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Conservation and restoration of the tropical landscape: Governance and multidisciplinary approaches

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Conservation and restoration of the tropical landscape: Governance and multidisciplinary approaches

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Floral Distribution of a Sub-Bituminous Coal Dumpsite in Enugu, Nigeria

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The remnant floral diversity of a naturally reclaimed area proximal to an abandoned mine was assessed. The coal mine site, the Incident zone (IZ) and an unsullied site, the Control zone (CZ), were sampled. Using a 5 m^2 quadrat, the floristic composition was determined by the occurrence, distribution analysis, and species diversity indices. The study revealed a degraded vegetation type and recorded a total of 60 species, 53 genera and 27 families. Both the IZ and CZ shared 26.67% species similarity, while 36.67% are were unique to each zone. Five species were dominant (*Ageratum conyzoides*, *Panicum maximum*, *Calopogonium mucunoides*, *Chromolaena odorata*). While the dominant genera vary between IZ (*Borreria*, *Dioscorea*, *Ipomoea*, and *Phyllanthus*) and CZ (*Desmodium*, *Euphorbia*, and *Ipomoea*), Asteraceae and Poaceae were the dominant families in both zones. Forbs were the most dominant life forms in both zones; *Cyclosorus* sp. and *Adiantum* sp. were only found on the IZ, whereas, *Kyllinga erecta* and *Mariscus alternifolius* were exclusive to the CZ. Our results reflect that species composition and vegetation paradigm in the study area could be influenced by coal mining, farming, infrastructural installations and climate. Hence, we suggest future studies to investigate how the species adapt to the environment. Although most of the species encountered belonged to lower-risk conservation, the conservation of the species to this area is imperative.

Keywords: sub-bituminous coal, flora, diversity index, conservation status, incident zone, control zone

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INTRODUCTION

Coal is a significant source of energy globally (National Research Council [NRC], 1995; Zou et al., 2016; International Energy Agency [IEA], 2018; Ritchie and Roser, 2019). It is one of the most abundant fossil fuel resources present in Nigeria (Ezekwe and Odukwe, 1980; Ogunsola, 1991), comprising bituminous, sub-bituminous, and lignite belts (Behre, 2006). Its most dominant form in Nigeria, sub-bituminous coal (Chukwu et al., 2016), is characterized by a high calorific value of (5,000–6,000 cal/g or 5,500–6,500 cal/g air-dried), low ash and low sulfur content (Afonja, 1979).

Coal mining, its related activities, including total clearance of vegetation and the indiscriminate disposal of waste coal material such as coal spoils and discards, which are usually stockpiled in the form of dumps, cause land degradation (Singh et al., 2002; Ghose, 2004; Ekka and Behera, 2011), thus, inducing changes in soil structure, typically the physical, chemical and microbiological properties of soil, as well as the destruction of vegetation which ultimately disrupt the functioning of ecosystems (Kundu and Ghose, 1997; Sheoran et al., 2010).

Due to plants' inability to move, they have to deal with changes in their environment (Sanchita, 2018). Environmental changes due to mine waste cause loss of biodiversity (Sheoran et al., 2010). Under a narrow range of conditions in the environment, plant community structure depends on plants' capabilities to develop certain adaptive morphological and physiological features (Dazy et al., 2009).

The loss of vegetation due to coal mining activities has become topical (Sarma, 2005; Huang et al., 2014). Perhaps, these activities may have influenced the concurrent colonization of the coal dumpsites by different invasive plant species, which resulted in the continuous disruption of vegetation. Previous studies have recorded several invasive plant species present in the coal mine dumpsites undergoing rehabilitation in different parts of the world, e.g., Columbia, India, and Indonesia (Sarma, 2005; Hazarika et al., 2006; Sánchez-Pinzón et al., 2010; Ekka and Behera, 2011; Singh, 2011; Komara et al., 2016; Novianti et al., 2017; Yusuf and Arisoesilaningsih, 2017; Hapsari et al., 2020). Together, those studies identified associated plant species such as *Adiantum* sp., *Ageratum conyzoides* (Mill.) M. Sharma, *Alternanthera sessilis* L., *Chromolaena odorata* (L.) R. M. King and H. Rob., *Ipomoea cairica* (L.) Sweet, *Oldenlandia corymbosa* L., *Panicum maximum* Jacq., *Pennisetum purpureum* Schumach., *Tridax procumbens* L., and *Urena lobata* L. to coal mine dumpsites. To our knowledge, there are no detailed studies documenting the plant species inventories of coal mining sites in Nigeria, especially Enugu (the Nigerian Coal City that used to be home to Iva Valley, Ogbete, Onyeama, and Okpara mines (Sikakwe et al., 2015; Agbalagba and Uzo, 2018). Nonetheless, there is an environmental impact assessment (EIA) report which highlighted the effects of coal mining activities on biodiversity, especially on economic tree species (e.g., *Khaya ivorensis* and *Milicia excelsa*) in Akwueke and Iva Valley communities within the Enugu Coal City (Ogbonna et al., 2015). Other previous studies focused instead of floral composition on the geo-environmental characteristics of the Okpara, Onyeama, and Ribadu coal mine sites (Sikakwe et al., 2015; Sikakwe, 2017).

Drawing from the lessons of the occurrence of the devastation by coal mining on the ecosystem (in abandoned mined lands) and its subsequent revegetation from different regions of the world (Baig, 1992; Skousen et al., 1994; Sheoran et al., 2010; Yang et al., 2015; Pauletto et al., 2016; Buta et al., 2019), it was essential to examine how coal mining and its concomitant wastes have impacted the floral distribution in the Coal Camp/Ogbete area. We evaluated the plant diversity by (1) comparing the remnant species in the polluted coal site (IZ) and an undisturbed site (CZ) and by the assessment of the conservation status of the species within the study area.

MATERIALS AND METHODS

Study Area

The study was carried out around the defunct coal mines in the Coal Camp located at Udi (6° 25′ N, 7° 28′ E), Enugu State (**Figure 1** and **Table 1**) between 2014 and 2016 (Nsa et al., 2017). The map for the study site was generated with the ArcGIS software (Environmental Systems Research Institute [ESRI], 2014). Enugu has different vegetation types ranging from the tropical rainforest in the south (where Udi is) to open grassland and savannah in the north (Obi, 2009). The site is surrounded by hills and disturbed secondary regrowth forest vegetation comprising trees, shrubs, and herbs. Although the decline in mining began over 40 years ago, the operations were stalled entirely around 2005. The project area has experienced infrastructural development, urbanization, and farming activities over this period. The occupants of the study area were farmers, former workers of the defunct mine, and traders. For this study, the area near the coal mine, visibly contaminated with coal wastes, was referred to as the Incident zone (IZ), and an uncontaminated area referred to as the Control Zone (CZ). The three criteria considered prior to site selection were: (i) sampling points cover and spread, (ii) species heterogeneity, and (iii) coal mining sites.

Climatic Data

The climatic data for the site covering the period from January 1, 2014, through December 31, 2016, were obtained from two sources; Power SRB, NASA (https://power.larc.nasa.gov/; last accessed 2020/10/10; Briggs et al., 2003) and Nigerian Meteorological Agency (NIMET). The Global Positioning System (GPS; Garmin MAP 64S) coordinates of the plot in the Coal Camp were used to retrieve the precipitation data from NASA, whereas NIMET rainfall data covered the entire Enugu region (Chukwuike, 2018).

Sampling

Nine geo-referenced sampling points (IZ = 7 and CZ = 2) were systematically established for biodiversity assessment (**Figure 1**). The limited sampling points were attributable to acute environmental degradation and habitat fragmentation due to patchiness within the IZ. Two points (about 5 km away) free of mining activities outside the IZ were assessed to compare species presence and absence. Biodiversity assessment was conducted using a 5 m^2 quadrat sampling method (Barker, 2001; Unanaonwi and Amonum, 2017). The choice of the quadrat size was determined with consideration of the landscape of the study site. The points with heterogenous populations were selected for adequate representation of the different species. The coverage, abundance, frequencies of the species and life-form classes within the sampling areas were recorded.

Species Identification

Taxa identification and classification were made onsite, while representatives from indeterminate species were collected and preserved as described by Radford et al. (1974). The two

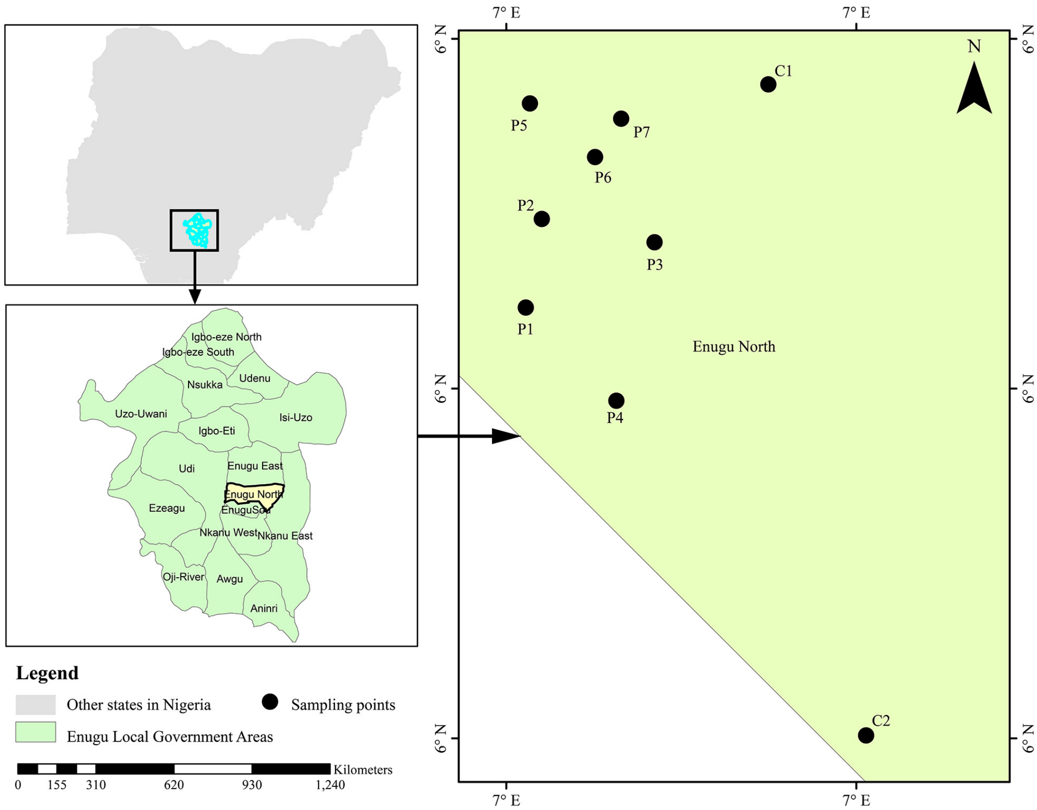

FIGURE 1 | Map of the study area.

TABLE 1 | Climate data (NASA and NIMET) and onsite temperature.

Months	Climate NASA (SRB)									Climate (NIMET, 2017)		
	2014			2015			2016					
	Precipitation (mm)	Temp. (°C)	Relative humidity	Precipitation (mm)	Temp. (°C)	Relative humidity	Precipitation (mm)	Temp. (°C)	Relative humidity	Year	Rainfall (mm)	Temp. (°C)
January	14.19	25.4	73.76	0.98	23.55	64.49	4.12	24.7	55.82	2014	1875.3	28
February	24.47	26.31	79.55	18.24	26.71	81.85	3.66	27.2	64.29	2015	1862.3	27.4
March	39.03	26.52	84.82	92.92	26.71	82.02	109.54	27.74	81.99	2016	1823.4	29.1
April	107.36	26.48	85.41	46.42	26.87	82.6	70.76	27.51	83.76			
May	120.3	26.35	88.16	98.61	26.41	86.6	152.77	26.62	86.4			
June	104.38	25.85	88.87	165.3	25.57	88.4	159.97	25.6	87.46			
July	128.6	25.26	88.99	172.7	25.42	87.85	220.36	25.01	88.64			
August*	185.84	24.8	87.97	106.02	25.29	87.82	310.15	24.86	89.39			
September	175.85	24.94	89.83	226.64	25.3	89.05	294.03	25.03	89.65			
October	132.26	25.45	88.33	157.9	25.48	88.89	229.41	25.7	88.14			
November	43.74	25.93	86.89	81.14	26.01	80.6	44.18	26.49	82.14			
December	0.3	25.43	79.41	0.2	22.96	58.43	8.85	25.22	72.71			
Total	1727.24			1076.32			1607.82					
Average (°C)		25.72	85.18		25.51	81.51		25.96	80.9			

**Month of sample collection August.*

approaches used for the species identification were based on: (1) the morphological description in Flora, manuals and monographs (Hutchinson and Dalziel, 1954, 1958, 1963, 1968; Alston, 1959; Keay et al., 1964; Hutchinson et al., 1972; Lowe and Stanfield, 1974; Akobundu and Agyakwa, 1998) and (2) the comparison of the plant specimen samples with the preserved dried samples at the University of Lagos Herbarium (Department of Botany), Nigeria. These identification methods have been used in previous studies (Adeniyi et al., 2016; Adeonipekun et al., 2018). Identification of species was delimited to the genus level for species that had incomplete morphological features. The plant families followed the Angiosperm Phylogeny Group (The Angiosperm Phylogeny Group [APG], 2016), while the species nomenclature and authorities were validated by the International Plant Name Index database (International Plant Names Index [IPNI], 2020). All plant samples collected were used solely for identification purposes.

Data Analysis

Species presence and absence were recorded for each sampled plot, followed by the comparative floristic analyses of the two zones (IZ and CZ) using four descriptive statistics and diversity indices (Patel et al., 2012, Patel et al., 2020; Oyebanji et al., 2020). The descriptive statistics include frequency (%), relative frequency (%), density (plant/m^2), and relative density (%) while the diversity indices were performed using Dominance, Evenness, Shannon—Wiener, and Simpson's indices (Simpson, 1949; Shannon and Weaver, 1963; Magurran, 1988) incorporated in Paleontological Statistical software (PAST v 2.17c, Hammer et al., 2001). The five-category life forms (climber, forb, grass, shrub, and tree) typical of the tropics were adopted to describe the forest morphology of the study area based on the classification scheme of Mueller-Dombois (1972).

Conservation Status Assessment

The conservation status of the species was classified as Not Evaluated (NE), Data Deficient (DD), Least Concern (LC), Near Threatened (NT), Vulnerable (VU), Endangered (EN), Critically Endangered (CR), Extinct in The Wild (EW), and Extinct (EX) based on the International Union for Conservation of Nature (IUCN) Red List of Threatened Species (IUCN, 2021). The species without complete taxonomic identification could not be assessed and were designated as Not Determined (ND).

RESULTS

Ecosystem and Habitat Characteristics

In the 3 years reported, the average precipitation rate was highest in 2016 (**Table 1**). The habitat ranged from degraded secondary forests to patches of flood plains. Some species are known to occupy flooded areas. Based on the IUCN's Habitat and Ecology ranking, 10 wetland (inland) species were present (*Alternanthera sessilis* L., *Ageratum conyzoides* (Mill.) M. Sharma, *Colocasia esculenta* (L.) Schott., *Commelina diffusa* Burm. F., *Eleusine indica* (L.) Gaertn., *Kyllinga erecta* Schumach., *Ipomoea cairica* (L.) Sweet, *Imperata cylindrica* (L.) Raeusch., *Pennisetum pupureum* Schumach., *Urena lobata* L.) and *Oldenlandia corymbosa* associated with the Artificial/Aquatic and Marine habitat. Other wetland species encountered without an IUCN status include *Sida acuta* Burm. fil., *Desmodium triflorum* (L.) DC., *Centrosema pubescens* Benth., *Calopogonium mucunoides* Desv., *Peltophorum pterocarpum* (DC.) K. Heyne, *Euphorbia heterophylla* L., *Borreria ocymoides* (Burm. F.) DC. *Panicum maximum* Jacq., and Axonopus *compressus* (Sw.) P. Beauv.

Overall Species Distribution

The study revealed a total of 60 species belonging to 53 genera and 27 families in the sampled area (**Figures 2**, **3A,B** and **Table 2**). The environment was dominated by two angiosperm families, Asteraceae and Poaceae, each represented by seven species. The genus *Ipomoea* L. (n = 3 species) and five other genera [*Borreria* G. Mey, *Desmodium* Desv., *Dioscorea* Plum. Ex L., *Euphorbia* L., and *Phyllanthus* L. (n = 2 species each)] were dominant. The distribution of the 60 species into the life-form categories was as follows: forb (27 species), climber (12 species), grass (9 species), shrub (7 species), and tree (5 species) (**Figure 3C**).

Species Diversity Comparison Between IZ and CZ

Comparatively, each zone had 38 species spread across genera (*IZ* = 34 and *CZ* = 35) and families (*IZ* = 18 and *CZ* = 19) (**Figure 3A** and **Table 2**). The dominant species within the IZ were *Ageratum conyzoides* (Mill.) M. Sharma (frequency = 77.78%, *R.F* = 10.14%, density = 0.28 m^2, and *R.D* = 10.29%), *Panicum maximum* Jacq. (frequency = 66.67%, *R.F* = 8.70%, density = 0.24 m^2, and *R.D* = 8.82%), *Calopogonium mucunoides* Desv. (frequency = 55.56%, *R.F* = 7.25%, density = 0.2 m^2, and *R.D* = 7.35%), and *Chromolaena odorata* (L.) R. M. King and H. Rob (frequency = 55.56%, *R.F* = 7.25%, density = 0.2 m^2 and *R.D* = 7.35%).

In both IZ and CZ, *Ipomoea* (n = 2 species each) was the dominant genus. Three genera (*Borreria*, *Dioscorea*, and *Phyllanthus*) and two genera (*Desmodium* and *Euphorbia*) each represented with two species were dominant in both zones.

Twenty-two species were exclusive to each zone, while 16 species were common to both zones; *A. conyzoides*, *Asystasia gangetica* (L.) T. Anderson, *Axonopus compressus* (Sw.) P. Beauv. *Brachiaria deflexa* (Schumach) C.E. Hubb. ex Robyns, *C. mucunoides*, *C. odorata*, *C. esculenta*, *Desmodium triflorum* (L.) DC., *Emilia coccinea* (Sims) G. Don, *Ipomoea cairica* (L.) Sweet, *Manihot esculenta* Crantz, *O. corymbosa*., *P. maximum*, *Setaria barbata* (Lam.) Kunth, and *Synedrella nodiflora* (L.) Gaertn., and *Urena lobata* L. (**Table 2** and **Figure 3D**).

The floristic inventory revealed that there were eight families (Commelinaceae, Costaceae, Dioscoreaceae, Loganiaceae, Melastomaceae, Passifloraceae, Pteridaceae, and Thelypteridaceae) and nine families (Anacardiaceae, Cucurbitaceae, Cyperaceae, Lamiaceae, Onagraceae, Piperaceae, Portulacaceae, Ulmaceae, and Urticaceae) unique to the IZ and CZ, respectively; the zones had ten families in common. The shared families were Acanthaceae, Amaranthaceae, Araceae, Asteraceae, Convolvulaceae, Euphorbiaceae, Fabaceae, Malvaceae, Rubiaceae, and Poaceae (**Figure 3B**). Among these, Asteraceae and Poaceae were dominant in the IZ and CZ, represented with six and five species, respectively (**Table 2**). All the life-form categories were found in the IZ and CZ (**Figure 3B**).

The diversity indices revealed that Plot 2 had the highest diversity, as confirmed by Simpson, Shannon and Margalef values of 0.09, 2.66, and 4.35, respectively. In contrast, Plot 4 had the highest Dominance (0.50), Evenness (∼1.00), and equitability values of (∼1.00). The diversity indices for the pooled

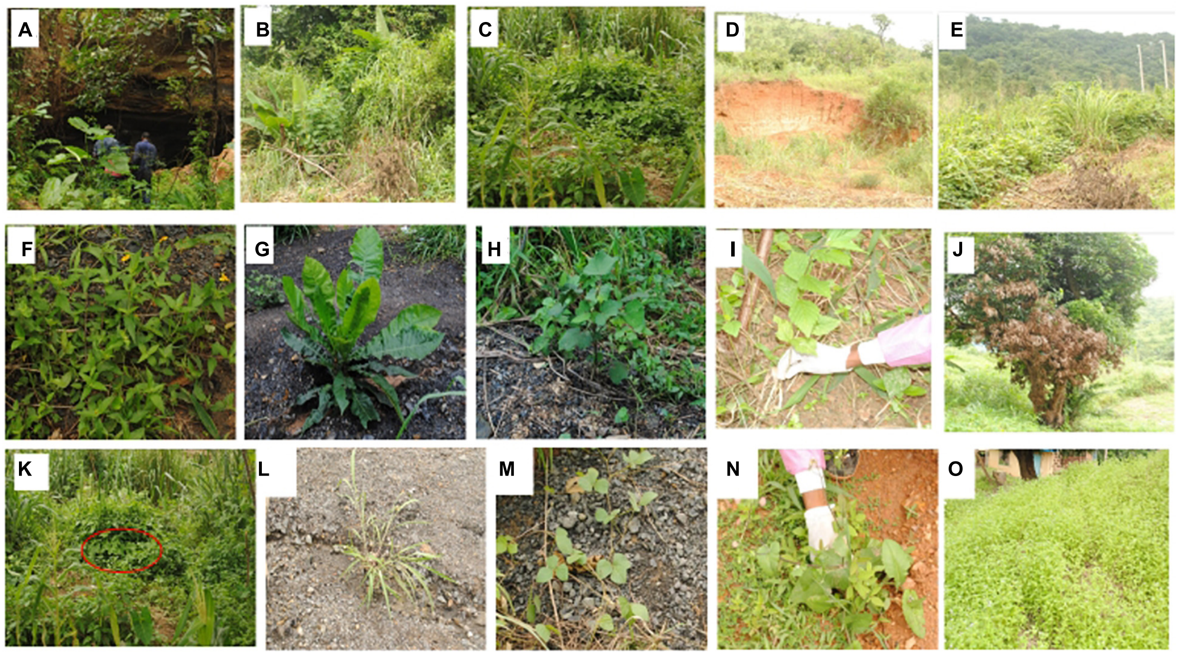

FIGURE 2 | Vegetation view and representative species of the IZ and CZ. **(A)** Coal cave. **(B,C)** Disturbed herbaceous vegetation on the coal dumpsite, **(D)** Hill surrounding area. **(E)** Physiognomy of the control zone vegetation. **(F)** *Aspilia africana* [IZ]. **(G)** Juvenile *Anthocleista vogelii*. **(H)** *Urena lobate*. **(I)** *Sida acuta*. **(J)** *Mangifera indica*. **(K)** Twinning *Luffa cylindrica* (red circle) around other species. **(L)** Juvenile *Panicum maximum*. **(M)** *Calopogonium mucunoides.* **(N)** *Emilia coccinea* and **(O)** Lush of *Ageratum conyzoides.* Photo credits: by I. Y. Nsa, O. O. Oyebanji, G. U. Anorue, and A. A. Odunsi.

