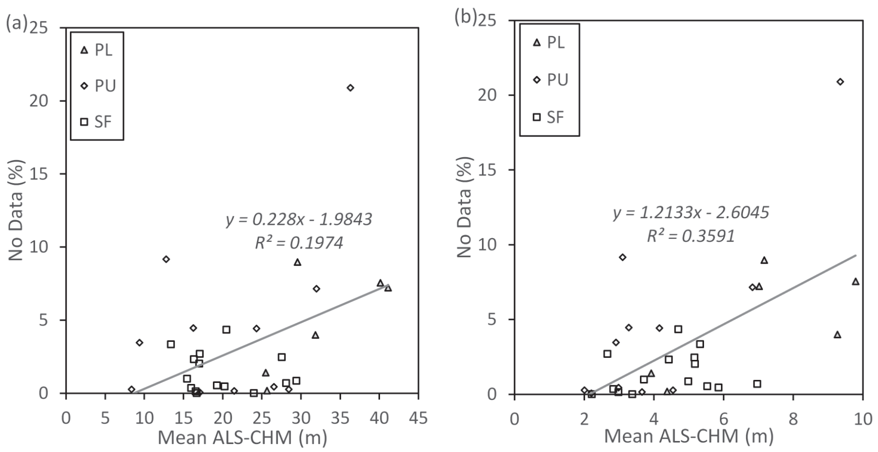

Fig. 12 (a) Percentage of no-data pixels (%) vs. mean canopy height of the ALS-CHM. (b) No-data pixels (%) vs. SD of the canopy height of the ALS-CHM (n=32).

non-forest areas when using very high resolution aerial photographs (GSD<50 cm).

The major limitation to use of digital photogrammetry for forestry applications can be attributed to the image-matching success, height accuracy including forest gap detection, and that the DEM is only limited to the outer forest canopy. Digital photogrammetry is an indirect technique that derives photogrammetric point-clouds by using the image data taken from passive sensors, unlike the direct height measurements of active sensors (i.e. those emitting and receiving their own energy) such as used with LiDAR and InSAR technology. Therefore, successfully deriving high-accuracy photogrammetric point clouds depends on several factors: (1) image-matching algorithm (Remondino et al., 2014), (2) type of camera/sensor (Müller et al., 2014), (3) camera and flight parameter settings (Agisoft, 2014; Mills et al., 2006), (4) overlap rate (Nurminen et al., 2013), (5) environmental conditions (Nex and Remondino, 2014) and (6) object characteristics (Baltsavias et al., 2008; Remondino et al., 2014; Fabris and Pesci, 2005). A combination of these factors could have influenced the results of image matching in our study site, where 13.9% of the aerial photographs failed in the camera alignment process. One feasible suggestion to improve the performance is evaluating whether increased forward overlap of up to 90% (although many recommendations suggest 80% forward overlap) could improve image-matching success and accuracy in forest environment.

The main advantage of the transition from analytical photogrammetry to digital photogrammetry is the capability to derive height in a dense point-cloud form by a fully automated process unlike in analytical photogrammetry

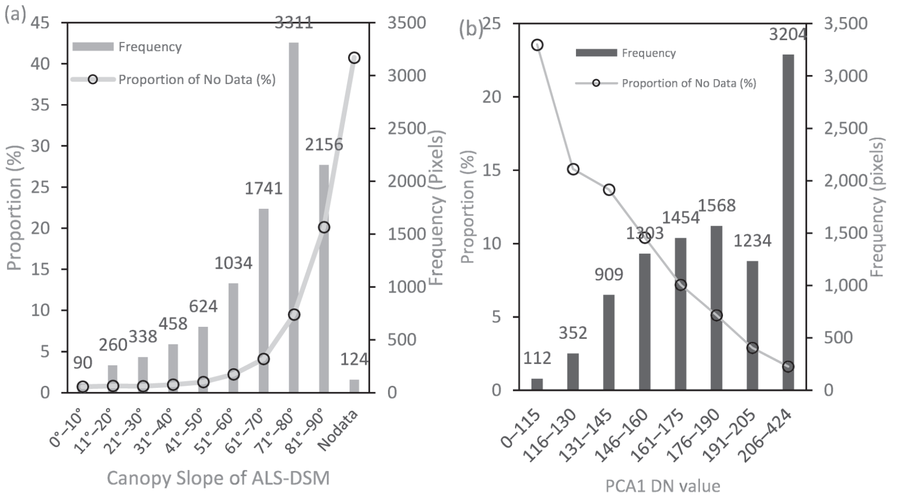

Fig. 13 Proportion (%) of no-data pixels (left *y*-axis) for the particular class based on (a) ALS-DSM canopy slope class and (b) brightness class (PCA1 DN value). Both right *y*-axes represent the frequency of no-data pixels in each class.