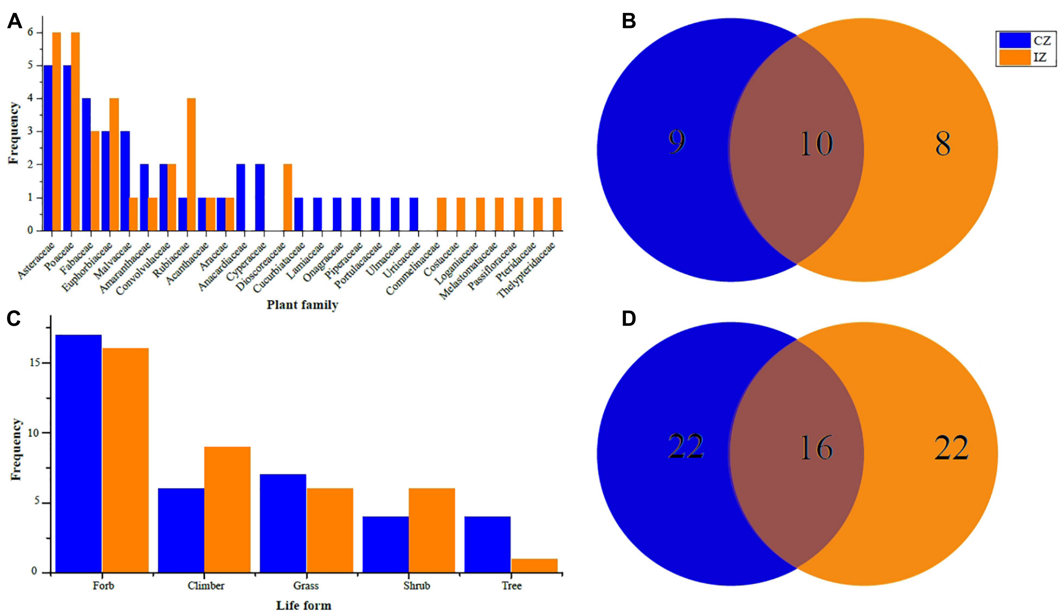

FIGURE 3 | Comparative Species distribution of the study area. **(A)** Family, **(B)** number of unique families, **(C)** life form, and **(D)** the number of unique species.

CZ were as follows: Dominance index (0.36), Simpson (0.64), Shannon—Wieners (1.25), Margalef (1.13), Evenness (0.70), and equitability values (0.77) (**Table 3**).

The floral composition also included agricultural species [*Abelmoschus esculentus* (L.) Moench., *Colocasia esculenta* (L.) Schott., *Dioscorea* sp., *Manihot esculenta* Crantz]. The frequently encountered species outside the quadrat were *Anacardium occidentale* L., *Ficus* Tourn. ex L. sp. and *Lantana camara* L. in the CZ and *Cyperus* L. sp. and *Hyptis suaveolens* (L.) Poit. in the IZ.

TABLE 2 | Species diversity, occurrence, and conservation status in the study area.

S/N	Family	Species	Life-form	IZ				CZ	IUCN status
				F	R.F (%)	D (M^2)	R.D (%)		
1	Acanthaceae Juss.	*Asystasia gangetica* (L.) T. Anderson	Forb	11.11	1.45	0.04	1.47	√	NA
2	Amaranthaceae Juss.	*Alternanthera sessilis* L.	Forb	11.11	1.45	0.04	1.47	_	LC
		Cyathula prostrata (L.) Blume	Forb	_	_	_	_	√	NA
		Gomphrena celosioides Mart.	Forb	_	_	_	_	√	NA
3	Anacardiaceae (R.Br) Lindl.	*Mangifera indica* L.	Tree	_	_	_	_	√	DD
		Spondias mombin Jacq.	Tree	_	_	_	_	√	NA
4	Araceae Juss.	*Colocasia esculenta* (L.) Schott.	Forb	11.11	1.45	0.04	1.47	√	LC
5	Asteraceae Bercht. and J. Presl	*Ageratum conyzoides* (Mill.) M. Sharma	Forb	77.78	10.14	0.28	10.29	√	NA
		Chromolaena odorata (L.) R. M. King and H. Rob.	Shrub	55.56	7.25	0.2	7.35	√	NA
		Emilia coccinea (Sims) G. Don	Forb	22.22	2.90	0.08	2.94	√	NA
		Synedrella nodiflora (L.) Gaertn.	Forb	11.11	1.45	0.04	1.47	√	NA
		Tridax procumbens L.	Forb	11.11	1.45	0.04	1.47	_	NA
		Vernonia cinerea L.	Forb	_	_	_	_	√	NA
		Aspilia africana (Pers.) C.D. Adams	Forb	22.22	2.90	0.08	2.94	_	NA
6	Commelinaceae Mirb.	*Commelina diffusa* Burm.f.	Forb	11.11	1.45	0.04	1.47	_	LC
7	Convolvulaceae Juss.	*Ipomoea cairica* (L.) Sweet	Climber	11.11	1.45	0.04	1.47	√	NA
		Ipomoea hederifolia L.	Climber	_	_	_	_	√	NA
		Ipomoea involucrata Beauv.	Climber	33.33	4.35	0.12	4.41	_	NA
8	Costaceae Nakai	*Costus afer* Ker Gawl.	Forb	11.11	1.45	0.04	1.47	_	NA
9	Cucurbiataceae Juss.	*Luffa cylindrica* M. Roem.	Climber	_	_	_	_	√	NA
10	Cyperaceae Juss.	*Kyllinga erecta* Schumach.	Sedge	_	_	_	_	√	NA
		Mariscus alternifolius Vahl	Sedge	_	_	_	_	√	NA
11	Dioscoreaceae R.Br.	*Dioscorea alata* L.	Climber	11.11	1.45	0.04	1.47	_	NA
		Dioscorea sp.	Climber	11.11	1.45	0.04	1.47	_	NA
12	Euphorbiaceae Juss.	*Alchornea cordifolia* (Schumach. and Thonn.) Müll.Arg.	Shrub	22.22	2.90	0.08	2.94	_	NA
		Euphorbia heterophylla L.	Forb	_	_	_	_	√	NA
		Euphorbia hirta L.	Forb	_	_	_	_	√	NA
		Manihot esculenta Crantz	Shrub	22.22	2.90	0.08	2.94	√	NA
		Phyllanthus amarus Schumach. and Thonn.	Shrub	11.11	1.45	0.04	1.47	_	NA
		Phyllanthus muellerianus (Kuntze) Exell	Shrub	11.11	1.45	0.04	1.47	_	NA
13	Fabaceae Lindl.	*Peltophorum pterocarpum* (DC.) K. Heyne	Tree	_	_	_	_	√	NA
		Calopogonium mucunoides Desv.	Climber	55.56	7.25	0.2	7.35	√	NA
		Centrosema pubescens Benth.	Climber	11.11	1.45	0.04	1.47	_	NA
		Desmodium tortuosum (Sw.) DC.	Climber					√	NA
		Desmodium triflorum (L.) DC.	Climber	11.11	1.45	0.04	1.47	√	LC
14	Lamiaceae Martinov	*Solenostemon monostachyus* (P. Beauv.) Briq.	Forb	_	_	_	_	√	NA
15	Loganiaceae R.Br. ex Mart.	*Anthocleista vogelii* Planch.	Tree	22.22	2.90	0.08	2.94	_	NA
16	Malvaceae Juss.	*Abelmoschus esculentus* (L.) Moench	Shrub	_	_	_	_	√	NA

(Continued)

TABLE 2 | Continued

S/N	Family	Species	Life-form	IZ				CZ	IUCN status
				F	R.F (%)	D (M^2)	R.D (%)		
		Sida acuta Burm.fil.	Forb	_	_	_	_	√	NA
		Urena lobata L.	Shrub	22.22	2.90	0.08	2.94	√	NA
17	Melastomataceae Juss.	*Tristemma hirtum* P. Beauv.	Forb	11.11	1.45	0.04	1.47	_	NA
18	Onagraceae Juss.	*Ludwigia* sp.	Forb	_	_	_	_	√	LC
19	Passifloraceae Juss. ex Roussel	*Adenia cissampeloides* (Planch. ex Hook.) Harms	Climber	11.11	1.45	0.04	1.47	_	NA
20	Piperaceae Giseke	*Peperomia pellucida* (L.) Kunth	Forb	_	_	_	_	√	NA
21	Poaceae Barnhart	*Axonopus compressus* (Sw.) P. Beauv.	Grass	33.33	4.35	0.12	4.41	√	NA
		Brachiaria deflexa (Schumach.) C.E. Hubb. ex Robyns	Grass	11.11	1.45	0.04	1.47	√	NA
		Eleusine indica (L.) Gaertn.	Grass	_	_	_	_	√	LC
		Imperata cylindrica (L.) Raeusch.	Grass	11.11	1.45	0.04	1.47	_	NA
		Panicum maximum Jacq.	Grass	66.67	8.70	0.24	8.82	√	NA
		Pennisetum purpureum Schumach.	Grass	11.11	1.45	0.04	1.47	_	NA
		Setaria barbata (Lam.) Kunth	Grass	11.11	1.45	0.04	1.47	√	NA
22	Portulacaceae Juss.	*Talinum triangulare* (Jacq.) Willd.	Forb	_	_	_	_	√	NA
23	Pteridaceae E.D.M.Kirchn	*Adiantum* sp.	Fern	11.11	1.45	0.04	1.47	_	NA
24	Rubiaceae Juss.	*Borreria ocymoides* (Burm.f.) DC.	Forb	33.33	4.35	0.12	4.41	_	NA
		Borreria verticillata (L.) G. Mey.	Forb	11.11	1.45	0.04	1.47	_	NA
		Diodia scandens Sw.	Climber	11.11	1.45	0.04	1.47	_	NA
		Oldenlandia corymbosa L.	Forb	11.11	1.45	0.04	1.47	√	LC
25	Thelypteridaceae Ching ex Pic. Serm.	*Cyclosorus* sp.	Fern	11.11	1.45	0.04	1.47	_	NA
26	Ulmaceae Mirb.	*Trema orientalis* (L.) Bl.	Tree	_	_	_	_	√	NA
27	Urticaceae Juss.	*Laportea aestuans* (L.) Gaud.	Forb	_	_	_	_	√	NA

F, Frequency; R.F, Relative frequency; D, Density; R.D, Relative density; _, Absent, √, Present, NA, Not Assessed, DD, Data Deficient, LC, Least Concern.

TABLE 3 | Species index table for the sampled points with in the studied area.

Sampled points	Incident zone							Pooled control zone
Indices	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**
Dominance_D	0.184	0.088	0.09511	0.5017	0.3841	0.2779	0.28	0.3581
Simpson_1-D	0.816	0.912	0.9049	0.4983	0.6159	0.7221	0.72	0.6419
Shannon_H	1.885	2.66	2.521	0.6914	1.115	1.335	1.332	1.247
Margalef	2.203	4.346	3.463	0.353	1.059	0.9568	0.932	1.134
Evenness_eˆH/S	0.8237	0.794	0.8297	0.9983	0.7624	0.9499	0.9473	0.6959
Equitability_J	0.9067	0.9202	0.9311	0.9975	0.8044	0.9629	0.961	0.7747

Conservation Status of the Species in the Study Area

The conservation status of the following species based on the IUCN global assessment, *Alternanthera sessilis* (L.), *Spondias mombin* Jacq., *C. esculenta*, *D. triflorum*, *Eleusine indica* (L.) Gaertn., *O. corymbosa*, *C. diffusa*, *I. cairica*, *K. erecta*, *Alchornea cordifolia* (Schumach. and Thonn.) Müll.Arg., *Urena lobata* L., Geartn, *Imperata cylindrica* (L.) Raeusch., *Anthocleista vogelii* Planch., *Pennisetum purpureum* Schumach, *Trema orientalis* (L.) Bl., and the Pan-Africa assessment for *A. conyzoides* was LC. *Mangifera indica* L. and *M. esculenta* were DD, while others were NE for those not accessed or ND for those not identified to the species level (**Table 2**).

DISCUSSION

The floristic composition of the sampled area was examined based on the impact of coal mining, farming, and climate change (precipitation). For instance, the annual rainfall for Enugu in 2016 (NIMET, 2017) was more than the average quantity recorded in the past (Ofomata, 1965; Ezeigbo and Ezeanyim, 1993; NIMET, 2013; Okwu-Delunzu et al., 2018). The increase in annual rainfall might have contributed to species diversity within the Coal Camp. Thus, it was not surprising that plants in the inventory were characteristic of forests, uplands, flooded and wet places.

The observed change in vegetation from tropical rain forest to savannah grasslands due to anthropogenic activities had also been reported by Ezeigbo and Ezeanyim (1993). The registered cultivated species are due to their propagation by the human communities, and their management is still carried out for consumption and income. This practice could also have contributed to the disruption of the natural succession within the study area.

The species distribution on the coal dumpsite (IZ) was compared with that from an area presumed free of visible coal discards (CZ) but not pristine. Since we had no access to pristine vegetation, reference was made to a technical report detailing a field survey of Abor (Environmental and Social Impact Assessment [ESIA], 2016), about 15 km away, which we adopted as baseline data for use as pre-mining information on the vegetation. We found similarities in the species composition between the reference survey and our study area (*A. vogelii*, *A. cordifolia*, *S. mombin*, *A. africana*, *C. odorata*, *A. gangetica*, and *E. indica*). However, our study failed to record some tree species (*Parkia biglobosa*, *Khaya senegalensis*, *Baphia nitida*, and *Daniellia oliveri*). Reasons for this is due to fewer anthropogenic activities and semi-tropical rainforest vegetation type (Environmental and Social Impact Assessment [ESIA], 2016).

The unequal sampling frequency between the IZ and CZ due to patchiness impeded the comparison of species richness and diversity for both zones. Instead, presence and absence were used (**Table 2**). From our study, both areas shared moderately similar species composition (family 37.04% and species 26.67%), contrasting previous studies where species diversity was higher in the pre-mining/unpolluted sites than in the polluted/reclamation sites (Komara et al., 2016; Hapsari et al., 2020). On the other hand, life forms such as forbs, grasses and trees in our CZ were higher than the polluted study area. The observed distribution patterns between both zones are driven by similar edaphoclimatic and physiographic features; the observed heterogeneity in species distribution within the IZ could have been due to the post-impact of coal mining activities and human activities. This may underlie the highest species diversity observed in this location.

The dominance of the families Asteraceae and Poaceae (**Table 2** and **Figure 3A**) in the revegetation of the coal dumpsite conform with other studies. For instance, they have been reported as the principal colonizers in a Kosovo coal ash dump site (Mustafa et al., 2012) and a Moravia dumpsite in Colombia (Sánchez-Pinzón et al., 2010). Ekka and Behera (2011) also noted that Poaceae was the prominent family in revegetation of 3–9-year-old dump sites post-coal mining. On the other hand, *A. conyzoides, P. maximum*, *C. mucunoides and C. odorata* were the most commonly found species in the study area. In line with previous studies, these species have been reported in various coal mining sites (Sánchez-Pinzón et al., 2010; Komara et al., 2016; Novianti et al., 2017; Yusuf and Arisoesilaningsih, 2017; Igwenagu et al., 2019; Hapsari et al., 2020). The frequent occurrence of these species in polluted sites might be due to their invasive nature (Borokini, 2011). Additionally, these species seem to have a highly competitive rate, ease of seed germination, and high resistance to varying environmental and soil conditions.

Our evaluation of species composition within the study area exposed the impact of previous mining activities and their waste on vegetation structure. Data from the IUCN Assessment revealed that most of the species being NE and DD were not in the conservation status spectrum, and the remnant quarter belonged to the Lower risk category of LC. there were no Threatened species within the sampled area., Therefore, there were no endangered, vulnerable or threatened species within the study area except for the trees. Even so, an assessment scheme needs to be established to monitor the remnant biodiversity within the study area for posterity.

The Economic Tree Species Present in the CZ Were *M. indica*, and *A. occidentale*. Food Crops encountered during the study include *C. esculenta* (IZ and CZ), *Dioscorea* spp. (IZ), *M. esculenta* (IZ and CZ). In agreement with previous studies, one or multiple of these species (*C. esculenta* and *M. esculenta*) have been found in some coal mine sites (Panda et al., 2011; Komara et al., 2016; Arshi, 2017; Igwenagu et al., 2019). Previously, some economic tree species that were logged from this ecological zone include *Canarium schweinfurtii* (black pear), *Pentaclethra macrophylla* (oil bean), *Garcinia kola* (bitter kola), *Vitex doniana* (black plum). *Milicia excelsa* (Iroko) *Khaya ivorensis* (mahogany) *Nauclea diderrichii* (Opepe) (Ogbonna et al., 2015). These species have disappeared as none was present in this current study. Anthropogenic activities such as overexploitation and habitat fragmentation and might have influenced the forest structure reducing the trees to mostly forbs. Those species would have been exploited because of their peculiar wood qualities (*M. excelsa*, *K. ivorensis*) and medicinal uses (*P. macrophylla*).

Revegetation for conservation must be mandatory for the maintenance of the ecosystem and future use of the area. There is a need to remove the invasive species as they could hinder profitable species (Yusuf and Arisoesilaningsih, 2017).

CONCLUSION

Our study evaluated the biodiversity assessment of a defunct coal mine site (IZ) and found marginal differences between its species composition and those within the CZ. This may be due to the fact that mining had ceased entirely at this location about 15 years ago. Post-mining activities (e.g., farming and community development projects) and the plasticity of the species surviving harsh climatic and environmental conditions, could explain the species population dynamics within the study area. Nevertheless, we propose future studies to investigate the rhizospheric microbial communities and their metabolic roles in coal degradation. This study could provide insights into the sustenance of plant growth in coal polluted environments. Our findings iterated the continuous decline of biodiversity in the Nigerian ecosystem following the irrational use of the resources. Although the study site might not be considered a high conservation priority area, a habitat restoration scheme for maintaining the ecosystem is recommended. Lastly, from the findings of this study, it would be helpful to form a guideline for future conservation management of the study area.

DATA AVAILABILITY STATEMENT

The original contributions presented in the study are included in the article/supplementary material, further inquiries can be directed to the corresponding author/s.

AUTHOR CONTRIBUTIONS

AO participated in the collection of experimental data for floral distribution, data analysis, and interpretation. EI participated in the conception and design of floral distribution experiments. OO participated in the conception and design of floral distribution, collection of experimental data, data analysis, and interpretation. IN initiated the research idea, participated in the conception, design, data analysis, and interpretation. All authors participated in writing and approved the final manuscript.

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REFERENCES

Adeniyi, T., Oyebanji, O., and Adeonipekun, P. (2016). Floral diversity in the wetlands of Ibeju-Lekki area. Lagos, Nigeria. *Ife J. Sci.* 18, 729–737.

Adeonipekun, P., Oyebanji, O., Adeniyi, T., and Oduoye, O. (2018). An ecological assessment of forest trees in Lamba area of Gashaka Gumti National Park, Taraba state, Implications for biodiversity conservation and monitoring. *Niger. J. Bot.* 31, 167–180.

Afonja, A. (1979). "Further Studies on the Lafia Coal Deposit, PHASE II," in *Research Report No. IFE/CHE/CP/11*. Stratford-upon-Avon: Coal Research Laboratory, Ife

Agbalagba, E., and Uzo, A. (2018). Investigating the Radiological Burden of Working and Residing Around Coal Mine Site in Nigeria. *Int. J. Innov. Res. Sci. Eng. Technol.* 7:8000–8011.

Akobundu, I. O., and Agyakwa, C. W. (1998). *A Handbook of West African Weeds*, 2nd Edn. Ibadan: IITA, 521.

Alston, A. H. (1959). *Fern and Fern-Allies of West Tropical Africa*. London: Crown Agents for Overseas Government and Administrations.

Arshi, A. (2017). Reclamation of coalmine overburden dump through environmental friendly method. *Saudi J. Biol. Sci.* 24, 371–378. doi: 10.1016/j.sjbs.2015.09.009

Baig, M. N. (1992). Natural Revegetation of Coal Mine Spoils in the Rocky Mountains of Alberta and Its Significance for Species Selection in Land Restoration. *Mt. Res. Dev.* 12, 285–300. doi: 10.2307/3673671

Barker, P. (2001). *A Technical Manual for Vegetation Monitoring. Resource Management and Conservation, Department of Primary Industries*. Hobart: Water and Environment.

Behre, D. (2006). "*Feasibility of Nigerian Coal Resources Development CMM Global Overview*," Report by the U.S. Environmental Protection Agency Coalbed Methane Outreach Program, Washington, DC.

Borokini, T. (2011). Invasive alien plant species in Nigeria and their effects on biodiversity conservation. *Trop. Conserv. Sci.* 4, 103–110. doi: 10.1177/194008291100400110

Briggs, R. S., Lucas, R. G., and Taylor, Z. T. (2003). Climate Classification for Building Energy Codes and Standards, Part 2—Zone Definitions, Maps, and Comparisons. *ASHRAE Trans.* 109, 122–130.

Buta, M., Blaga, G., Laura, P., Păcurar, I., Rosca, S., Borsai, O., et al. (2019). Soil Reclamation of Abandoned Mine Lands by Revegetation in Northwestern Part of Transylvania: a 40-Year Retrospective Study. *Sustainability* 11:3393. doi: 10.3390/su11123393

Chukwu, M., Folayan, C. O., Pam, G. Y., and Obada, D. O. (2016). Characterization of Some Nigerian Coals for Power Generation. *J. Combust.* 2016, 01–11. doi: 10.1155/2016/9728278

Chukwuike, B. (2018). *Analysis of Rainfall and Temperature variations and its implication on climate change in Enugu state, Nigeria*. Thesis. University of Technology, Owerri. Available online at: https://africantheses.org/abstracts/6432-197560

Dazy, M., Béraud, E., Cotelle, S., Grévilliot, F., Férard, J., and Masfaraud, J. (2009). Changes in plant communities along soil pollution gradients, Responses of leaf antioxidant enzyme activities and phytochelatin contents. *Chemosphere* 77, 376–383. doi: 10.1016/j.chemosphere.2009.07.021

Ekka, N., and Behera, N. (2011). Species composition and diversity of vegetation developing on an age series of coal mine spoil in an open cast coal field in Orissa, India. *Trop. Ecol.* 52, 337–343.

Environmental and Social Impact Assessment [ESIA]. (2016). *Environmental & Social Impact Assessment (Esia). for the Proposed 132/33kV Transmission Substation at Abor (9th Mile), Enugu State. Draft Final Report*. Nigeria: Transmission Company of Nigeria - Project Management Unit (TCN-PMU). 1–255.

Environmental Systems Research Institute [ESRI]. (2014). *ArcGIS Desktop Help 10.2 Geostatistical Analyst*. URL: http://resources.arcgis.com/en/help/main/10.2/index.html

Ezeigbo, H. I., and Ezeanyim, B. N. (1993). Environmental Pollution from Coal Mining Activities in the Enugu Area Anamka State Nigeria. *Mine. Water Environ.* 112, 53–62. doi: 10.1007/bf02914799

Ezekwe, C., and Odukwe, A. (1980). Coal in Nigeria. *Energy* 5, 177–180. doi: 10.1016/0360-5442(80)90006-7

Ghose, M. (2004). Effect of opencast mining on soil fertility. *J. Sci. Ind. Res. India* 63, 1006–1009.

Hammer, O., Harper, D., and Paul, D. (2001). Past, Paleontological Statistics Software Package for Education and Data Analysis. *Palaeontol. Electron.* 4:1–9.

Hapsari, L., Trimanto, T., and Budiharta, S. (2020). Spontaneous plant recolonization on reclaimed post-coal mining sites in East Kalimantan, Indonesia, Native versus alien & succession progress. *Biodiversitas* 21, 2003–2018.

Hazarika, P., Talukdar, N., and Singh, Y. (2006). Natural colonization of plant species on coal mine spoils at Tikak Colliery, Assam. *Trop. Ecol.* 47, 37–46.

Huang, Y., Tian, F., Wang, Y., Wang, M., and Hu, Z. (2014). Effect of coal mining on vegetation disturbance and associated carbon loss. *Environ. Earth Sci.* 73, 2329–2342. doi: 10.1007/s12665-014-3584-z

Hutchinson, J., and Dalziel, J. (1958). *Flora of West Tropical Africa. Volume 1*, 2nd Edn. Milbank, London: Crown Agents for Overseas Government and Administrations, 828.

Hutchinson, J., and Dalziel, J. (1963). *Flora of West Tropical Africa. Volume II*, 2nd Edn. Milbank, London: Crown Agents for Overseas Government and Administrations, 544.

Hutchinson, J., and Dalziel, J. (1968). *Flora of West Tropical Africa. Vol III. Part 1*. Milbank, London: Crown Agents for Overseas Government and Administrations.

Hutchinson, J., Dalziel, J., and Hepper, F. N. (1972). *Flora of West Tropical Africa. Vol 3*, 2nd Edn. Milbank, London: Crown Agents for Overseas Government and Administrations, 574.

Hutchinson, J., and Dalziel, M. D. (1954). *Flora of West Tropical Africa. Vol. I. Part 1*, 2nd Edn. London: Crown Agents, 661.

Igwenagu, E., Okoye, P., Ogbuagu, O., and Ubaoji, I. (2019). Heavy Metal Levels in Plants Growing on Abandoned Coal Ash Dumpsite in Oji, Enugu State, Nigeria. *Am. J. Environ. Sustain. Dev.* 4, 43–49.

International Energy Agency [IEA]. (2018). *Coal Information (2018)*. Paris: OECD Publishing, doi: 10.1787/coal-2018-en

International Plant Names Index [IPNI]. (2020). *Published on the Internet http,// www.ipni.org, The Royal Botanic Gardens, Kew, Harvard University Herbaria & Libraries & Australian National Botanic Gardens*. Kew: The Royal Botanic Gardens, Kew. [Retrieved June 05 2020].

IUCN. (2021). *The IUCN Red List of Threatened Species. Version 2021-1*. URL: https://www.iucnredlist.org. Downloaded on [April 16 2021]

The Angiosperm Phylogeny Group [APG]. (2016). An update of the Angiosperm Phylogeny Group classification for the orders and families of flowering plants. APG III. *Bot. J. Linn. Soc.* 181, 1–20. doi: 10.1111/boj.12385

Keay, R., Wadwa, J., Onochie, C. F. A., and Stanfield, D. P. (1964). *Nigerian Trees. Vol. II*. Ibadan, Nigeria: Department of Forest Research, 384.

Komara, L. L., Choesin, D. N., and Syamsudin, T. S. (2016). Plant diversity after sixty years post coal mining in East Kalimantan, Indonesia. *Biodiversitas* 17, 531–538.

Kundu, N. K., and Ghose, M. K. (1997). Soil profile Characteristic in Rajmahal Coalfield area. *Ind. J. Soil Water Conserv.* 25, 28–32.

Lowe, J., and Stanfield, D. (1974). *The Flora of Nigeria Sedge (Family Cyperaceae)*. Ibadan: Ibadan University Press, 144.

Magurran, A. E. (1988). *Ecological Diversity And Its Measurement*. London: Chapman and Hall, 179.

Mueller-Dombois, D. (1972). Crown distortion and elephant distribution in the woody vegetation of Ruhuna National Park, Ceylon. *Ecology* 53, 208–226. doi: 10.2307/1934074

Mustafa, B., Hajdari, A., Krasniqi, F., Morina, I., Riesbeck, F., and Sokoli, A. (2012). Vegetation of the Ash Dump of the "Kosova A" Power Plant and the Slag Dump of the "Ferronikeli" Smelter in Kosovo. *Res. J. Environ. Earth Sci.* 4, 823–834.

National Research Council [NRC]. (1995). *Coal, Energy for the Future*. Washington, DC: The National Academies Press, 304. doi: 10.17226/4918

Novianti, V., Choesin, D. N., Iskandar, D. T., and Suprayogo, D. (2017). Plant species from coal mine overburden dumping site in Satui, South Kalimantan, Indonesia. *J. Degrade. Min. Land. Manag.* 4, 927–936. doi: 10.15243/jdmlm.2017.044.927

Nsa, I., Igbinigie, E., Adeogun, O., Adekunle, A., Ilori, M., Adongbede, E., et al. (2017). The potential of mycorrhizal associations on a coal dump site. *JIIRT* 6, 165–177.

Obi, F. (2009). *The Geology of Enugwu – Ngwo and Enviros*. Academia: Manuscript submitted to Anambra State University.

Ofomata, G. E. K. (1965). Factors of soil erosion in Enugu area of Nigeria. *J. Geo. A Niger.* 10, 3–9.

Ogbonna, P., Nzegbule, E., and Okorie, P. (2015). Environmental Impact Assessment of Coal mining at Enugu, Nigeria. *Impact Assess. Proj. Apprais.* 33, 73–70. doi: 10.1080/14615517.2014.941711

Ogunsola, O. (1991). Coal Production and Utilization Trends in Nigeria. *Fuel Sci. Technol. Int.* 9, 1211–1222. doi: 10.1080/08843759108942323

Okwu-Delunzu, V., Iwueke, N., and Aniagolu, C. (2018). Gully erosion and its environmental impact in Eke, Udi local government area of Enugu state. *Environ. Ecol. Res.* 6, 507–517. doi: 10.13189/eer.2018.060511

Oyebanji, O. O., Bolarinwa, K. A., and Olowokudejo, J. D. (2020). Diversity and distribution patterns of aquatic macrophytes in selected ecosystems of southwestern Nigeria. *Afr. J. Ecol.* 58, 468–480. doi: 10.1111/aje.12717

Panda, S., Minz, A., and Dhal, N. (2011). Floristic Studies of Lajkura Coal Mines Area Jharsuguda, Odisha, An Overview. *Int. J. Sci. Res. Publications* 4, 1–11.

Patel, R. M., Ashishkumar, and Kaneria, M. J. (2020). "Floristic Diversity, Distribution and Conservation Status in the Vicinity of Coal Mines of Kachchh District in Gujarat, India," in *Environmental Concerns and Sustainable Development*, eds V. Shukla and N. Kumar (Singapore: Springer). doi: 10.26438/ijsrbs/v5i5.15

Patel, R. M., Kumar, V. V., and Ashishkumar (2012). Structure and composition of woody vegetation in western Kachchh, Gujarat. *J. Biosci. Res.* 3, 163–174.

Pauletto, E., Stumpf, L., Pinto, L., Silva, T., Ambus, J., Garcia, G., et al. (2016). Reclamation of a Degraded Coal-Mining Area with Perennial Cover Crops. *Rev. Bras. Ciênc. Solo* 40:e0150482. doi: 10.1590/18069657rbcs20150482

Radford, A., Dickison, W., Massey, J., and Bell, C. (1974). *Vascular Plant Systematics*. New York: Harper and Row, 891.

Ritchie, H., and Roser, M. (2019). *"Fossil Fuels". Published online at OurWorldInData.org*. URL: https://ourworldindata.org/fossil-fuels' [Online Resource].

Sánchez-Pinzón, M. S., Cogollo-Pacheco, A., Arroyave-Quiceno, C., Torrenegra-Guerrero, R. D., Giraldo-Mejía, A. M., Barahona-Rosales, R. (2010). Botanical Diversity and Heavy Metal Content in the Residue Matrix and Plants at the Moravia Dump in Medellín, *Colombia*. *Rev. Fac. Nac. Agron. Medellín* 63, 5209–5224.

Sanchita, A. S. (2018). "Gene expression analysis of medicinal plants under abiotic stress conditions," in *: Plant Metabolites Under Regulation and Environmental Stress*, eds P. Ahmad, M. Ahanger, V. Singh, D. Tripathi, P. Alam, and M. Alyemeni (London, United Kingdom: Elsevier Ltd. Academic Press), doi: 10.1016/B978-0-12-812689-9.00023-6

Sarma, K. (2005). "Impact of coal mining on vegetation, a case study in Jaintia Hills district in Jaintia Hills, District of Meghalaya, India," in. *Submitted to the International Institute for Geo-Information Science and Earth Observation*. Ph.D. thesis, (The Netherlands: Enschede), 1–76. .

Shannon, C. E., and Weaver, W. (1963). *The Mathematical Theory of Communication*. Illinois: University of Illinois Press.

Sheoran, V., Sheoran, A. S., and Poonia, P. (2010). Soil Reclamation of abandoned mine land by revegetation: a review. *Int. J. Soil Sediment Water* 3:13.

Sikakwe, G. (2017). A comparative study of environmental impact of mine drainage from Enugu coal mines, Southeastern Nigeria. *Environ. Sci. Pollut. Res.* 3, 203–207.

Sikakwe, G. U., Ephraim, B. E., Nganje, T. N., Ntekim, E. E. U., and Amah, E. A. (2015). Geoenvironment impact of Okpara coal mine Enugu Southeastern Nigeria. *Adv. Appl. Sci. Res.* 6, 5–16.

Simpson, E. H. (1949). Measurement of diversity. *Nature* 163:688. doi: 10.1038/163688a0

Singh, A. (2011). Vascular flora on coal mine spoils of Singrauli coalfields, India. *J. Ecol. Nat. Environ.* 3, 309–318. doi: 10.5897/JENE.9000113

Singh, A., Raghubanshi, A., and Singh, J. (2002). Plantations as a tool for mine spoil restoration. *Curr. Sci.* 82, 1436–1441.

Skousen, J. G., Johnson, C. D., and Garbutt, K. (1994). Natural Revegetation of 15 Abandoned Mine Land Sites in West Virginia. *J. Environ. Qual.* 23:1224. doi: 10.2134/jeq1994.00472425002300060015x

Unanaonwi, O. E., and Amonum, J. P. (2017). Effect of Mining Activities on Vegetation Composition and nutrient status of Forest Soil in Benue Cement Company, Benue State, Nigeria. *Int. J. Environ. Agric. Biotechnol.* 2, 297–305. doi: 10.22161/ijeab/2.1.39

Yang, Q., Zhao, B., and Guo, D. (2015). A review on vegetation restoration of opencast coal mine areas in northern China. *Chinese J. Ecol.* 34, 1152–1157.

Yusuf, M., and Arisoesilaningsih, E. (2017). Exotic Plant Species Attack Revegetation Plants in Post-Coal Mining Areas. *AIP Conf. Proc.* 1908, 040002. doi: 10.1063/1.5012716

Zou, C., Zhao, Q., Zhang, G., and Xiong, B. (2016). Energy revolution: from a fossil energy era to a new energy era. *Nat. Gas Ind.* 3, 1–11. doi: 10.1016/j.ngib.2016.02.001

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Benefits Beyond Borders: Assessing Landowner Willingness-to-Accept Incentives for Conservation Outside Protected Areas

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Unplanned land-use change surrounding protected areas (PAs) can lead to degradation and fragmentation of wildlife habitats, thereby placing tremendous pressure on PAs especially in tropical countries. Incentivizing the expansion of habitats beyond PAs will not only benefit wildlife but also has the potential to create livelihood opportunities for marginalized communities living adjacent to PAs. Our study explored landowners' willingness to participate in an incentive-based, wildlife-friendly land-use program using a discrete choice modeling approach. We surveyed 699 landowners living in 287 villages within a five-kilometer buffer around Nagarahole and Bandipur National Parks in India. We found that landowners preferred wildlife-friendly land-use over their ongoing farming practices. Landowners preferred short-term programs, requiring enrolling smaller parcels of land for wildlife-friendly land-use, and offering higher payment amounts. Landowners with larger landholdings, a longer history of living next to the PA, and growing fewer commercial crops were more likely to prefer enrolling large parcels of land. Landowners who grew more commercial crops were likely to prefer long term programs. We also estimated the average monetary incentive to be INR 64,000 (US$ 914) per acre per year. Wildlife-friendly land use, in developing economies like India with shrinking wildlife habitats and expanding infrastructural developments, could supplement rural incomes and potentially expand habitat for wildlife, thereby being a promising conservation strategy.

Keywords: agriculture, choice experiment, incentive, land sharing, land-use, wildlife

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INTRODUCTION

Protected Areas (PAs) are designated with the aim of conserving the world's biodiversity. In 2010, the Parties to the United Nations (UN) Convention on Biological Diversity (CBD) set five strategic goals and 20 Aichi targets to reduce the rate of biodiversity loss by 2020 (CBD, 2010). Aichi Target 11 calls explicitly for PA expansion to increase ecological representativeness and improve connectivity through well-connected PA networks and other effective area-based conservation measures (CBD, 2010). However, PA expansion has not been met globally and biodiversity loss continues (Stokstad, 2020). New targets for 2030 are being set and one of the proposals suggests bringing 30% of total land and marine habitat under protection (CBD, 2020).

Protected areas have been one of the most effective conservation strategies and represent the last remaining strongholds for certain imperiled species (Pacifici et al., 2020). By themselves, PAs are inadequate to conserve biodiversity and arrest its decline in the long-term (Geldmann et al., 2019).This is because the existing global PA network is under immense human pressure (Jones et al., 2018) and less than 10% are structurally connected (Saura et al., 2018; Ward et al., 2020). Poor connectivity driven by habitat loss and fragmentation can negatively impact ecological processes and dispersal of wide-ranging species (Crooks et al., 2011; Jayadevan et al., 2020; Nayak et al., 2020). Further, political pressure, insufficient funds, and land tenure prevent absolute protection of land adjoining PAs where wildlife co-occurs with people (Watson et al., 2014).

Protected Areas and their surrounding human-modified regions are linked components of an interacting ecological system, rather than mutually independent entities (Anand et al., 2010). Accordingly, the focus of conservation has evolved, recognizing the dynamic relationships between people and nature (Kareiva and Marvier, 2012). Planning conservation efforts that take both people and nature into consideration is challenging but essential. Identifying regions for habitat expansion centered around where potential beneficiaries are located could also ensure that sites surrounding PAs are more accessible for on-ground conservation effort and subsequent monitoring (Naidoo et al., 2019). A more inclusive and pragmatic approach can integrate the role of private lands adjoining statutory PAs in biodiversity conservation (Kamal et al., 2015; Drescher and Brenner, 2018). Private lands have the potential to contribute to conservation by supplementing additional habitat for wildlife, restoring structural connectivity, reinforcing corridors and buffer zones, and providing economic benefits through financial and market-based instruments such as payments for ecosystem services and ecotourism (Pegas and Castley, 2014; Maciejewski et al., 2016; Clements et al., 2018; Kremen and Merenlender, 2018; Capano et al., 2019). Expanding conservation initiatives beyond PAs requires understanding private landowners' interest in and willingness to undertake conservation through different land management strategies (Selinske et al., 2015; Gooden, 2019).

Private land conservation models such as agroforest ecosystems often harbor biodiversity of significant conservation value (Bhagwat et al., 2008; Anand et al., 2010; Ferreira et al., 2020). Multifunctional systems like these provide a wide range of economic, sociocultural, and environmental benefits including enhancing livelihood and food security, reducing biodiversity decline, and mitigating climate change (Kremen and Miles, 2012; Kremen and Merenlender, 2018). Agroforestry is particularly beneficial for smallholders as it produces diverse products and services on a small parcel of land (Hughes et al., 2020). Small farms also tend to have a high capacity for sustaining biodiversity and rural livelihoods (Kumaraswamy and Kunte, 2013; Ricciardi et al., 2021). Policies favoring smallholders, given their reliance on traditional farming techniques and in depth knowledge of the land, could benefit biodiversity conservation. This approach could further conserve local agroecosystems and rural livelihoods by increasing farmer resilience to external threats such as fluctuating crop prices or environmental variability (Harvey et al., 2008). While private land conservation is not new in African and Latin American countries, it is not yet a widespread practice in South Asian countries such as India (Karanth and Karanth, 2012; Drescher and Brenner, 2018; Capano et al., 2019).

India being a megadiverse country with a growing human population and <5% total land cover under protection, there is considerable concern over how the new biodiversity targets being set in the post-2020 framework could be met (ENVIS, 2021). Achieving 30% protected land area is contested in a country like India, and could come at the cost of people's livelihood requirements. Unlike other nations such as the United States and China, where large tracts of land with sparse human population are accorded PA status, India's PAs are relatively small, poorly connected, and surrounded by dense settlements (Pimm et al., 2018; Ghosh-Harihar et al., 2019). Despite benefits to the local communities, such as tourism, access to forest resources and grazing lands, and employment with park management, the socioeconomic, psychological, and human costs of living alongside wildlife are often high (Ogra, 2008; Baskaran et al., 2013; Karanth and Kudalkar, 2017). These often result in declining tolerance and increasing resentment among the local communities toward the forest, wildlife, and PA management in the form of retaliatory killing, arson, and even tussles with the PA management (Talukdar and Gupta, 2017; Kalam et al., 2018).

There is potential to foster stewardship and build tolerance among local landowners living beside PAs toward wildlife. This can be accomplished by building partnerships with local landowners to promote and support conservation efforts on private lands by directly transferring conservation benefits to local communities through innovative land-use policy and incentive schemes (Anand et al., 2010; Karanth and Karanth, 2012; Ghosh-Harihar et al., 2019). To this end, the objectives of our study were to examine the willingness of private landowners to (1) enroll in voluntary incentive-based wildlife-friendly land-use programs, and (2) undertake tourism activities on their land in the future. We also examined socioeconomic, demographic, and geographic factors that influenced landowner's choice to practice wildlife-friendly land-use.

We hypothesized that a landowner would be likely to choose a wildlife-friendly land-use program if the risk associated with the venture was low. We used a small proportion of land to be enrolled for the program, short enrollment period, and high payment amount as descriptors of low risk (Moon and Cocklin, 2011). We predicted younger landowners with higher economic security would be willing to take greater risk and invest in a novel wildlife-friendly land-use program. Wealthier farmers are likely to see the wildlife-friendly land-use income as less motivating. In contrast, landowners who lost crops to wildlife earlier may resent wildlife and hence, be unwilling to enroll larger parcels of land and for longer periods of time. With inadequate understanding of private land conservation in Asia (Capano et al., 2019), our study from India represents one of the few that examines landowners' interest in undertaking wildlife-friendly land-use.

MATERIALS AND METHODS

Study Area

Bandipur and Nagarahole National Parks are located in a landscape comprising tropical (dry and moist) deciduous forest and savannah in the global biodiversity hotspot, Western Ghats (**Figure 1**; Devidas and Puyravaud, 1995). This landscape represent PAs with very hard edges and almost zero forest cover outside the core boundary. The landscape supports some of the largest global populations of wide-ranging species such as tigers (*Panthera tigris*), elephants (*Elephas maximus*) along with populations of threatened taxa such as leopards (*Panthera pardus*), and Asiatic wild dogs (*Cuon alpinus*) (Jathanna et al., 2015; Karanth et al., 2020). This protected landscape has been facing tremendous pressures from linear infrastructure such as roads and power line projects (Jayadevan et al., 2020; Nayak et al., 2020). Increasing tourism pressure is another threat faced by these PAs, with more than 100,000 tourists visiting them annually (Karanth et al., 2017). The area adjacent to the parks comprises three districts with population densities ranging between 135 and 443 people/km^2 (Census, 2011). The buffer area consists of communities whose primary source of income is from rainfed/unirrigated agriculture, animal husbandry, and daily-wage labor (Karanth et al., 2013; Margulies and Karanth, 2018; Karanth and Vanamamalai, 2020). Majority of the annual household income is less than US $1600 (Karanth and Vanamamalai, 2020). Local communities often face wildlife-related losses in the form of crop-raiding, livestock depredation, property damage, and occasional injuries to and loss of human lives. Between 2014 and 2019, compensation was claimed for nearly 20,000 cases of wildlife-related losses by people living adjacent to these PAs (ibid).