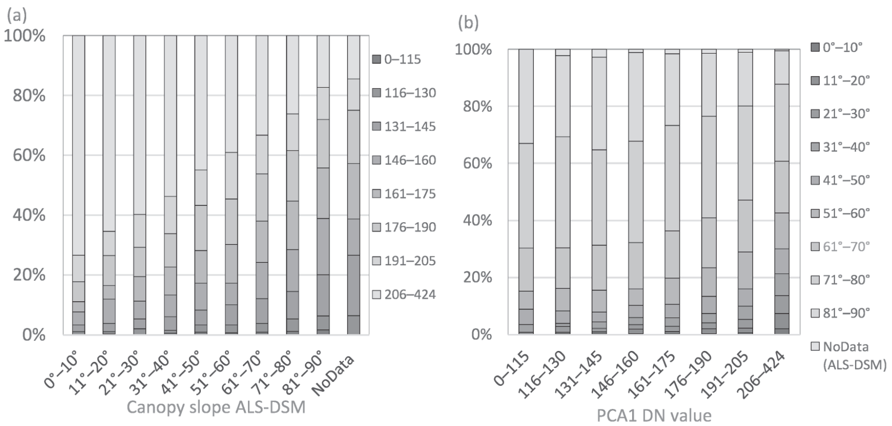

Fig. 14 A 100% stacked bar chart of cross tabulation for no-data areas between brightness class (PCA1 DN value) and ALS-DSM canopy slope class. (a) Proportion of PCA1 DN value class by different ALS-DSM canopy slope class. (b) Proportion of ALS-DSM canopy slope class by different PCA1 DN value class.

where height must be manually digitized. Our study area (2000 ha) is up to 200 times the size of the study by Fujita et al. (2003a), and by using digital photogrammetry, 935 million points were automatically generated. Additionally, digital photogrammetry is relatively cost-effective in comparison with ALS or InSAR technology. Leberl et al. (2010) reported that the effective strip width for aerial photography is up to 5 times the effective strip width for ALS, and that aircraft can be flown at 2.5 times the speed. This means that aerial photography requires only 8% of the time that LiDAR needs to cover an area of similar size. However, consideration on accuracy must be taken when flying with higher altitude which will result to a reduced flying times (i.e. lower cost for flight mission) but increased GSD for a given area. Mills et al. (2006) showed that flying height increased the RMSE as the GSD increases, and there is always tradeoff between cost and accuracy. In addition, the advancement of unmanned aerial vehicles allows cameras to be installed and used for small-scale projects, which will significantly reduce costs of such projects (e.g. Lisein et al., 2013; Tonkin et al., 2014).

CONCLUSIONS

We examined the performance of our photo-DSM in forest area. The RMSE and mean error of our photo-DSM were influenced by metrics of the ALS-CHM (i.e. SD and mean), canopy slope, and dark-area class (derived from PCA of the orthophoto). Standard deviation of the ALS-CHM explained the variance in RMSE by 78.62%. In areas of higher canopy slope, the RMSE increased to 8.63 m, with highest correlation between canopy slope and RMSE occurring at a threshold value of > 70°. The RMSE in dark areas increased to 5.8 m. No-data areas were also influenced by higher ALS-DSM canopy slope and dark-area class.

Our findings will be useful when accurate local information is required in forestry applications such as forest dynamics studies, where understanding the degree of overestimation and underestimation is important. However, further research on integrating the photo-DSM with

Table 3 Performance of photogrammetric DSMs over different forest and non-forest environments

Study	Sensor (spatial resolution)	Reference dataset	Forest accuracy/ type of forest	Non-forest accuracy/ environment
Baltsavias et al. (2008)	IKONOS Pan (1 m)	ALS	6.61 m (RMSE)/ Deciduous (80%) and coniferous (20%) forest	2.05 m (RMSE)/ excluding forest area 1.41 m (RMSE)/ bare ground
Buhler et al. (2012)	ADS80 (25 cm)	ALS	2.55 m (RMSE)/ larch forest	0.82 m (RMSE)/ all 0.5 m (RMSE)/ grassland
Hese & Lehmann (2000)	HRSC-A (30 cm)	Ground measurement (Dendrometer)	1.45 m (ME) 0.974 (R^2)/ beech 3.29 m (ME) 0.756 (R^2)/ spruce	—
Hobi & Ginzler (2012)	WorldView-2 (PAN: 0.5 m/MS: 1.84 m) ADS80 (25 cm)	ALS	7.06 m (RMSE) for ADS80 8.02 m(RMSE) for WV2/ eastern Swiss plateau	0.85 m (RMSE) / grass and herb vegetation
Mills et al. (2006)	ADS40 (15, 20, 25, 30 cm)	ALS	—	0.58 (RMS)/ flat 0.60 (RMS)/ hilly 1.66 (RMS)/ urban
Müller et al. (2014)	HRSC-A (~10 cm) RC30 (~40 cm) ADS40 (~50 cm) ADS80 (~50 cm)	ALS & Geodetic survey	—	1 to 1.3 m (RMSE)/ mountain environment
Næsset (2002)	Agfa Aviphot Pan 200 PE1 (19 cm)	Ground measurement	−5.42 m (ME)/ all −5.68 m (ME)/ spruce, −5.20 m(ME)/ pine −5.31 m(ME)/ mixed forest	—
Tonkin et al. (2014)	Canon EOS-M 18 MP in Uas (<3 cm)	Geodetic (Total station)	—	0.517 m (RMSE)/ moraine-mound complex

Note: Agfa Aviphot Pan 200 PE1 and RC30 are analogue camera system; HRSC-A, ADS40 and ADS80 are multi-sensor pushbroom instrument; Canon EOS-M is a small-format consumer digital camera; WorldView-2 and IKONOS are satellite sensors.

delineation of individual tree crowns will be needed to assess the accuracy at the single-tree level. The 86.1% success rate of fully automatic image matching achieved in our study offers great potential for conducting operational tasks at relatively low cost in large forest areas, such as in estimating changes in forest canopy height over past decades or monitoring forest carbon under REDD+ (reducing emissions from deforestation and forest degradation).

To further develop photogrammetric DEMs for forestry applications and to increase the robustness of this technology, continuous research is needed over several broad topics including study of flight parameters for achieving cost-effective and optimum accuracy, gap detection and height correction, DTM correction, improvement in estimations of forest variables, as well as continuous evaluation in different forest types.

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