Data Collection

We conducted interviews with farmers living within a five-kilometer buffer area around the two PAs from April to July 2019 (**Figure 1**). We obtained ethical approval for the study from the Human Subjects Review Board of the Centre for Wildlife Studies. All respondents were older than 18 years and consent was sought orally. A team of 16 trained volunteers conducted the surveys in the local language Kannada. We chose a previously established 13 sq.km grid-based sampling approach by Karanth et al. (2013) in the study area. We sampled 287 villages from 132 grids out of a potential 154 grids. Twenty-two grids were not sampled due to lack of consent, accessibility, or presence of water bodies or forest. We selected respondents by randomly selecting lands in each village. We ensured that surveyed lands were not directly adjacent

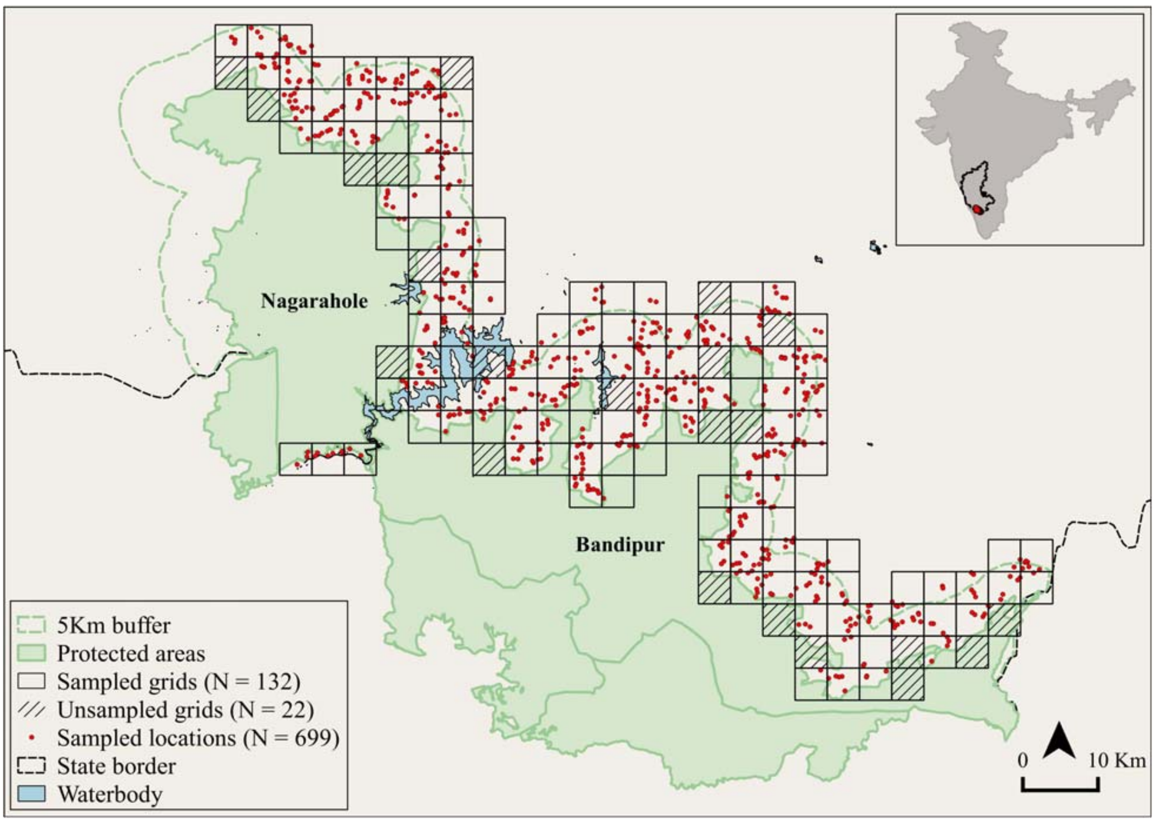


FIGURE 1 | Map of the study location with India shown in inset.

to one another. Surveys were carried out at the land parcel that the owners were willing to enroll in the program. We used the Open Data Kit (v1.25.2) to minimize errors due to paper-based data collection methods (Hartung et al., 2010).

Choice Experimental Design

We used a discrete choice experiment to examine the drivers of landowner's willingness to undertake wildlife-friendly land-use. Choice experiment is a stated preference method in economics used to elicit an individual's preferences for alternative hypothetical states (Hensher et al., 2005). Multiple attributes with varying levels describe alternatives presented to an individual. An attribute is a feature that influences an individual to choose an alternative state and a level is a measure that varies at fixed intervals (ibid). Responses are used to estimate the influence of attributes on preferences and establish their relative importance (ibid). We followed the study design and survey protocols as presented in Puri et al. (2021). We considered three attributes namely, percentage of land to be allocated, number of years of enrollment, and payment amount per acre per year. The details of the attributes are presented in **Table 1**.

The "Land" attribute was defined as the proportion of total land area to be set aside for wildlife-friendly land-use. Given the considerable heterogeneity in landholding across landowners in India, we consider the proportion of total land instead of specifying the land units in acres. Since the regeneration of degraded or highly modified agricultural lands is a long term process, our second attribute, "Year," represented the duration of the program enrollment. The monetary attribute considered here is the "Payment" that the landowners would receive per year per acre of land allowed for wildlife-friendly land-use. With the annual income of most households in the landscape being less than \$1600, all payment levels were set below this value (Karanth and Vanamamalai, 2020). We focused on only three attributes to reduce the cognitive burden on landowners while making decisions and reduce attribute non-attendance (Carlsson et al., 2010).

In the choice experiment, we presented landowners with a questionnaire containing descriptions of wildlife-friendly land-use contracts. Based on results from our pilot study (conducted in November 2018), we developed a D-efficient design with fixed priors (d-error of 0.12), with a total of nine choice cards and we blocked them into three choice sets (Louviere et al., 2008; Bliemer and Collins, 2016). Each landowner was presented with three choice cards. In each choice set, they were asked to choose between two types of wildlife-friendly land-use contracts and a status quo option (to not enroll in any program). An example of a choice card presented to landowners is provided in **Figure 2**.

The choice experiment survey was explained with the help of visual tools such as photographs. In addition to the choice experiment survey, we recorded landowner's background information including: (1) demographic factors (age and education, family size, and number of years living next to the PA); (2) socioeconomic factors (agricultural land size, type of livestock owned, different sources of household income, and number and type of crops grown); and (3) experience with wildlife-related losses (crop and asset loss, livestock depredation, and human casualties). We also derived geographic factors such as the distance of the landholding to the PA boundary using "geosphere" in RStudio 3.5.1 (Hijmans et al., 2016). Finally, to examine the potential of tourism as an additional source of income at the end of program enrollment, we asked the farmers about (1) their interest in various activities related to tourism on their land and (2) the type of support (government, skill development, and financial) they required to initiate tourism activities.

TABLE 1 | Attributes and corresponding levels used in the choice experiment.

Attribute	Description	Level
Land	Amount of land to be enrolled	25% 50% 75%
Year	Contract period	4 years 8 years 12 years
Payment	Amount paid per acre per year	INR 45,000 (US\$ 643) INR 60,000 (US\$ 857) INR 75,000 (US\$ 1071) INR 90,000 (US\$ 1285)

INR, Indian rupees (US\$ 1 = INR 70 at the time of the survey).

Choice Model

We assessed the aggregate preference of the landowners toward the policy options using Multinomial Logit model (MNL) analysis implemented in LIMDEP NLOGIT 5.0 (Green, 2012). An alternative specific constant (ASC) was included in the models to check if there was an inherent preference to forgo the status quo.

Assuming linear indirect utility function, utility derived by landowner *i* from alternative wildlife-friendly land-use *j* in choice set *k* is,

$$V_{ijk} = \alpha + \beta' x_{jk} + \lambda \left(y_i - payment_{jk} \right) + \varepsilon_{ijk} \qquad (1)$$

where, α is the ASC which captures the landowner's preference to be in the status quo. The vector x_{jk} is the vector of attributes of jth alternative of kth choice set and β' are the corresponding parameter estimates. y_i is the investment cost of landowner *i*, and ε_{ijk} the random error term. The probability of landowner *i* choosing alternative *j* over alternatives *h* is expressed as,

$$P_{ijk} = P \left\{ \beta' x_{jk} + \lambda \left(y_i - payment_{jk} \right) + \varepsilon_{ijk} > \beta' x_{hk} + \lambda \left(y_i - payment_{hk} \right) + \varepsilon_{ihk};\ \forall j \neq h \right\} \qquad (2)$$

However, MNL assumes homogeneous preference across the sampled population, which is less likely. Therefore, we tested for heterogeneity in terms of landowners' choices toward the attributes of wildlife-friendly land-use programs using Random Parameter Logit models (RPL). We interacted the "Land" and "Year" attributes with landowners' socioeconomic and geographic characteristics to test the relative influence of these variables on people's choices. We assumed the "Payment" attribute as fixed, and the "Land" and "Year" attributes to be normally distributed (Carlsson et al., 2003). The model estimation was done with 1000 Halton draws, and the top model

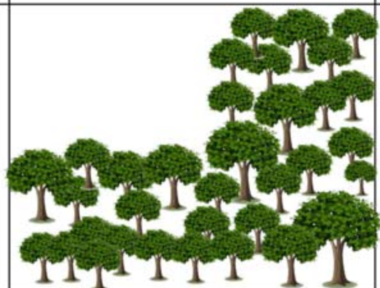
ATTRIBUTE	PROGRAM A	PROGRAM B	PROGRAM C
LAND			X
YEAR	4 YEARS	12 YEARS	X
PAYMENT	INR 45000 per acre per year	INR 75000 per acre per year	X

FIGURE 2 | Sample of choice card used in the choice experiment (based on study design used in Puri et al., 2021).

was identified based on Akaike's Information Criterion (AIC) values (Burnham and Anderson, 2002).

Lastly, we calculated the reservation price or the amount that would need to be paid to a landowner as a monetary incentive to enroll in the wildlife-friendly land-use program. This amount is the value at which the respondents switch from being unwilling to enroll in the program to willing. The value is derived by estimating the best fit indirect utility function (using a basic MNL model), and setting it equal to zero. The model included respondent characteristics that we assumed had direct policy relevance such as landholding size, distance to PA, experience with crop loss in the previous year, and annual agricultural income.

RESULTS

Landowner Characteristics

We surveyed 699 landowners who were responsible for making decisions concerning their respective land parcels. When long-term decisions regarding land management were discussed, women were not comfortable participating in the choice experiment. They often deferred judgment to the males of the household. Of the surveyed landowners, 98% were males and 2% were females with an average age of 47 (range: 19–93). About half of the landowners reported earning between INR (Indian rupees) 25,000 and 100,000 (US$ 357 and US$ 1428, US$ 1 = 70 INR at the time of the survey) as annual agricultural income (**Table 2**). The average household size was 5 (range: 1–15). Almost all (86%) respondents owned livestock. Most landowners (73%) were smallholders owning less than 5 acres of land. On average, these landowners grew three crops in a year (range: 0–10). Landowners grew nearly 60 varieties of crops, and the top five crops included cotton (48%), finger millet (38%), maize (32%), banana (21%), and tobacco (20%). Among these, finger millet is produced mostly for subsistence purposes, while the rest are produced primarily for commercial sale. Majority of the landowners (84%) reported facing losses due to wildlife, with crop damage (83%) being the major form of loss.

Model Estimates

Of the total surveyed landowners, 81% were interested in adopting at least one of the offered programs. Landowners opted out of the programs (i.e., rejected all offered programs) for the following reasons: (1) continue current agricultural practices (10%), (2) insufficient economic benefit (10%), (3) lack of available land to commit to the program (3%), (4) lack of faith in the institutions (2%), and (5) water scarcity (2%). We found the ASC to be significantly positive indicating that the landowners do not have an inherent preference for the status quo (i.e., they preferred to enroll in at least one of the offered programs). Across the models, we found that landowners preferred programs of shorter duration [β (SE) = −0.057 (0.008)], and enrollment of smaller parcels of land [β (SE) = −0.015 (0.001), **Table 3**]. We also found significant positive coefficient for "Payment" suggesting that farmers are more likely to choose programs with higher payment [β (SE) = 0.009 (0.001)].

TABLE 2 | Landowner characteristics.

Characteristics	Sub-characteristics	Percentage (%)
Education	Illiterate	26
	<10th grade	43
	10th grade pass	15
	12th grade pass	7
	Undergraduate	5
	Graduate	4
Number of years living next to the PA	Upto 5 years	2
	6–19 years	3
	20–49 years	14
	>50 years	81
Agricultural land size	<3 acres	28
	3–5 acres	45
	6–10 acres	18
	>10 acres	9
Crops grown	Commercial	71
	Vegetables	51
	Food grain	23
	Horticulture and others	6
Livestock owned	Cattle	84
	Sheep	20
	Goat	13
	Poultry	9
	Buffalo	1
Agricultural income	<INR 10,000 (US$ 143)	6
	INR 10,001–24,999 (US$ 143–357)	17
	INR 25,000–49,999 (US$ 357–714)	24
	INR 50,000–100,000 (US$ 714–1428)	24
	>INR 100,000 (US$ 1428)	29
Non-agricultural income	Dairy and livestock husbandry	46
	Daily wage labor	26
	Pension	25
	Service job	9
	Business	6
	Non-service job	3

Random Parameter Logit models showed that the landowners are heterogeneous in their preference for the proportion of land and number of years, represented by the statistical significance of the estimated standard deviation for "Land" and "Year" attributes, i.e., some landowners preferred enrolling larger parcels of land and for longer program duration (**Table 3**). On interacting the socioeconomic, demographic and geographic variables with the "Land" and "Year" attributes, we found that respondents who had a longer history of living next to the PA, owned larger parcels of land, and grew fewer numbers of commercial crops preferred to enroll larger parcels of land. Concurrently, respondents who grew more commercial crops preferred to enroll for longer duration (**Table 3**).

We calculated the reservation price using parameter estimates from the basic MNL model (**Supplementary Table 1**). The top model included land size, distance to PA, and number of commercial crops grown as the explanatory variables. The estimated average monetary incentive value amounted to INR 64,000 (US$ 914) per acre per year.

Tourism Potential on Wildlife-Friendly Land

We found that majority of the surveyed landowners were interested in conducting tourism activities such as guided walks and tours in the village (74%), creating accommodation facilities in their house (71%), or hosting overnight stays in their *machan*/tree house (69%) in the land set aside for the wildlife-friendly land-use program. Most (76%) landowners considered financial support in the form of interest-free loans would be necessary, 67% sought support from the government for planning and marketing, and 59% believed training and skill development would be useful.

DISCUSSION

Globally, we are witnessing rapid biodiversity loss, unsustainable land-use practices and rising climate change – collectively comprising the triad of challenges of the Anthropocene (Kremen and Merenlender, 2018). Finding conservation solutions to this triad of interlinked challenges requires a consolidated approach that takes social, ecological, and economic perspectives into consideration. India is among the few developing nations predicted to be most vulnerable to climate change, and this will likely exacerbate the existing food insecurity, poverty, inequality, and undernutrition (IPCC, 2014; Mendelsohn, 2014). In addition to the sociological challenges, wildlife habitats are shrinking within and outside PAs and intensive land-use change has affected animal movement and biodiversity (Jayadevan et al., 2020; Nayak et al., 2020). Land management practices that aim to expand habitat for biodiversity beyond PAs through the participation of local stakeholders are needed to address the conservation issues being faced.

Through our study, we assessed the willingness of landowners living in the buffer of two premiere Indian PAs to participate in a voluntary, incentive-based wildlife-friendly land-use program. We examined how socioeconomic, demographic, and geographic factors influenced the willingness of landowners to participate in such programs. Our results highlight the importance of incorporating local willingness and the need for policy makers to integrate heterogeneity in preferences to make feasible conservation options.

We found that there is a high willingness among landowners to modify their ongoing agricultural practices and adopt more wildlife-friendly land-use. Most rural livelihoods around Bandipur and Nagarahole depend upon agro-pastoral farming for their primary source of income (Karanth et al., 2013; Margulies and Karanth, 2018). However, there is an increasing dependency on non-farming sectors for income in rural India, including the study landscape (Pingali et al., 2019; Li et al., 2020). The lower contribution of agriculture toward household income in the landscape due to high input costs, increasing levels of uncertainty in agricultural yields, and high-interest rates on agricultural credit have prompted farmers to adopt low-risk

TABLE 3 | Results of the Multinomial Logit (MNL) and Random Parameter Mixed Logit (RPL), estimations, standard errors in parentheses.

	MNL model	RPL model		RPL model with interactions	
Attributes and interaction	**Coefficient (SE)**	**Coefficient (SE)**	**Coeff.Std. (SE)**	**Coefficient (SE)**	**Coeff.Std. (SE)**
ASC	1.012*** (0.141)	5.006*** (0.588)	6.931*** (0.718)	3.297*** (0.249)	
Land	−0.015*** (0.001)	−0.022*** (0.002)	0.026*** (0.004)	−0.084*** (0.018)	0.043*** (0.004)
Year	−0.057*** (0.008)	−0.081*** (0.014)	0.140*** (0.026)	−0.229** (0.114)	0.297*** (0.026)
Payment	0.009*** (0.001)	0.013*** (0.002)	–	0.012*** (0.002)	–
Land × family size	–	–	–	−0.0006 (0.001)	–
Land × Edu	–	–	–	0.001 (0.002)	–
Land × land size	–	–	–	0.008*** (0.003)	–
Land × years living	–	–	–	0.010** (0.004)	–
Land × Dist PA	–	–	–	0.001 (0.001)	–
Land × HH div				−0.0006 (0.002)	–
Land × Comm	–	–	–	−0.006** (0.002)	–
Year × family size	–	–	–	−0.0002 (0.008)	–
Year × Edu	–	–	–	0.004 (0.012)	–
Year × land size	–	–	–	0.008 (0.018)	–
Year × years living	–	–	–	0.008 (0.025)	–
Year × Dist PA	–	–	–	−0.003 (0.008)	–
Year × HH div	–	–	–	−0.014 (0.015)	–
Year × Comm	–	–	–	0.029* (0.016)	–
Log-likelihood	−2168.96	−1733.08		−1847.69	
AIC/N	2.072	1.66		1.78	
N (observations)	2097	2097		2097	

Estimates are from the best-fit model based on AIC.
ASC, alternative specific constant; HH div, household income diversity; Comm, number of commercial crops grown; Dist PA, distance to protected area from the survey location; Edu, education of the landowner; years living, number of years living next to the protected area.
$^*p < 0.1$; $^{**}p < 0.05$; $^{***}p < 0.01$; Coeff.Std., coefficients of standard deviation.

farming strategies as well as seek opportunities to supplement cash incomes (Margulies and Karanth, 2018; Pingali et al., 2019; Li et al., 2020). These results mirror the willingness of farmers in Pakistan who were inclined toward growing trees on their land. Farmers attribute multiple benefits to trees, compared with traditional cropping where they face various constraints, including inadequate access to credit, natural calamities, and limited support from local authorities (Mahmood and Zubair, 2020). This shift in the rural agricultural economy highlights the opportunity for private land conservation through capacity building and skill development of rural landowners.

We found that landowners preferred short-term programs, although there was heterogeneity in their preferences. This can be attributed to a faster turnover of benefits, and the uncertainty associated with signing long-term contracts. Creating wildlife-friendly land-use as proposed in this study can take many years; practitioners should include growing native trees and fruiting tree species that require a shorter growth period and yield quicker economic returns. We also found that people prefer to enroll smaller parcels of land (with heterogeneous preferences) pointing to the risks and uncertainties associated with land conversion. Since the agricultural lands around these PAs are usually small and highly productive, it makes it harder to opt for alternative forms of land-use as seen in a "greening" program proposed to German farmers (Schulz et al., 2013). Preference for shorter programs and reluctance to enroll larger land parcels can be attributed to loss aversion from adopting programs with unknown returns (Moon and Cocklin, 2011; Haile et al., 2019). Another potential reason could be the absence of similar programs in the study landscape or country. Higher payment amounts tend to attract landowners toward programs like wildlife-friendly land-use, which is in accordance with other studies that showed that landowners who are dependent on farming income tend to go for higher compensation (Broch and Vedel, 2012; Schulz et al., 2013). Some landowners also suggested that even if the profit is less in the initial stages of the program, the profit margin will rise in the subsequent years as they will have to invest only in the upkeep.

Large landholders were more likely to choose programs that required enrolling larger parcels of land. This corroborates the results of Haile et al. (2019) who also found that large landholders had a stronger preference for alternative land-use programs rather than remaining in the status quo. Our results suggest that landowners who had a long history of living next to the PA also chose programs that required enrolling larger land parcels. This could be due to the ecological value they associate with living adjacent to a forest. During field surveys, about half of the landowners reported that the forests regulate climate and rainfall, improve quality of life and provide livelihoods. Farmers in Pakistan have also been found to attribute environmental benefits such as reduced pollution and soil erosion to increased tree cover (Zubair and Garforth, 2006). We also found that landowners who grew more commercial crops are less likely to participate in programs that require enrolling large pieces of land. While our results also suggest that landowners who grew more commercial crops are more likely to choose programs with longer contract periods, the association is weak.

Conservation interventions can be constrained due to lack of resources, lack of faith in institutions, and limited economic benefits (Rasul and Thapa, 2006; Dhakal and Rai, 2020; Dhyani et al., 2021). The offered payment amounts in the programs were, at times, insufficient to gain farmer interest, as continuing with ongoing agricultural practices was perceived to be more beneficial. Unfamiliarity with a program coupled with a perception about the capacity of small landholdings to generate insufficient returns through wildlife-friendly land-use can lead to lower participation levels in conservation programs. Program participation can be augmented through outreach and extension activities including awareness campaigns, knowledge sharing, and capacity building.

The average monetary incentive of INR 64,000 (US$ 914) per acre per year derived from our study is higher than the average annual income from cultivation (US$ 537) across Indian states (NABARD, 2018). Previous studies from India have estimated the monetary value to be paid to the landowners for leaving land uncultivated (US$ 1429 per acre per year), or for the services derived from farmland (US$ 970 per acre per year) (Devi et al., 2017; Badola et al., 2021). The monetary amount offered may vary across the country given variation in local socio-economic contexts, agricultural incomes and land productivity. As such, we recommend that funds from CAMPA (Compensatory Afforestation Fund Management and Planning Authority), an afforestation program under the central government of India that procures compensation from forest diversion activities, can be utilized for incentivizing rural landowners and encouraging wildlife-friendly land-use adoption. In the year 2019–2020, CAMPA released INR 13.5 billion (US$ 192.91 million) to the state of Karnataka where the study landscape is located (MoEFCC/GOI, 2020). Beyond government subsidies and incentives, there is also scope for public-private partnerships, which mobilizes private finance in landscape initiatives to leverage private sector investment (Clarvis, 2014).

Sustaining Wildlife-Friendly Land Through Tourism

Wildlife-friendly land-use can expand conservation efforts beyond PAs by securing habitat for wildlife and creating alternative livelihood opportunities for local landowners through market-based instruments such as tourism. Currently, in India, PAs listed as Tiger Reserves practice a safari-based tourism model while non-Tiger Reserves conduct safaris and other activities such as nature walks, boating, trekking, and camping (Karanth and DeFries, 2011; Puri et al., 2018). Across India, the need to expand tourism beyond core PAs and integrate buffer areas is being recognized. This would help offset the pressure of a high number of tourists restricted within 10–20% of PA. The existing ecotourism policy in India defines ecotourism as "responsible travel to natural areas that conserves the environment and improves the well-being of local people" (MoEFCC, 2018). This establishes local communities as the principal stakeholders in ecotourism ventures. It suggests that buffer areas, private lands, revenue lands, and Reserved Forests around PAs with high-quality wildlife habitat be developed for ecotourism to reduce pressure on sensitive "core" areas of reserves. This will simultaneously increase benefits to local communities. However, current ecotourism practices benefit private tourism enterprises and government agencies, with local communities reduced to mere bystanders (Karanth and DeFries, 2011; Rastogi et al., 2015). There is little participation of local communities, and those that benefit from tourism either live close to safari gates or belong to socially elite classes (Rastogi et al., 2015). While most of the interviewed landowners expressed positive interest in bringing tourism into the wildlife-friendly land developed through our proposed program, financial support continues to be a key catalyst in undertaking such initiatives. Communities should also be provided with essential training and skill development for hosting tourists, better communication and management skills. More inclusive ecotourism ventures will involve and benefit marginalized communities, simultaneously paving the route for sustainable, long-term ecotourism on private lands surrounding PAs.

In a rapidly growing economy where exclusive reliance on agriculture is challenging, our approach suggests a way of designing incentive-based mechanisms that incorporate the needs of key stakeholders while moving conservation beyond current PA networks. Our approach can be used to assess the feasibility of alternative conservation actions outside PAs based on the site-specific preferences identified in collaboration with local landowners. Future efforts to encourage landowners to engage in conservation practices on land adjoining PAs will facilitate the expansion of critical wildlife habitats and strengthen PA connectivity, which is currently poor (Ward et al., 2020). Finally, our research provides information for landscape-scale policy development and equips us with a baseline understanding of the long-term sustainability of conservation efforts derived through modification of land-use and tourism.

DATA AVAILABILITY STATEMENT

The datasets presented in this article are not readily available because it includes information that could compromise the privacy of the research participants. Data supporting the findings of this study are available on request from the corresponding author. Requests to access the datasets should be directed to DM, dincy.mariyam@cwsindia.org.

ETHICS STATEMENT

The studies involving human participants were reviewed and approved (dated 24.12.2018) by the Human Subjects Review Board of the Centre for Wildlife Studies, Bengaluru, India.

Written informed consent for participation was not required for this study in accordance with the national legislation and the institutional requirements.

AUTHOR CONTRIBUTIONS

DM: data curation, formal analysis, investigation, methodology, project administration, software, visualization, and writing – original draft. MP: conceptualization, formal analysis, methodology, software, and visualization. AH: formal analysis. KK: conceptualization, funding acquisition, project administration, resources, supervision, and validation. All authors contributed to manuscript revision, read and approved the submitted version.

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SUPPLEMENTARY MATERIAL

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REFERENCES

Anand, M. O., Krishnaswamy, J., Kumar, A., and Bali, A. (2010). Sustaining biodiversity conservation in human-modified landscapes in the Western Ghats: remnant forests matter. *Biol. Conserv.* 143, 2363–2374. doi: 10.1016/j.biocon.2010.01.013

Badola, R., Ahmed, T., Gill, A. K., Dobriyal, P., and Das, G. C. (2021). An incentive - based mitigation strategy to encourage coexistence of large mammals and humans along the foothills of Indian Western Himalayas. *Sci. Rep.* 11:5235. doi: 10.1038/s41598-021-84119-7

Baskaran, N., Kannan, G., and Anbarasan, U. (2013). A landscape-level assessment of Asian elephant habitat, its population and elephant – human conflict in the Anamalai hill ranges of southern Western Ghats. India. *Mamm. Biol.* 78, 470–481. doi: 10.1016/j.mambio.2013.04.007

Bhagwat, S. A., Willis, K. J., Birks, H. J. B., and Whittaker, R. J. (2008). Agroforestry: a refuge for tropical biodiversity? *Trends Ecol. Evol.* 23, 261–267. doi: 10.1016/j.tree.2008.01.005

Bliemer, M. C. J., and Collins, A. T. (2016). On determining priors for the generation of efficient stated choice experimental designs. *J. Choice Model.* 21, 10–14. doi: 10.1016/j.jocm.2016.03.001

Broch, S. W., and Vedel, S. E. (2012). Using choice experiments to investigate the policy relevance of heterogeneity in farmer agri-environmental contract preferences. *Environ. Resour. Econ.* 51, 561–581. doi: 10.1007/s10640-011-9512-8

Burnham, K. P., and Anderson, D. R. (2002). *Model Selection and Multimodel Inference. A Practical Information-Theoretic Approach*. New York, NY: Springer.

Capano, G. C., Toivonen, T., Soutullo, A., and Di Minin, E. (2019). The emergence of private land conservation in scientific literature: a review. *Biol. Conserv.* 237, 191–199. doi: 10.1016/j.biocon.2019.07.010

Carlsson, F., Frykblom, P., and Liljenstolpe, C. (2003). Valuing wetland attributes: an application of choice experiments. *Ecol. Econ.* 47, 95–103. doi: 10.1016/j.ecolecon.2002.09.003

Carlsson, F., Kataria, M., and Lampi, E. (2010). Dealing with ignored attributes in choice experiments on valuation of Sweden's environmental quality objectives. *Environ. Resour. Econ.* 47, 65–89. doi: 10.1007/s10640-010-9365-6

CBD (2010). *The Strategic Plan for Biodiversity 2011–2020 and the Aichi Biodiversity Targets*. Nagoya: CBD.

CBD (2020). *Zero Draft of the Post-2020 Global Biodiversity Framework (CBD/WG2020/2/3, CBD, 2020)*. Rome: International Union For Conservation Of Nature.

Census (2011). *Census of India 2011*. New Delhi. Available online at: https://censusindia.gov.in/2011census/ (accessed October 26, 2020).

Clarvis, M. H. (2014). "Review of financing institutions and mechanisms," in *Financing Strategies for Integrated Landscape Investment*, ed. S. Shames (Washington, DC: EcoAgriculture Partners).

Clements, H. S., Kerley, G. I. H., Cumming, G. S., De Vos, A., and Cook, C. N. (2018). Privately protected areas provide key opportunities for the regional persistence of large- and medium-sized mammals. *J. Appl. Ecol.* 56, 537–546. doi: 10.1111/1365-2664.13300

Crooks, K. R., Burdett, C. L., Theobald, D. M., Rondinini, C., and Boitani, L. (2011). Global patterns of fragmentation and connectivity of mammalian carnivore habitat. *Philos. Trans. R. Soc. B Biol. Sci.* 366, 2642–2651. doi: 10.1098/rstb.2011.0120

Devi, P. I., Kumar, L., Suresh Kumar, D., Manjula, M., Mukhopadhyay, P., Raghu, P., et al. (2017). Payment for ecosystem services guaranteed farm income and sustainable agriculture. *Econ. Polit. Wkly.* 52, 12–14.

Devidas, S., and Puyravaud, J.-P. (1995). Primary productivity of the herbaceous layer in a grazed savanna woodland. Bandipur National Park, southern India. *Acta Ecol.* 16, 491–505.

Dhakal, A., and Rai, R. K. (2020). Who adopts agroforestry in a subsistence economy?— Lessons from the Terai of Nepal. *Forests* 11, 1–15. doi: 10.3390/f11050565

Dhyani, S., Murthy, I. K., Kadaverugu, R., Dasgupta, R., Kumar, M., and Gadpayle, K. A. (2021). Agroforestry to achieve global climate adaptation and mitigation targets: are south Asian countries sufficiently prepared? *Forests* 12, 1–21.

Drescher, M., and Brenner, J. C. (2018). The practice and promise of private land conservation. *Ecol. Soc.* 23:3. doi: 10.5751/ES-10020-230203

ENVIS (2021). *Protected Areas of India*. Available online at at: http://www.wiienvis.nic.in/Database/Protected_Area_854.aspx. (accessed May 24, 2021).

Ferreira, A. S., Peres, C. A., Dodonov, P., and Cassano, C. R. (2020). Multi-scale mammal responses to agroforestry landscapes in the Brazilian Atlantic Forest: the conservation value of forest and traditional shade plantations. *Agrofor. Syst.* 94, 2331–2341. doi: 10.1007/s10457-020-00553-y

Geldmann, J., Manica, A., Burgess, N. D., Coad, L., and Balmford, A. (2019). A global-level assessment of the effectiveness of protected areas at resisting anthropogenic pressures. *Proc. Natl. Acad. Sci. U. S. A.* 116, 23209–23215. doi: 10.1073/pnas.1908221116

Ghosh-Harihar, M., An, R., Athreya, R., Borthakur, U., Chanchani, P., Chetry, D., et al. (2019). Protected areas and biodiversity conservation in India. *Biol. Conserv.* 237, 114–124. doi: 10.1016/j.biocon.2019.06.024

Gooden, J. L. (2019). Cultivating identity through private land conservation. *People Nat.* 1, 362–375. doi: 10.1002/pan3.32

Green, W. H. (2012). *NLOGIT version 5.0: Reference Guide*. New York, NY: Econometric Software Inc.

Haile, K. K., Tirivayi, N., and Tesfaye, W. (2019). Farmers' willingness to accept payments for ecosystem services on agricultural land: the case of climate-smart agroforestry in Ethiopia. *Ecosyst. Serv.* 39:100964. doi: 10.1016/j.ecoser.2019.100964

Hartung, C., Lerer, A., Anokwa, Y., Tseng, C., Brunette, W., and Borriello, G. (2010). "Open data kit: tools to build information services for developing regions," in *Proceedings of the 4th ACM/IEEE International Conference on Information and Communication Technologies and Development ICTD '10* ((New York, NY: Association for Computing Machinery), doi: 10.1145/2369220.2369236

Harvey, C. A., Komar, O., Chazdon, R., Ferguson, B. G., Finegan, B., Griffith, D. M., et al. (2008). Integrating agricultural landscapes with biodiversity conservation in the Mesoamerican hotspot. *Conserv. Biol.* 22, 8–15. doi: 10.1111/j.1523-1739.2007.00863.x

Hensher, D. A., Rose, J. M., and Greene, W. H. (2005). *Applied Choice Analysis: A Primer*. Cambridge: Cambridge University Press, doi: 10.1007/9781316136232

Hijmans, R. J., Williams, E., and Vennes, C. (2016). *Geosphere: Spherical Trigonometry. R Package. version 1.5–5.* Available online at: https://cran.r-project.org/web/packages/geosphere/index.html (accessed September 12, 2019)

Hughes, K., Morgan, S., Baylis, K., Oduol, J., Smith-dumont, E., Vågen, T., et al. (2020). Assessing the downstream socioeconomic impacts of agroforestry in Kenya. *World Dev.* 128, 104835. doi: 10.1016/j.worlddev.2019.104835

IPCC (2014). "Summary for policymakers," in *Climate Change 2014: Impacts, Adaptation, and Vulnerability. Part A: Global and Sectoral Aspects. Contribution of Working Group II to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change*, eds C. B. Field, V. R. Barros, D. J. Dokken, K. J. Mach, M. D. Mastrandrea, T. E. Bilir, et al. (Cambridge: Cambridge University Press), 1-32.

Jathanna, D., Karanth, K. U., Kumar, N. S., Goswami, V. R., and Vasudev, D. (2015). Reliable monitoring of elephant populations in the forests of India: analytical and practical considerations. *Biol. Conserv.* 187, 212–220. doi: 10.1016/j.biocon.2015.04.030

Jayadevan, A., Nayak, R., Karanth, K. K., Krishnaswamy, J., Defries, R., Karanth, K. U., et al. (2020). Navigating paved paradise: evaluating landscape permeability to movement for large mammals in two conservation priority landscapes in India. *Biol. Conserv.* 247:108613. doi: 10.1016/j.biocon.2020.108613

Jones, K. R., Venter, O., Fuller, R. A., Allan, J. R., Maxwell, S. L., Negret, P. J., et al. (2018). One-third of global protected land is under intense human pressure. *Science* 360, 788–791. doi: 10.1126/science.aap9565

Kalam, T., Kumar Baishya, H., and Smith, D. (2018). Lethal fence electrocution: a major threat to Asian elephants in Assam, India. *Trop. Conserv. Sci.* 11, 1–8. doi: 10.1177/1940082918817283

Kamal, S., Grodzinska-Jurczaka, M., and Brown, G. (2015). Conservation on private land: a review of global strategies with a proposed classification system. *J. Environ. Plan. Manag.* 58, 576–597. doi: 10.1080/09640568.2013.875463

Karanth, K. K., and DeFries, R. (2011). Nature-based tourism in Indian protected areas: new challenges for park management. *Conserv. Lett.* 4, 137–149. doi: 10.1111/j.1755-263X.2010.00154.x

Karanth, K. K., Gopalaswamy, A. M., Prasad, P. K., and Dasgupta, S. (2013). Patterns of human – wildlife conflicts and compensation: insights from Western Ghats protected areas. *Biol. Conserv.* 166, 175–185. doi: 10.1016/j.biocon.2013.06.027

Karanth, K. K., Jain, S., and Mariyam, D. (2017). "Emerging trends in wildlife and tiger tourism in India," in *Nature Tourism*, eds J. S. Chen and N. K. Prebensen (Milton Park: Routledge), 159–170. doi: 10.4324/9781315659640

Karanth, K. K., and Kudalkar, S. (2017). History, location, and species matter: insights for human– wildlife conflict mitigation from India. *Hum. Dimens. Wildl.* 22, 331–346. doi: 10.1080/10871209.2017.1334106

Karanth, K. K., and Vanamamalai, A. (2020). Wild Seve: a Novel conservation intervention to monitor and address human-wildlife conflict. *Front. Ecol. Evol.* 8:198. doi: 10.3389/fevo.2020.00198

Karanth, K. U., and Karanth, K. K. (2012). A tiger in the drawing room: can luxury tourism benefit wildlife? *Econ. Polit. Wkly.* 47, 38–43.

Karanth, K. U., Kumar, N. S., and Karanth, K. K. (2020). Tigers against the odds: applying macro-ecology to species recovery in India. *Biol. Conserv.* 252:108846. doi: 10.1016/j.biocon.2020.108846

Kareiva, P., and Marvier, M. (2012). What Is Conservation Science? *BioScience* 62, 962–969. doi: 10.1525/bio.2012.62.11.5

Kremen, C., and Merenlender, A. M. (2018). Landscapes that work for biodiversity and people. *Science.* 362:eaau6020. doi: 10.1126/science.aau6020

Kremen, C., and Miles, A. (2012). Ecosystem services in biologically diversified versus conventional farming systems: benefits, externalities, and trade-offs. *Ecol. Soc.* 17:40.

Kumaraswamy, S., and Kunte, K. (2013). Integrating biodiversity and conservation with modern agricultural landscapes. *Biodivers. Conserv.* 22, 2735–2750. doi: 10.1007/s10531-013-0562-9

Li, B. V., Reardon, K., Satheesh, N., Liu, C., and Karanth, K. K. (2020). Effects of livestock loss and emerging livestock types on livelihood decisions around protected areas: case studies from China and India. *Biol. Conserv.* 248:108645. doi: 10.1016/j.biocon.2020.108645

Louviere, J. J., Wasi, N., Street, D., Islam, T., and Marley, A. A. J. (2008). Modeling the choices of individual decision-makers by combining efficient choice experiment designs with extra preference information. *J. Choice Model.* 1, 128–164. doi: 10.1016/S1755-5345(13)70025-3

Maciejewski, K., Baum, J., and Cumming, G. S. (2016). Integration of private land conservation areas in a network of statutory protected areas: implications for sustainability. *Biol. Conserv.* 200, 200–206. doi: 10.1016/j.biocon.2016.05.027

Mahmood, M. I., and Zubair, M. (2020). Farmer's perception of and factors influencing agroforestry practices in the Indus river basin. Pakistan. *Small Scale For.* 19, 107–122. doi: 10.1007/s11842-020-09434-9

Margulies, J. D., and Karanth, K. K. (2018). The production of human-wildlife conflict: a political animal geography of encounter. *Geoforum* 95, 153–164. doi: 10.1016/j.geoforum.2018.06.011

Mendelsohn, R. (2014). The impact of climate change on agriculture in Asia. *J. Integr. Agric.* 13, 660–665. doi: 10.1016/S2095-3119(13)60701-7

MoEFCC (2018). *Policy for Eco-Tourism in Forest and Wildlife Areas*. New Delhi: Government of India.

MoEFCC/GOI (2020). *E-Green Watch Funds Management System*. Available online at: http://egreenwatch.nic.in/Funds/Public/SummaryFundsTransferToState.aspx?StateId1=MTc= (accessed January 5, 2021)

Moon, K., and Cocklin, C. (2011). A landholder-based approach to the design of private-land conservation programs. *Conserv. Biol.* 25, 493–503. doi: 10.1111/j.1523-1739.2010.01639.x

NABARD (2018). *NABARD All India Rural Financial Inclusion Survey 2016-17*. Mumbai. Available online at: http://library1.nida.ac.th/termpaper6/sd/2554/19755.pdf (accessed December 22, 2020).

Naidoo, R., Gerkey, D., Hole, D., Pfaff, A., Ellis, A. M., Golden, C. D., et al. (2019). Evaluating the impacts of protected areas on human well-being across the developing world. *Sci. Adv.* 5:eaav3006. doi: 10.1126/sciadv.aav3006

Nayak, R., Karanth, K. K., Dutta, T., Defries, R., and Karanth, K. U. (2020). Bits and pieces: forest fragmentation by linear intrusions in India. *Land Use policy* 99:104619. doi: 10.1016/j.landusepol.2020.104619

Ogra, M. V. (2008). Human – wildlife conflict and gender in protected area borderlands: a case study of costs, perceptions, and vulnerabilities from Uttarakhand (Uttaranchal), India. *Geoforum* 39, 1408–1422. doi: 10.1016/j.geoforum.2007.12.004

Oldekop, J. A., Holmes, G., Harris, W. E., Evans, K. L., and Evaluaci, U. (2015). A global assessment of the social and conservation outcomes of protected areas. *Conserv. Biol.* 30, 133–141. doi: 10.1111/cobi.12568

Pacifici, M., Di Marco, M., and Watson, J. E. M. (2020). Protected areas are now the last strongholds for many imperiled mammal species. *Conserv. Lett.* 13:e12748. doi: 10.1111/conl.12748

Pegas, F. V., and Castley, J. G. (2014). Ecotourism as a conservation tool and its adoption by private protected areas in Brazil. *J. Sustain. Tour.* 22, 604–625. doi: 10.1080/09669582.2013.875550

Pimm, S. L., Jenkins, C. N., and Li, B. V. (2018). How to protect half of earth to ensure it protects sufficient biodiversity. *Sci. Adv.* 4, 1–9. doi: 10.1126/sciadv.aat2616

Pingali, P., Aiyar, A., Abraham, M., and Rahman, A. (2019). *Transforming Food Systems for a Rising India*. Palgrave Macmillan: Cham, doi: 10.1007/978-3-030-14409-8

Puri, M., Karanth, K. K., and Thapa, B. (2018). Trends and pathways for ecotourism research in India. *J. Ecotourism* 18, 122–141. doi: 10.1080/14724049.2018.1474885

Puri, M., Piennar, E. F., Karanth, K. K., and Loiselle, B. A. (2021). Food for thought – examining farmers' willingness to engage in conservation stewardship around a protected area in central India. *Ecol. Soc.* 26:46. doi: 10.5751/ES-12544-260246

Rastogi, A., Hickey, G., Anand, A., Badola, R., and Hussain, S. A. (2015). Wildlife-tourism, local communities and tiger conservation: a village-level study in Corbett Tiger Reserve, India. *For. Policy Econ.* 61, 11–19. doi: 10.1016/j.forpol.2015.04.007

Rasul, G., and Thapa, G. B. (2006). Financial and economic suitability of agroforestry as an alternative to shifting cultivation: the case of the. *Agric. Syst.* 91, 29–50. doi: 10.1016/j.agsy.2006.01.006

Ricciardi, V., Mehrabi, Z., Wittman, H., James, D., and Ramankutty, N. (2021). Higher yields and more biodiversity on smaller farms. *Nat. Sustain.* 1–7. doi: 10.1038/s41893-021-00699-2

Saura, S., Bertzky, B., Bastin, L., Battistella, L., Mandrici, A., and Dubois, G. (2018). Protected area connectivity: shortfalls in global targets and country-level priorities. *Biol. Conserv.* 219, 53–67. doi: 10.1016/j.biocon.2017.12.020

Schulz, N., Breustedt, G., and Latacz-Lohmann, U. (2013). Assessing farmers' willingness to accept "greening": insights from a discrete choice experiment in Germany. *J. Agric. Econ.* 65, 26–48. doi: 10.1111/1477-9552.12044

Selinske, M. J., Coetzee, J., Purnell, K., Knight, A. T., and Lombard, A. T. (2015). Understanding the motivations, satisfaction, and retention of landowners in private land conservation programs. *Conserv. Lett.* 8, 282–289. doi: 10.1111/conl.12154

Stokstad, E. (2020). Global efforts to protect biodiversity fall short. *Science* 369:1418. doi: 10.1126/science.369.6510.1418

Talukdar, S., and Gupta, A. (2017). Attitudes towards forest and wildlife, and conservation-oriented traditions, around Chakrashila Wildlife Sanctuary, Assam, India. *Oryx* 52, 508–518. doi: 10.1017/S0030605316001307

Ward, M., Saura, S., Williams, B., Ramírez-Delgado, J. P., Arafeh-Dalmau, N., Allan, J. R., et al. (2020). Just ten percent of the global terrestrial protected area network is structurally connected via intact land. *Nat. Commun.* 11, 1–10. doi: 10.1038/s41467-020-18457-x

Watson, J. E. M., Dudley, N., Segan, D. B., and Hockings, M. (2014). The performance and potential of protected areas. *Nature* 515, 67–73. doi: 10.1038/nature13947

Zubair, M., and Garforth, C. (2006). Farm level tree planting in Pakistan: the role of farmers' perceptions and attitudes. *Agrofor. Syst.* 66, 217–229. doi: 10.1007/s10457-005-8846-z

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frontiers in Environmental Science

BRIEF RESEARCH REPORT
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Measuring Metrics of Climate Change and Its Implication on the Endangered Mammal Conservation in the Leuser Ecosystem

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The Leuser Ecosystem is one of the essential landscapes in the world for biodiversity conservation and ecosystem services. However, the Leuser Ecosystem has suffered many threats from anthropogenic activities and changing climate. Climate change is the greatest challenge to global biodiversity conservation. Efforts should be made to elaborate climatic change metrics toward biological conservation practices. Herein, we present several climate change metrics to support conservation management toward mammal species in the Leuser Ecosystem. We used a 30-year climate of mean annual temperature, annual precipitation, and the BIOCLIM data to capture the current climatic conditions. For the future climate (2050), we retrieved three downscaled general circulation models for the business-as-usual scenario of shared socioeconomic pathways (SSP585). We calculated the dissimilarities of the current and 2050 climatic conditions using the standardized Euclidean distance (SED). To capture the probability of climate extremes in each period (i.e., current and future conditions), we calculated the 5th and 95th percentiles of the distributions of monthly temperature and precipitation, respectively, in the current and future conditions. Furthermore, we calculated forward and backward climate velocities based on the mean annual temperature. These metrics can be useful inferences about species conservation. Our results indicate that almost all of the endangered mammals in the Leuser Ecosystem will occur in the area with threats to local populations and sites. Different conservation strategies should be performed in the areas likely to present different threats toward mammal species. Habitat restoration and long-term population monitoring are needed to support conservation in this mega biodiversity region.

Keywords: climate change, biodiversity, tropical landscape, mammal, conservation

INTRODUCTION

The Leuser Ecosystem is one of the essential landscapes in the world for biodiversity conservation and ecosystem services (Le Saout et al., 2013). It harbors various ecoregions such as tropical lowland and montane rainforests, coastal ecosystem, and peatland areas (Olson et al., 2001). This landscape represents the most critical refugia for many endangered mammals in Asia, for example, Sumatran orangutan, elephant, tiger, and rhinoceros, including a massive endemic diversity plant species (Cochard, 2017). Furthermore, the Leuser Ecosystem also contributed to regulating services like carbon sequestration (Warren et al., 2017), water provisioning, and economic benefits toward the local community (Janssen, 2003; Cochard, 2017).

Nevertheless, the Leuser Ecosystem has been subjected to anthropogenic activities that lead to deforestation and climate disturbance. Infrastructure development has been rapidly known as a potential threat to the Leuser Ecosystem (Sloan et al., 2018). Besides, illegal logging and agricultural expansion are also responsible for forest disturbance within the Leuser Ecosystem (Gaveau et al., 2009). Degradation of tropical peatlands within the area caused high carbon emissions to the atmosphere due to biomass loss, wildfires, and peat oxidation that lead to climate change (Page et al., 2011). Previous studies show that the destruction of tropical rainforests due to anthropogenic activities has been linked to population declines in many mammals such as Sumatran rhino, tiger, elephant, and orangutan (Kinnaird et al., 2003; Linkie et al., 2008; Wich et al., 2016; Weiskopf et al., 2019). Climate change can also induce habitat range contraction and distribution shifting for mammal species that lead to species extinction and extirpation (Chen et al., 2011; Dirzo et al., 2014; Ribeiro et al., 2016; Condro et al., 2021). Furthermore, the previous study also showed that the Leuser Ecosystem would experience significant biodiversity extirpations due to climate and habitat changes (Wich et al., 2016; Condro et al., 2021). However, there are still few studies about climate change impacts on biodiversity in the Leuser Ecosystem (Gaveau et al., 2009; Wich et al., 2016). Therefore, measuring the climate change impacts on biodiversity within the Leuser Ecosystem should be carried out to point out future conservation strategies.

Conservation strategies designed to improve biodiversity's resilience to climate change are inextricably linked to broader conservation activities (Game et al., 2011). Climate change metrics are crucial methods for mapping potential biodiversity risks in the future that helps predict which species are likely to adapt in space to a novel climate, migrate, and stay in a habitat with a newly suitable climate (Carroll et al., 2015). For instance, climate change metrics commonly used are climatic anomalies, change in climate extremes probability, and climatic change velocity (Borges and Loyola, 2020). Climate anomalies and change in climate extremes probability calculate the magnitude of change in the average and extreme conditions, respectively, at a given locality through time (Garcia et al., 2014). The velocity of climate change evaluates the exposure of species in the landscape to climate change (Loarie et al., 2009; Hamann et al., 2015). Those metrics allow us to assess whether the current protected areas will serve as refugia or suffer biodiversity extirpation and consequent changes in ecosystem processes (Araújo et al., 2011; Carroll et al., 2015). This study defines *refugia* as the areas with high future climatic stability (Watson et al., 2013; Sales et al., 2019).

Protected areas should be recognized as part of a comprehensive ecosystem-based management strategy that considers the complex and cumulative effects of anthropogenic activities. Nevertheless, taking climate change into account while designing and evaluating protected areas is still in its beginning (Brito-Morales et al., 2018). Herein, we present several metrics of the climate change, that is, standardized climatic anomalies, change in the probability of local climate extremes, and climate velocity to support conservation management toward endangered mammal biodiversity in the Leuser Ecosystem.

MATERIALS AND METHODS

Study Area

The study was conducted in the Leuser Ecosystem (**Figure 1**), located in the north of Sumatera, Indonesia. The Leuser Ecosystem covered two provincial administrative areas, that is, North Sumatra and Aceh provinces. The Leuser Ecosystem covers 2.6 million hectares and consists of either protected areas (i.e., nature conservation areas, nature reserve areas, and hunting parks) or unprotected areas (i.e., production forests, limited production forests, and convertible forests). The Leuser Ecosystem exemplifies how discrepancies in forest management by central and regional governments have undermined conservation planning and infrastructure growth (Sloan et al., 2018). We evaluated protected areas within the Leuser Ecosystem based on climate change metrics and mammal species richness.

Climatic Data

We used a 30-year climate of mean annual temperature, annual precipitation, and BIOCLIM variables at 30–arc second spatial resolution (~1 km) retrieved from WorldClim version 2.1 (Hijmans et al., 2005; Booth et al., 2014; Fick and Hijmans, 2017) for current climatic conditions. For the future climate in 2050, we retrieved three downscaled general circulation models (GCMs) for the business-as-usual scenario of shared socioeconomic pathways (SSP585) from the Coupled Model Intercomparison Project phase 6 multi-model dataset projections (Riahi et al., 2017). We followed the delta methods described by Navarro–Racines et al. (2020) to perform statistical downscaling of the GCM dataset and used a simple average to obtain the ensemble model from 3 GCMs (Watson et al., 2013). The three models used in the study were MIROC-ES2L and MIROC6 from the University of Tokyo, National Institute for Environmental Studies, and the Japan Agency for Marine-Earth Science and Technology (Tatebe et al., 2019; Hajima et al., 2020); and MRI-ESM 2.0 from the Meteorological Research Institute, Tsukuba, Japan (Yukimoto et al., 2019).

Metrics of Climate Change Calculations

Standardized Climatic Anomalies

We calculated the dissimilarities of the current and 2050 climatic conditions by using the standardized Euclidean distance (SED)

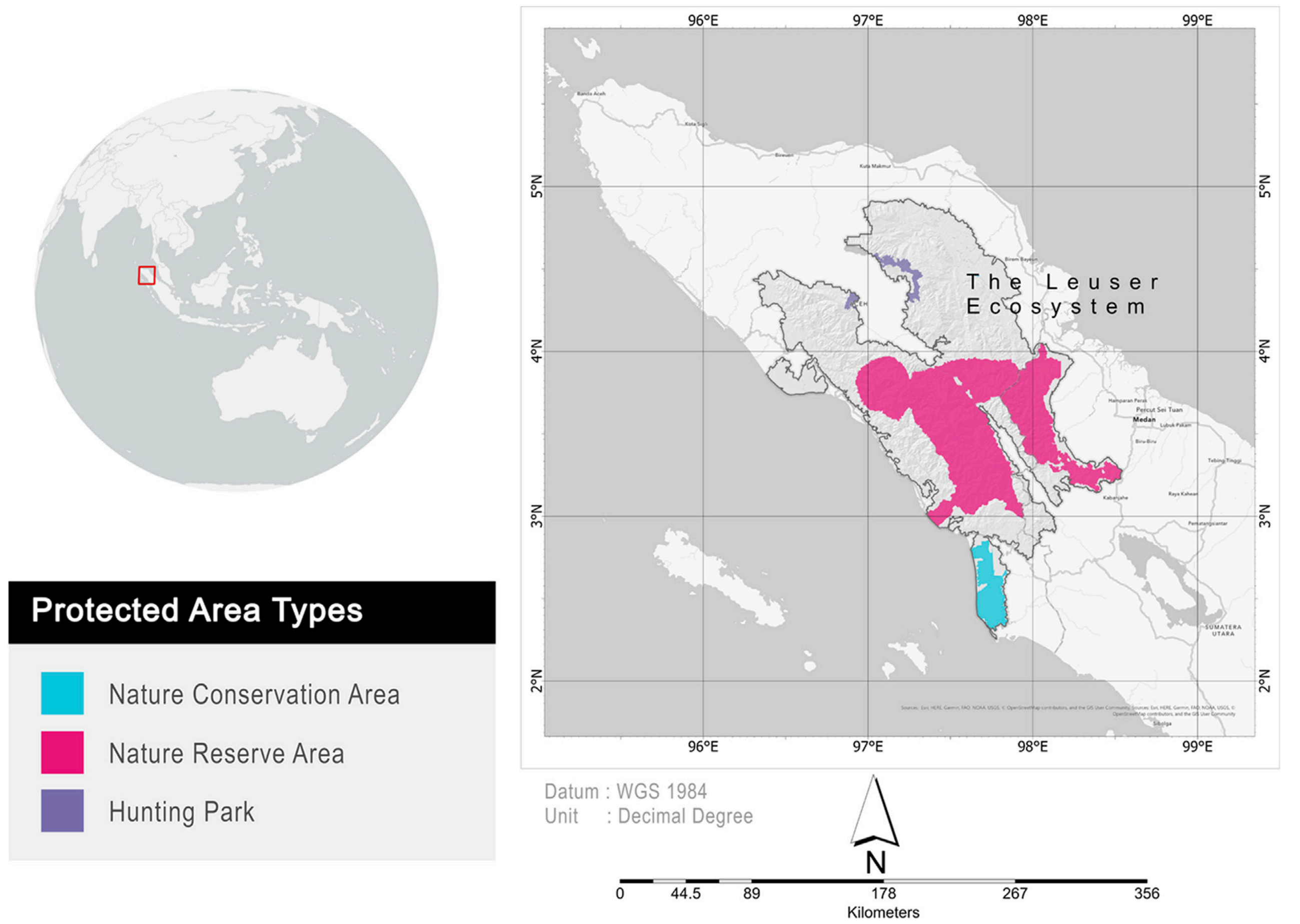

FIGURE 1 | Location of the Leuser Ecosystem and its protected areas. The conservation statuses of individual land management units were published by the Ministry of Environment and Forestry, Republic of Indonesia. Nature conservation area consists of national parks, nature recreation parks, and grand forest parks. Besides, the nature reserve area consists of strict nature reserve and wildlife reserve.

for temperature and precipitation according to Williams et al. (2007). This study used the mean annual temperature (Bio1) and annual precipitation (Bio12) to evaluate temperature and precipitation, respectively. The temporal differences for the climatic variables were standardized by the interannual standard deviation for both variables from the seasonality of temperature (Bio4) and seasonality of precipitation (Bio15) in the current condition. High SED values correspond to significant changes in temperature and precipitation (Garcia et al., 2014).

Change in Probability of Local Climate Extremes

To capture the probability of climate extremes in each period (i.e., current and future conditions), we calculated the 5th and 95th percentiles of the distributions of monthly temperature and precipitation, respectively, in the current and future conditions. Since the daily data were unavailable, the monthly data used here can point out to the information enclosed within the gradual trends (Garcia et al., 2014). We used both data to determine the probability of an extreme event of precipitation and temperature by a generalized extreme value distribution (Katz et al., 2005). To avoid double-counting in probabilities, we subtracted the sum of two probabilities (i.e., temperature and precipitation extremes) with the product of two probabilities. The change in the probability of local climate extremes was calculated by subtracting future probability with the current probability of climate extreme events. Negative values represented the decrease in the climate extreme events, whereas positive values indicated the increase in the climate extreme events in the future condition. The calculations captured dry and hot aspects of the climate extreme from the fifth percentile of precipitation and 95th percentile of temperature. On the other hand, wet and cold aspects of the climate extreme were retrieved from the 95th percentile of precipitation and fifth percentile of temperature, respectively. Several recent studies also used the monthly climatic data to capture the climate extremes (Albright et al., 2011; Suggitt et al., 2011; Stewart et al., 2021).

Climate Velocity

We calculated forward and backward climate velocities based on the mean annual temperature in the current and future conditions (Garcia et al., 2014; Carroll et al., 2015). Basically, climate change velocity is the ratio of the climatic parameter's temporal and spatial gradients (Loarie et al., 2009). This study performed the climate-analog velocity algorithm developed by Hamann et al. (2015) that calculates the distance of the climatic parameter from present to the future climate match using a rounding operation on the data to create more efficient calculation. The forward velocity represents the minimum distance the species in the current landscape has to migrate to maintain climate conditions in the future. Conversely, backward velocity shows how fast the species would have to migrate to colonize in a particular landscape (Loarie et al., 2009; Carroll et al., 2015).

Implications for Conservation

We investigated the protected area's susceptibility within the Leuser Ecosystem from the World Database on Protected Areas (WDPA; www.protectedplanet.net; accessed on March 13, 2021) based on standardized Euclidean distance (SED) by quantifying an average of the metrics within the protected area patches (Williams et al., 2007). We calculated the residence times (i.e., the maximum diameter of each protected area patch divided by velocity) to examine the interaction between protected area extents and climatic velocities that is needed to maintain pace with climate change (Loarie et al., 2009).

We created the bivariate plot based on forward and backward velocities to assess the four regions of threat for biodiversity within the study area. We classified the forward velocity and backward velocity values into two categories: values greater than the median and below the median. Low forward and backward velocities represent low threat, low forward and high backward velocities indicate threats to habitats, high forward and low backward velocities represent threats to local populations, and high for both velocities indicate threats to sites and populations (Carroll et al., 2015). Furthermore, we calculated the proportion of 45 endangered mammal species (i.e., near threatened, vulnerable, endangered, and critically endangered based on the IUCN red list categories) of interest and the geographic distribution within the multifaceted threats (Carroll et al., 2015; Borges and Loyola, 2020). We obtained the geographical distribution of species from the IUCN red list spatial data for terrestrial mammals (available from https://www.iucnredlist.org; accessed on March 13, 2021).

RESULTS

Metrics of Climate Change

We observed a climatic anomaly value across the landscape ranging from 0.25 to 0.57 ($\bar{x}$ = 0.32, SE = 0.001). The Leuser Ecosystem that was possibly exposed to a higher climate anomaly (combined temperature and precipitation) would have occurred in the northwest, southeast, and several montane areas within the landscape. Future precipitation anomalies ($\bar{x}$ = 0.27, SE = 0.002) would have likely greater values than the temperature anomaly ($\bar{x}$ = 0.17, SE = 0.002). Furthermore, a higher temperature anomaly was generated at high altitude areas, where the current climate conditions become locally disappeared as climate shifting becomes more pronounced (**Figure 2A**).

The fifth percentile of the extreme precipitation distribution was 131 mm/month, and the 95th percentile was 453 mm/month. Besides, the fifth and the 95th percentile of the extreme temperature distributions were 17.7°C/month and 27.1°C/month, respectively. The probability of climate extremes would be increased about +26% by 2050 based on business as usual. We found the future increase in temperature extremes in hot or cold conditions ($\bar{x}$ = +24%, SE = 0.001%) in the lowland areas. Moreover, drought and extreme precipitation events had considerably become more frequent ($\bar{x}$ = +0.9%, SE = 5%) by 2050 in the montane areas (**Figure 2B**).

The results show that forward and backward climatic velocities within the Leuser ecosystem are ranging from 0.08 m/year to 12.59 m/year ($\bar{x}$ = 1.25 m/year, SE = 0.002 m/year) and 0.08 m/year to 38.02 m/year ($\bar{x}$ = 8.98 m/year, SE = 0.015 m/year), respectively (**Figures 2C,D**). We found only 31.2% of the landscape areas that cannot survive climate change to maintain current climatic conditions in the future. The residence times from forward velocity were ranging from 7.9 to 1,548 years ($\bar{x}$ = 275.3 years, SE = 0.072 years).

Conservation Opportunities

We found that the local climate change exposure was ranging from 0.247–0.575 ($\bar{x}$ = 0.344, SE = 0.002), and from 0.249 to 0.467 ($\bar{x}$ = 0.334, SE = 0.001) within the protected areas and unprotected areas, respectively (**Figure 3A**). The results suggest that protected area is more susceptible to climate change than the unprotected areas within the Leuser Ecosystem based on SED values (Kolmogorov–Smirnov test; D = 0.20397, p-value < 0.005). Moreover, we found that the climatic anomaly in the nature reserve area (NRA) was ranging from 0.248 to 0.575 ($\bar{x}$ = 0.353, SE = 0.005). Dissimilarities between current and future climates within the Hunting Park (HP) was ranging from 0.316 to 0.436 ($\bar{x}$ = 0.344, SE = 0.003). Furthermore, the climatic anomaly within the Nature Conservation Area (NCA) was ranging from 0.258 to 0.281 ($\bar{x}$ = 0.269, SE = 0.001) (see **Figure 3B**).

The result shows that 32% (~840,490 km^2) of the Leuser Ecosystem would be threatened to sites and local populations for mammal species until 2050 (i.e., high-risk areas). Areas with threats to sites and endangered mammal populations are found in the south, northeast, and small area in the northwest regions (**Figure 4B**). Potential refugia areas with low threat cover only 28% of the Leuser Ecosystem. We found the potential refugia areas in the center of the landscape. Areas with the threats only to habitat (19%) will be in the center of the Leuser Ecosystem within the relatively low steepness. Besides, areas with the threats only to local populations (21%) will be also in the center of the Leuser Ecosystem but within relatively high steepness (**Figure 4B**).

All endangered mammal species within the Leuser Ecosystem will occur inside areas with a massive threat (i.e., habitat and local populations; see **Figure 4A** for the species richness range distribution) until 2050 from 3 to 100% of their ranges (see **Supplementary Material**, **Supplementary Table 1**). Moreover, forty-four species will maintain from 6 to 22% of their ranges in the areas with threats to sites, while a similar number of mammal species also will maintain from 16 to 41% of their ranges in the areas with threats to local populations until 2050. Furthermore,

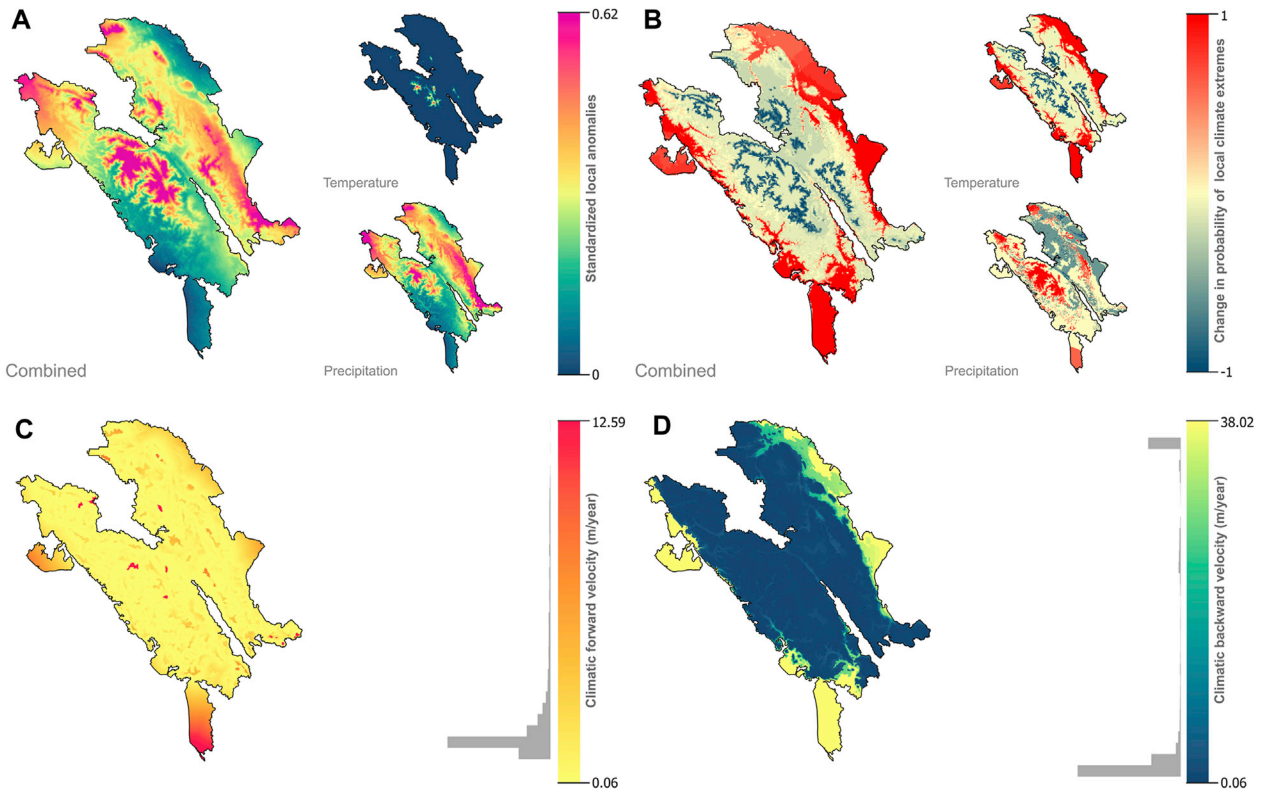

FIGURE 2 | Projected climate change in the Leuser Ecosystem based on climate change metrics. The maps show projections of change based on business-as-usual scenario (SSP585) by 2050. The metrics depicted characterize three dimensions of climate change: the magnitude of local changes in standardized climate anomalies **(A)**, change in the probability of extreme climates **(B)**, and climate change velocity, either forward **(C)** or backward **(D)** velocities. Gray-filled histograms indicate the statistical distributions of climate velocities.

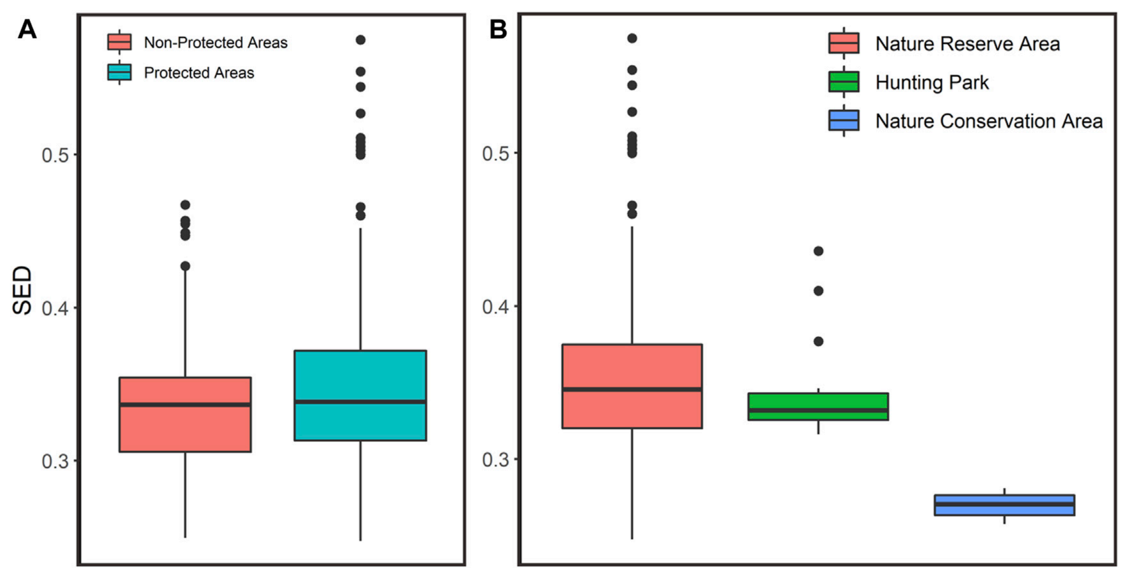

FIGURE 3 | Local climate exposure variation within protected and non-protected areas **(A)** and different protected area types **(B)** in the Leuser Ecosystem.

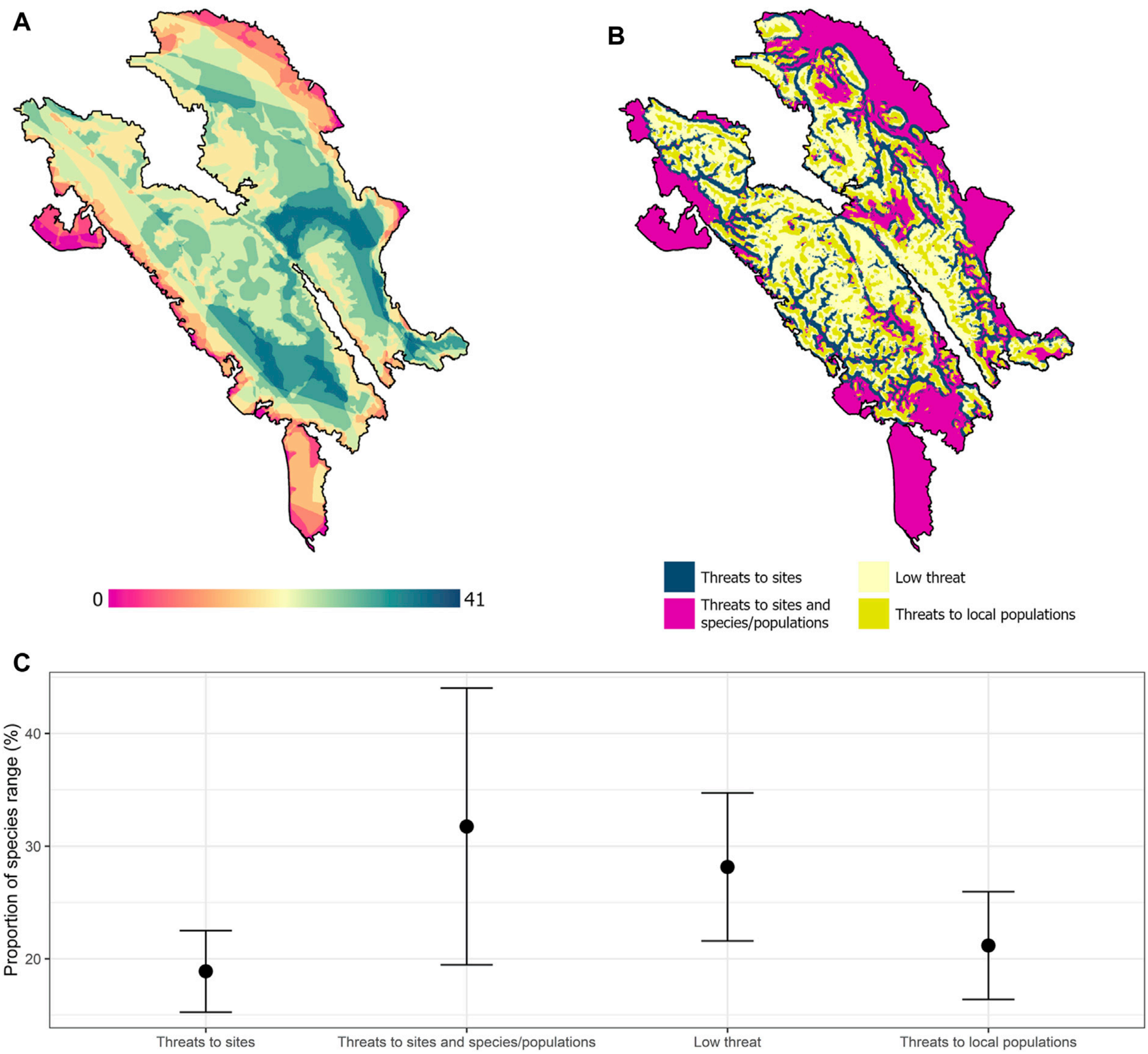

FIGURE 4 | Vulnerability assessment based on forward and backward velocities until 2050 **(A)**, current richness of mammal species in the Leuser Ecosystem **(B)**, mean and standard deviation in the proportion of 45 species ranges in each region within the different combinations of climatic velocity **(C)**.

areas with low threat and potentially used as refugia can maintain mammal species from 14 to 50% of their ranges (**Figure 4C**). We found that agile gibbon (*Hylobates agilis*) will be totally out of the refugia areas and almost 100% of species range occurred in the high-risk areas. Sumatran elephant (*Elephas maximus sumatranus*), bare-backed rousette (*Rousettus spinalatus*), and Sumatran orangutan (*Pongo abelii*) were also the most vulnerable as they broadly occur in the areas with threats to sites and local populations as well (ranging from 33 to 51% of their ranges).

DISCUSSION

In this study, we calculated the climate change metrics and elaborated on the same (i.e., forward and backward velocities) with mammal species range information within the protected areas of the Leuser Ecosystem for identifying threats to support conservation management. We found that the protected areas of the Leuser Ecosystem have become more susceptible to climate disturbance than the unprotected areas. The results show that future mammal species will likely migrate to the unprotected areas for niche conservatism as a consequence to the global climate change. Many anthropogenic activities that can lead to habitat destruction occur outside the protected areas (Sloan et al., 2018). A previous study shows that protected areas have to protect biodiversity effectively compared with the unprotected areas (Condro et al., 2021). However, according to another research, Indonesian protected areas do not seem to be an adequate option to prevent habitat disturbance (Brun et al., 2015). Thus, identification of future biodiversity threats based on climate change metrics is an infancy step to determine conservation management planning (Carroll et al., 2015).

In response to changing climate, the distributions of endangered mammals are shifting (Parmesan and Yohe, 2003). Many studies prove that global climate change is encouraging species occurrences poleward and approaching higher altitudes, where the temperature gradient induces upward shifts more likely than poleward shifts (Colwell et al., 2008). Our results indicate that the magnitude of climate change effects will be higher on the montane areas for both temperature and precipitation in the study area. Previous studies also showed that high altitude ecosystems are considered to be more threatened (Colwell et al., 2008; Loarie et al., 2009). However, we found that the local climate extremes will be greater in the lowland regions than montane areas that indicate a novel climate, which mostly appear in the lowland tropical areas such as rainforests (Williams et al., 2007). Moreover, in the lowland areas of the Leuser Ecosystem (e.g., peatlands, mangrove forests, and lowland rainforests), where climate velocities were high, adaptation strategies should be required (Ordonez et al., 2014). Possible conservation actions in this landscape are *in situ* management of species and site to increase resilience (Millar et al., 2007), *ex situ* conservation, habitat modification through engineering (Brook et al., 2008), and species translocation (Hughes et al., 2008). In contrast, in the high altitudinal areas (e.g., montane forests), where climate velocities (i.e., both forward and backward velocity) were relatively low, adaptation strategies should be facilitated along with protecting the remaining habitat and maintaining ecosystem functions (Colwell et al., 2008).

Our analyses showed that only 28% of the Leuser Ecosystem will preserve stable climate conditions, and can be performed as climatic refugia for mammal species until 2050. Moreover, most of the mammal species occurred in high-risk areas (i.e., threats to sites, local populations, and both of them). Mammal populations within the study area can be impacted in many ways: habitat disturbance can lead to species extirpation (Kinnaird et al., 2003; Dirzo et al., 2014), local changes in climate can induce species range contractions (Condro et al., 2021), behavior and physiology, as well as decline in the reproductive rate (Bronson, 2009), and dispersal ability (Schloss et al., 2012). We found that four species (*H. agilis*, *E. maximus*, *R. spinalus*, and *P. abelii*) will be most susceptible to climate change as they occur in the areas with threats to sites and local populations. A previous study showed that *P. abelii* would completely suffer range contraction by 2050, while *H. agilis* would likely expand their ranges to maintain climatic niches (Condro et al., 2021). In areas where local populations are threatened, species monitoring is required to determine current species status and assess whether further assisted conservation interventions such as species translocations to habitat refugia are required. In addition, habitat restoration and corridor development are needed to facilitate species dispersal and to increase connectivity of the species in areas with threats to sites (Hodgson et al., 2011; Borges and Loyola, 2020).

Predicting the long-term impacts of changing climate on biodiversity is quite challenging due to the complexity of species responses toward physiological and evolutionary mechanisms and the species interaction with anthropogenic activities that lead to range contractions. In particular, tropical species are susceptible to climate change as the species already exist near their maximum thermal tolerance (Araújo et al., 2013). Future climate projections were derived from an ensemble of three different general circulation models to capture the spatio-temporal representativeness of the study area. The spatial resolution of future climate models was increased by statistical downscaling to a local scale (Wiens et al., 2009). In many cases, statistical techniques make the climate model output more realistic than the other downscaling techniques (Ehret et al., 2012; Hawkins et al., 2013). The uncertainties of the climate models used in this study were provided by Navarro-Racines et al. (2020). Furthermore, the IUCN species range dataset can also be used to create the general conservation actions since there are no adequate species occurrence data for endangered mammal species in the Leuser Ecosystem. If the occurrence data are available, a species distribution or ecological niche modeling approach can be used to assess the impacts of climate change on species.

CONCLUSION

In conclusion, we highlighted that most of the mammal species in the Leuser Ecosystem would be impacted by changing climate. The use of climate change metrics can provide valuable information on the species exposure under a changing climate and a considerable scope to inform conservation actions. Standardized anomalies and changes in the probability of climate extremes are useful to inform the magnitude of climate change mean and variation within the study area. Moreover, climatic forward velocity can inform conservation of species and locally adapted populations, respectively, and along with backward velocity, can facilitate conservation of biodiversity at multi-levels in the face of climate change. These approaches can provide broad and generic suggestions for identifying areas that are most suitable for species as refugia while considering the multifaceted future threats of climate change to support biodiversity conservation planning. Further study also should incorporate the vegetation intactness dynamics into the model to capture adaptive capacity on biodiversity and resulting detailed conservation actions for specific areas and species (Watson et al., 2013; Alagador et al., 2016).

DATA AVAILABILITY STATEMENT

The raw data supporting the conclusion of this article will be made available by the authors, without undue reservation.

AUTHOR CONTRIBUTIONS

Conceptualization: LP, AC, SR, and IS; methodology: AC, LP, IS, and EI; formal analysis: EI and LP; data curation, LP, IS, SR, and EI; and writing—original draft preparation: AC and LP.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fenvs.2021.713837/full#supplementary-material.

REFERENCES

Alagador, D., Cerdeira, J. O., and Araújo, M. B. (2016). Climate change, species range shifts and dispersal corridors: an evaluation of spatial conservation models. *Methods Ecol. Evol.* 7, 853–866. doi:10.1111/2041-210X.12524

Albright, T. P., Pidgeon, A. M., Rittenhouse, C. D., Clayton, M. K., Flather, C. H., Culbert, P. D., et al. (2011). Heat waves measured with MODIS land surface temperature data predict changes in avian community structure. *Remote Sensing Environ.* 115, 245–254. doi:10.1016/j.rse.2010.08.024

Araújo, M. B., Alagador, D., Cabeza, M., Nogués-Bravo, D., and Thuiller, W. (2011). Climate change threatens European conservation areas. *Ecol. Lett.* 14, 484–492. doi:10.1111/j.1461-0248.2011.01610.x

Araújo, M. B., Ferri-Yáñez, F., Bozinovic, F., Marquet, P. A., Valladares, F., and Chown, S. L. (2013). Heat freezes niche evolution. *Ecol. Lett.* 16, 1206–1219. doi:10.1111/ele.12155

Booth, T. H., Nix, H. A., Busby, J. R., and Hutchinson, M. F. (2014). bioclim: the first species distribution modelling package, its early applications and relevance to most currentMaxEntstudies. *Divers. Distrib.* 20, 1–9. doi:10.1111/ddi.12144

Borges, F. J. A., and Loyola, R. (2020). Climate and land-use change refugia for Brazilian Cerrado birds. *Perspect. Ecol. Conservation* 18, 109–115. doi:10.1016/j.pecon.2020.04.002

Brito-Morales, I., García Molinos, J., Schoeman, D. S., Burrows, M. T., Poloczanska, E. S., Brown, C. J., et al. (2018). Climate Velocity Can Inform Conservation in a Warming World. *Trends Ecol. Evol.* 33, 441–457. doi:10.1016/j.tree.2018.03.009

Bronson, F. H. (2009). Climate change and seasonal reproduction in mammals. *Phil. Trans. R. Soc. B* 364, 3331–3340. doi:10.1098/rstb.2009.0140

Brook, B., Sodhi, N., and Bradshaw, C. (2008). Synergies among extinction drivers under global change. *Trends Ecol. Evol.* 23, 453–460. doi:10.1016/j.tree.2008.03.011

Brun, C., Cook, A. R., Lee, J. S. H., Wich, S. A., Koh, L. P., and Carrasco, L. R. (2015). Analysis of deforestation and protected area effectiveness in Indonesia: A comparison of Bayesian spatial models. *Glob. Environ. Change* 31, 285–295. doi:10.1016/j.gloenvcha.2015.02.004

Carroll, C., Lawler, J. J., Roberts, D. R., and Hamann, A. (2015). Biotic and climatic velocity identify contrasting areas of vulnerability to climate change. *PLoS One* 10, e0140486. doi:10.1371/journal.pone.0140486

Chen, I.-C., Hill, J. K., Ohlemüller, R., Roy, D. B., and Thomas, C. D. (2011). Rapid range shifts of species associated with high levels of climate warming. *Science* 333, 1024–1026. doi:10.1126/science.1206432

Cochard, R. (2017). "Scaling the Costs of Natural Ecosystem Degradation and Biodiversity Losses in Aceh Province, Sumatra," in *Redefining Diversity and Dynamics of Natural Resources Management in Asia* (Amsterdam, Netherlands: Elsevier), 231–271. doi:10.1016/B978-0-12-805454-3.00013-X

Colwell, R. K., Brehm, G., Cardelús, C. L., Gilman, A. C., Longino, J. T., Cardelus, C. L., et al. (2008). Global Warming, Elevational Range Shifts, and Lowland Biotic Attrition in the Wet Tropics. *Science* 322, 258–261. Available at:. doi:10.1126/science.1162547

Condro, A. A., Prasetyo, L. B., Rushayati, S. B., Santikayasa, I. P., and Iskandar, E. (2021). Predicting Hotspots and Prioritizing Protected Areas for Endangered Primate Species in Indonesia under Changing Climate. *Biology* 10, 154–177. doi:10.3390/biology10020154

Dirzo, R., Young, H. S., Galetti, M., Ceballos, G., Isaac, N. J. B., and Collen, B. (2014). Defaunation in the Anthropocene. *Science* 345, 401–406. doi:10.1126/science.1251817

Ehret, U., Zehe, E., Wulfmeyer, V., Warrach-Sagi, K., and Liebert, J. (2012). HESS Opinions "Should we apply bias correction to global and regional climate model data?". *Hydrol. Earth Syst. Sci.* 16, 3391–3404. doi:10.5194/hess-16-3391-2012

Fick, S. E., and Hijmans, R. J. (2017). WorldClim 2: new 1-km spatial resolution climate surfaces for global land areas. *Int. J. Climatol* 37, 4302–4315. doi:10.1002/joc.5086

Game, E. T., Lipsett-Moore, G., Saxon, E., Peterson, N., and Sheppard, S. (2011). Incorporating climate change adaptation into national conservation assessments. *Glob. Chang. Biol.* 17, 3150–3160. doi:10.1111/j.1365-2486.2011.02457.x

Garcia, R. A., Cabeza, M., Rahbek, C., and Araújo, M. B. (2014). Multiple dimensions of climate change and their implications for biodiversity. *Science* 344, 1247579. doi:10.1126/science.1247579

Gaveau, D. L. A., Wich, S., Epting, J., Juhn, D., Kanninen, M., and Leader-williams, N. (2009). The future of forests and orangutans (Pongo abelii) in Sumatra: predicting impacts of oil palm plantations, road construction, and mechanisms for reducing carbon emissions from deforestation. *Environ. Res. Lett.* 4, 034013. doi:10.1088/1748-9326/4/3/034013

Hajima, T., Watanabe, M., Yamamoto, A., Tatebe, H., Noguchi, M. A., Abe, M., et al. (2020). Development of the MIROC-ES2L Earth system model and the evaluation of biogeochemical processes and feedbacks. *Geosci. Model. Dev.* 13, 2197–2244. doi:10.5194/gmd-13-2197-2020

Hamann, A., Roberts, D. R., Barber, Q. E., Carroll, C., and Nielsen, S. E. (2015). Velocity of climate change algorithms for guiding conservation and management. *Glob. Change Biol.* 21, 997–1004. doi:10.1111/gcb.12736

Hawkins, E., Osborne, T. M., Ho, C. K., and Challinor, A. J. (2013). Calibration and bias correction of climate projections for crop modelling: An idealised case study over Europe. *Agric. For. Meteorology* 170, 19–31. doi:10.1016/j.agrformet.2012.04.007

Hijmans, R. J., Cameron, S. E., Parra, J. L., Jones, P. G., and Jarvis, A. (2005). Very high resolution interpolated climate surfaces for global land areas. *Int. J. Climatol.* 25, 1965–1978. doi:10.1002/joc.1276

Hodgson, J. A., Thomas, C. D., Cinderby, S., Cambridge, H., Evans, P., and Hill, J. K. (2011). Habitat re-creation strategies for promoting adaptation of species to climate change. *Conserv. Lett.* 4, 289–297. doi:10.1111/j.1755-263X.2011.00177.x

Hughes, L., Mcintyre, S., Lindenmayer, D. B., Parmesan, C., Possingham, H. P., and Thomas, C. D. (2008). U. Oceania (esp. to its colonization). *Science* 321, 345–346. doi:10.1515/9783110971231.345

Janssen, M. (2003). Economic valuation of the Leuser National Park on Sumatra, Indonesia. *Ecol. Econ.* 44, 43–62. doi:10.1016/S0921-8009(02)00224-0

Katz, R. W., Brush, G. S., and Parlange, M. B. (2005). Statistics of extremes: Modeling ecological disturbances. *Ecology* 86, 1124–1134. doi:10.1890/04-0606

Kinnaird, M. F., Sanderson, E. W., O'Brien, T. G., Wibisono, H. T., and Woolmer, G. (2003). Deforestation trends in a tropical landscape and implications for endangered large mammals. *Conservation Biol.* 17, 245–257. doi:10.1046/j.1523-1739.2003.02040.x

Le Saout, S., Hoffmann, M., Shi, Y., Hughes, A., Bernard, C., Brooks, T. M., et al. (2013). Protected areas and effective biodiversity conservation. *Science* 342, 803–805. doi:10.1126/science.1239268

Linkie, M., Haidir, I. A., Nugroho, A., and Dinata, Y. (2008). Conserving tigers *Panthera tigris* in selectively logged Sumatran forests. *Biol. Conservation* 141, 2410–2415. doi:10.1016/j.biocon.2008.07.002

Loarie, S. R., Duffy, P. B., Hamilton, H., Asner, G. P., Field, C. B., and Ackerly, D. D. (2009). The velocity of climate change. *Nature* 462, 1052–1055. doi:10.1038/nature08649

Millar, C. I., Stephenson, N. L., and Stephens, S. L. (2007). Climate change and forests of the future: managing in the face of uncertainty. *Ecol. Appl.* 17, 2145–2151. doi:10.1890/06-1715.1

Navarro-Racines, C., Tarapues, J., Thornton, P., Jarvis, A., and Ramirez-Villegas, J. (2020). High-resolution and bias-corrected CMIP5 projections for climate change impact assessments. *Sci. Data* 7, 1–14. doi:10.1038/s41597-019-0343-8

Olson, D. M., Dinerstein, E., Wikramanayake, E. D., Burgess, N. D., Powell, G. V. N., Underwood, E. C., et al. (2001). Terrestrial ecoregions of the world: A new map of life on Earth. *Bioscience* 51, 933–938. doi:10.1641/0006-3568(2001)051[0933:teotwa]2.0.co;2

Ordonez, A., Martinuzzi, S., Radeloff, V. C., and Williams, J. W. (2014). Combined speeds of climate and land-use change of the conterminous US until 2050. *Nat. Clim Change* 4, 811–816. doi:10.1038/nclimate2337

Page, S. E., Rieley, J. O., and Banks, C. J. (2011). Global and regional importance of the tropical peatland carbon pool. *Glob. Chang. Biol.* 17, 798–818. doi:10.1111/j.1365-2486.2010.02279.x

Parmesan, C., and Yohe, G. (2003). A globally coherent fingerprint of climate change impacts across natural systems. *Nature* 421, 37–42. doi:10.1038/nature01286

Riahi, K., van Vuuren, D. P., Kriegler, E., Edmonds, J., O'Neill, B. C., Fujimori, S., et al. (2017). The Shared Socioeconomic Pathways and their energy, land use, and greenhouse gas emissions implications: An overview. *Glob. Environ. Change* 42, 153–168. doi:10.1016/j.gloenvcha.2016.05.009

Ribeiro, B. R., Sales, L. P., De Marco, P., and Loyola, R. (2016). Assessing mammal exposure to climate change in the Brazilian Amazon. *PLoS One* 11, e0165073–13. doi:10.1371/journal.pone.0165073

Sales, L. P., Ribeiro, B. R., Pires, M. M., Chapman, C. A., and Loyola, R. (2019). Recalculating route: dispersal constraints will drive the redistribution of Amazon primates in the Anthropocene. *Ecography* 42, 1789–1801. doi:10.1111/ecog.04499

Schloss, C. A., Nuñez, T. A., and Lawler, J. J. (2012). Dispersal will limit ability of mammals to track climate change in the Western Hemisphere. *Proc. Natl. Acad. Sci.* 109, 8606–8611. doi:10.1073/pnas.1116791109

Sloan, S., Campbell, M. J., Alamgir, M., Collier-Baker, E., Nowak, M. G., Usher, G., et al. (2018). Infrastructure development and contested forest governance threaten the Leuser Ecosystem, Indonesia. *Land use policy* 77, 298–309. doi:10.1016/j.landusepol.2018.05.043

Stewart, S. B., Elith, J., Fedrigo, M., Kasel, S., Roxburgh, S. H., Bennett, L. T., et al. (2021). Climate extreme variables generated using monthly time-series data improve predicted distributions of plant species. *Ecography* 44, 626–639. doi:10.1111/ecog.05253

Suggitt, A. J., Gillingham, P. K., Hill, J. K., Huntley, B., Kunin, W. E., Roy, D. B., et al. (2011). Habitat microclimates drive fine-scale variation in extreme temperatures. *Oikos* 120, 1–8. doi:10.1111/j.1600-0706.2010.18270.x

Tatebe, H., Ogura, T., Nitta, T., Komuro, Y., Ogochi, K., Takemura, T., et al. (2019). Description and basic evaluation of simulated mean state, internal variability, and climate sensitivity in MIROC6. *Geosci. Model. Dev.* 12, 2727–2765. doi:10.5194/gmd-12-2727-2019

Warren, M., Hergoualc'h, K., Kauffman, J. B., Murdiyarso, D., and Kolka, R. (2017). An appraisal of Indonesia's immense peat carbon stock using national peatland maps: uncertainties and potential losses from conversion. *Carbon Balance Manage* 12, 12. doi:10.1186/s13021-017-0080-2

Watson, J. E. M., Iwamura, T., and Butt, N. (2013). Mapping vulnerability and conservation adaptation strategies under climate change. *Nat. Clim Change* 3, 989–994. doi:10.1038/nclimate2007

Weiskopf, S. R., McCarthy, J. L., McCarthy, K. P., Shiklomanov, A. N., Wibisono, H. T., and Pusparini, W. (2019). The conservation value of forest fragments in the increasingly agrarian landscape of Sumatra. *Envir. Conserv.* 46, 340–346. doi:10.1017/S0376892919000195

Wich, S. A., Singleton, I., Nowak, M. G., Utami Atmoko, S. S., Nisam, G., Arif, S. M., et al. (2016). Land-cover changes predict steep declines for the Sumatran orangutan (Pongo abelii). *Sci. Adv.* 2, e1500789–9. doi:10.1126/sciadv.1500789

Wiens, J. A., Stralberg, D., Jongsomjit, D., Howell, C. A., and Snyder, M. A. (2009). Niches, models, and climate change: Assessing the assumptions and uncertainties. *Proc. Natl. Acad. Sci.* 106, 19729–19736. doi:10.1073/pnas.0901639106

Williams, J. W., Jackson, S. T., and Kutzbach, J. E. (2007). Projected distributions of novel and disappearing climates by 2100 AD. *Proc. Natl. Acad. Sci.* 104, 5738–5742. doi:10.1073/pnas.0606292104

Yukimoto, S., Kawai, H., Koshiro, T., Oshima, N., Yoshida, K., Urakawa, S., et al. (2019). The meteorological research institute Earth system model version 2.0, MRI-ESM2.0: Description and basic evaluation of the physical component. *J. Meteorol. Soc. Jpn.* 97, 931–965. doi:10.2151/jmsj.2019-051

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Using Geographic Information Systems and the Analytical Hierarchy Process for Delineating Erosion-Induced Land Degradation in the Middle Citarum Sub-Watershed, Indonesia

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Land degradation (LD) is an important issue worldwide because it affects food production and people's welfare. Many factors cause land degradation, but in humid tropical areas, erosion is the main factor. More than 100 countries including Indonesia are affected by LD. Watershed management should be prioritized owing to budget constraints, while on the other side, the area affected by LD is very large compared to the size of the existing land area. The middle Citarum sub-watershed (MCSW) is one of the most degraded drylands in Indonesia, where the environment can be considered a typical humid tropical watershed. The objective of this study was to map degraded lands and prioritize restoration using a combined approach of the universal soil loss equation (USLE), the analytical hierarchy process (AHP) and geographic information systems (GIS) in a multiple-criteria decision analysis (MCDA) environment. The severity of LD was estimated quantitatively by analyzing the parameters of land use and land cover, slope, soil erosion, productivity, and management. The results indicated that the MCSW is dominated by the potentially degraded land classes (38%), followed by the degraded land classes (21%). The prioritization of LD restoration is suggested in the area of very high and high degraded land. The method developed in this research work could be adopted as a tool to guide decision-makers toward sustainable land resource management in humid tropical watersheds affected by LD.

Keywords: erosion, LULC, MCDA, prioritization, USLE

1 INTRODUCTION

Land degradation (LD) is a dynamic environmental process that can reduce ecosystem functions and disrupts agricultural production. The implications of LD include environmental setbacks and reduced food security as well as diminished sustainable economy (Zhao et al., 2013). LD refers to the reduced or lost biological and economic productivity of agricultural land or forests resulting from human activities (Expert Group on SDG Indicators [IAEG-SDGs], 2016). In the past two decades, LD has affected more than 20% of the vegetated land area and affects more than 1.5 billion people worldwide (United Nations Convention to Combat Desertification [UNCCD], 2017). The global issue of LD has become the target of the Sustainable Development Goals (SDGs).

LD can be caused by physical, chemical, and biological factors of the soil (Brevik et al., 2015), as well as human factors (Khaledian et al., 2017). It has been estimated that around 30% of the world's land is degraded (Lal, 2015; Nkonya et al., 2016). Bai et al. (2008) have estimated that the area of the degraded land worldwide is 18.1 m km^2, 92% of which due to land mismanagement. In the past 45 years, LD has caused world rice yields to decline by an estimated 1.6–2.7% and financial losses to exceed US$ 10.6 billion (Chen et al., 2012). The depletion of natural resources due to LD can occur in dry and subhumid climates alike (Omuto et al., 2014).

Many natural resource processes can cause LD including erosion, deforestation, soil compaction, soil acidity, salting, and desertification (Turner et al., 2016). Among the various forms of LD, water erosion represents the most important form (Hermassi and Amami, 2018). In Indonesia, the main cause of LD is erosion owing to high rainfall intensity. Erosion-induced LD is an environmental problem that occurs worldwide, causing land productivity to decline (García–Ruiz et al., 2013). Indonesia's land area covers 191.19 m ha, which is dominated by 144.47 m ha (75.60%) of dry land; hence, erosion-induced LD can affect a large area. Indeed, dry land is very vulnerable to erosion and degradation. The area of degraded land in Indonesia in 2018 was estimated at 14 million ha (Indonesian Statistics, 2020), and its restoration will be expensive. Thus, it is necessary to prioritize the handling of land affected by LD. As targeted in the SDGs, degraded land needs to be restored. Funding for watershed restoration is limited, and so it is important to define priority areas in order to optimize the results.

Geographic information systems (GISs) can be used throughout the watershed restoration process. Several studies related to the prioritization of watersheds for conservation actions have been carried out, specifically on soil erosion (Singh et al., 2019; Choudhary et al., 2020), watershed morphometry (Mohammed et al., 2018; Tukura et al., 2021), and the sediment yield index (Jang et al., 2013). Furthermore, Ahmad and Pandey (2018) have used the GIS to map LD. Thus, the GIS can be used to map complex and spatial LD (AbdelRahman, 2014).

Multiple-criteria decision analysis (MCDA) integrates GIS models with decision-making processes and is widely used in complex natural resource and environmental management, including watershed management (Halefom et al., 2019). Many MCDA methods are available, one of which uses the analytical hierarchy process (AHP) (Saaty, 2008) to carry out weighting before being integrated with GIS (Hassan et al., 2015; Widiatmaka et al., 2016). MCDA integrating the AHP with GIS can be applied in the study of LD because the model is very flexible, enabling possible decisions to be made based on both quantitative and qualitative data (Weerakoon, 2014). Many researchers have used MCDA approaches for LD mapping. AbdelRahman et al. (2019) considered soil, slope, rainfall, DEM, land use, and spatial land characteristics. Gessesse et al. (2015) studied LD using the land use/land cover (LULC) dynamics model, while in Iraq, Shareef et al. (2020) combined LULC change with the GIS and AHP model in order to consider LULC parameters such as urban area and vegetation.

Many models have been used to estimate soil loss due to water erosion, one of which is the universal soil loss equation (USLE) (Wischmeier and Smith, 1978). This is one of the most relevant methods for assessing soil loss (Alewell et al., 2019). Several researchers (Kefi et al., 2009; López–García et al., 2020) have integrated the USLE and GIS in their studies.

Citarum watershed is a degraded watershed in Indonesia. It plays an important role in the economy and the lives of the inhabitants of the provinces of West Java, for instance, for water supply for irrigation, industry, and energy supply through electric power generation. The Citarum watershed is one of the most eroded watersheds in Indonesia. Therefore, its conservation needs to be prioritized.

The objective of this research was to estimate the LD caused by soil erosion by applying the USLE along with the GIS and the AHP in MCDA approaches, in order that this degraded land area's restoration can be prioritized. The MCDA considered five parameters: LULC, soil erosion, slope, management, and productivity. One part of the Citarum watershed, namely, the middle Citarum sub-watershed (MCWS), was chosen as the study area, owing it to having the characteristics of natural resources in a humid tropical environment. Referring to the extent of a degraded land area in Indonesia, this method for prioritization needs to be developed in typical areas so that it can be extended to other areas.

2 MATERIALS AND METHODS

2.1 Location Description

The MCSW has an extension of 71,091.66 ha or 21.94% of the Citarum watershed. The MCSW is located in Purwakarta Regency, Indonesia. Geographically, the MCSW area extends between 107°13′12.4″–107°35′2.7″ east and 6°26′49.9″–6°46′36.7″ south. The MCSW has a tropical climate with a temperature range of 20–30°C and receives 1,000–4,000 mm of rainfall per year. Some soil types that make up the MCSW are Dystrudepts, Eutrudepts, Hapludalfs, and Hapludands. The MCSW is dominated by a hilly morphology with elevations 200–800 m above sea level and slopes ranging from 3 to 40%.

2.2 Data Input

This study used a topographic map at a scale of 1:25,000 (Indonesian Geospatial Information Agency [IGIA], 2015), monthly rainfall data for 2010–2019 (Meteorological, Climatological, and Geophysical Agency, 2020), a soil map at a scale of 1:50,000 (Indonesian Center for Agricultural Land Resources Research and Development, 2017), a map of the official regional spatial plan (ORSP) of Purwakarta Regency (Development Planning Agency, 2012), and SPOT 6 imagery of

May 26, 2019. Maps and image interpretation results were processed with GIS to obtain thematic maps in order to make soil erosion maps, LD maps, and a priority watershed restoration map. Mostly, the data type used in this study is in a vector format.

2.3 Methods

This research was divided into four stages: calculating the weight of each criterion of LD using the AHP model, calculating soil erosion using the USLE model, preparing thematic maps for mapping LD, and making a map of LD and the prioritization of its handling. The research area was divided into three areas with different LULC: protection forests and conservation forests (PFCFs), production forests, and areas protected outside forest areas (PFPO), and cultivation, agriculture/plantation, and other use areas (CAPO). The differentiation between the three areas was based on the fact that the distinctive LULC and area statuses implied differences in the amount of erosion as well as the conservation measures needed. The ORSP map was used to divide the three areas, covering 5.34, 65.58, and 28.97% for PFCF, PFPO, and CAPO, respectively.

2.3.1 Calculating Weight Using the Analytical Hierarchy Process

The LD map was developed based on the parameters: LULC, slope, soil erosion, productivity, and management. The AHP was applied to weight these parameters, via three steps: creating a hierarchy of problem structures, priority analysis by comparing each parameter in pairs to obtain weights, and verification by calculating the consistency in pairwise comparisons. Five experts in the field of land degradation and watershed management were involved in determining the weights. The consistency of expert opinions was verified by determining the consistency ratio value (Saaty, 2008). The weights were given to describe the extent to which each criterion contributed to LD. Five experts were involved in weight estimation to create a pairwise comparison matrix. A scale of 1–9 was used to identify the importance of the LD parameters: if two criteria were equally important, it was given a weight of 1, while a weight of 9 indicated that one criterion was more important than the others. The sub-criteria scoring assessment was based on the weight of each sub-criterion determined by the experts.

2.3.2 Prediction of Erosion Using Universal Soil Loss Equation

Erosion was predicted using the USLE model and was applied spatially using ArcGIS to integrate the parameters of **Eq. 1**. According to the Indonesian government standard document (Indonesian Ministry of Forestry [IMF], 2013), soil erosion loss (A) is divided into five levels: very severe ($A > 480\,\mathrm{t\,ha^{-1}y^{-1}}$), severe ($180 \geq A \leq 480\,\mathrm{t\,ha^{-1}y^{-1}}$), moderate ($60 \geq A < 180\,\mathrm{t\,ha^{-1}y^{-1}}$), slight ($15 \geq A < 60\,\mathrm{t\,ha^{-1}y^{-1}}$), and very slight ($<15\,\mathrm{t\,ha^{-1}y^{-1}}$).

$$A = R\,x\,K\,x\,LS\,x\,C\,x\,P \tag{1}$$

where A is the average annual soil loss per unit area ($\mathrm{t\,ha^{-1}\,y^{-1}}$), R is the rainfall erosivity factor ($\mathrm{MJ\,mm\,ha^{-1}\,h^{-1}\,y^{-1}}$), K is the soil erodibility factor ($\mathrm{t\,h\,MJ^{-1}\,mm^{-1}}$), LS is the slope length and slope steepness factor, C is the cover and management factor, and P is the support and conservation practice factor.

Rainfall Erosivity Factor

The annual rainfall values of 2010–2019 were used to calculate the R factor based on monthly rainfall using **Eq. 2** (Li et al., 2014). The R factor reflects the impact of raindrops on the ground (Wischmeier and Smith, 1978).

$$R = \sum_{i=1}^{12} -1.15527 + 1.792P_i, \tag{2}$$

where P_i is the monthly precipitation.

Soil Erodibility Factor

The K factor reflects the rate of soil loss per rainfall erosivity index. The K factor varies from 0 to 1; a low K value indicates less susceptibility to soil erosion. K values were calculated using **Eq. 3** (Hammer, 1978).

$$K = \frac{\{2.713\left(M^{1.14}\right)\left(10^{-4}\right)(12-a) + 3.25\,(b-2) + 2.5\,(c-3)\}}{100}, \tag{3}$$

where M is (very fine sand % + silt %) x (100–clay %), a is the organic matter content (%), b is the soil structure code, and c is the soil permeability code. Analysis of the soil texture and the soil organic content was carried out in the laboratory of the Department of Soil Science, IPB University, Indonesia.

Slope Length and Slope Steepness

LS is the ratio of soil loss on a standard slope with a length of 72.6 feet and a slope of 9%. The values of length (L) and slope (S) can be gathered from the table in Wischmeier and Smith (1978).

Cover and Management Factor

This factor reflects the effect of planting and tillage on soil erosion rates. It ranges from 1 to 0. A value equal to 1 indicates no land cover and is treated as barren land, whereas a value close to 0 indicates a very strong cover effect, the soil being well-protected from erosion. The value of C in this study was obtained from the C index table (Wischmeier and Smith, 1978) and applied to the land use map.

Conservation Practices Factor

This factor is a reflection of the application of soil conservation practices such as planting in strips and terracing. In this study, the p value was obtained from the land cover and the P table (Wischmeier and Smith, 1978) prevailing in Indonesia.

2.3.3 Preparation Input Map for the Land Degradation Map

LULC

The LULC map was obtained from a visual interpretation by on-screen digitizing using ArcGIS 10.6.1 of SPOT 6 satellite imagery. The LULC is classified into 10 classes based on the Indonesian

TABLE 1 | Calculating Criteria Weights with Consistency Ratio in the MCSW.

Protection forest and conservation forests (PFCF)					
	LULC	Slope	Erosion	Management	Weight
LULC	1	3	3	5	0.519
Slope	1/3	1	1	3	0.201
Erosion	1/3	1	1	3	0.201
Management	1/5	1/3	1/3	1	0.079
Production forests and protected areas outside forests areas (PFPO)					
LULC	1	5	5	2	0.502
Slope	1/5	1	1	4	0.090
Erosion	1/5	1	1	4	0.090
Management	1/2	1/4	1/4	1	0.319
Cultivation, agriculture/plantation, and other use areas (CAPO)					
	Productivity	Slope	Erosion	Management	Weight
Productivity	1	3	2	1	0.351
Slope	1/3	1	1/2	1/3	0.109
Erosion	1/2	2	1	1/2	0.189
Management	1	3	2	1	0.351

Consistency ratio analysis	**PFCF**	**PFPO**	**CAPO**
Λmax	4.05873	4.041678	4.012181
CI	0.019577	0.013893	0.00406
RI	0.9	0.9	0.9
CR	0.022	0.015	0.005

National Standard (SNI) for LULC, specifically SNI No. 7645.1: 2014. The field check has been done on July 2020 for validation. The overall accuracy and a KAPPA analysis were 89.00 and 86.3%, respectively. The 10 classes are forests, shrubs, gardens/moors, plantation, seasonal plants, open land, water bodies, rice fields, settlements, and non-settlement buildings. In this study, LULC was reclassified into five LD classes: very good, good, moderate, bad, and very bad.

Slope

Slope classes were created from the results of the contour layer processing on the topographic map using ArcGIS 10.6.1. The slope class criteria were based on the guidelines P.4/V-Set/2013 (Indonesian Ministry of Forestry [IMF], 2013).

Productivity

Productivity is an important factor when assessing degraded land in agricultural areas. It was calculated based on comparing the productivity achieved by rice and maize with the average productivity under traditional management. The productivity classifications for LD were divided into very high, high, moderate, low, and very low (Indonesian Ministry of Forestry [IMF], 2013).

Management

Management was used to assess LD based on the completeness of parameters: management aspects, the existence of area boundaries, security and supervision, and whether or not counseling is implemented. The management criteria in determining LD were divided into good, moderate, and bad.

2.3.4 Mapping Land Degradation and Prioritizing Restoration

LD was analyzed using five thematic map inputs. Each criterion was reclassified based on the score of each sub-criterion. An overlay was carried out using ArcGIS software on the reclassified thematic map, multiplied by the respective criterion weights generated from the AHP. Subsequently, the LD map was reclassified into five classes: very highly degraded, highly degraded, degraded, potentially degraded, and non-degraded. The prioritization of watershed management is very important in natural resource management, especially in the context of integrated watershed development and limited watershed restoration funding. Planners and decision makers should therefore pay attention to restoration prioritization. In this study, prioritization restoration was carried out by assigning a rating to the level of LD, divided into three classes: high priority, moderate priority, and low priority.

3 RESULTS AND DISCUSSION

3.1 Weight of Criteria

The pairwise comparison and weight values of the criteria obtained from the AHP are presented in **Table 1**. According to the judgments of the five experts, the highest weight values were given to the LULC in the PFCF and PFPO areas, with weights of 0.519 and 0.502, respectively. These high weight values can be explained by the fact that in these two forest areas, changes in the LULC easily cause land to become degraded. Such findings are in line with the results of Narendra et al.'s (2019) research on

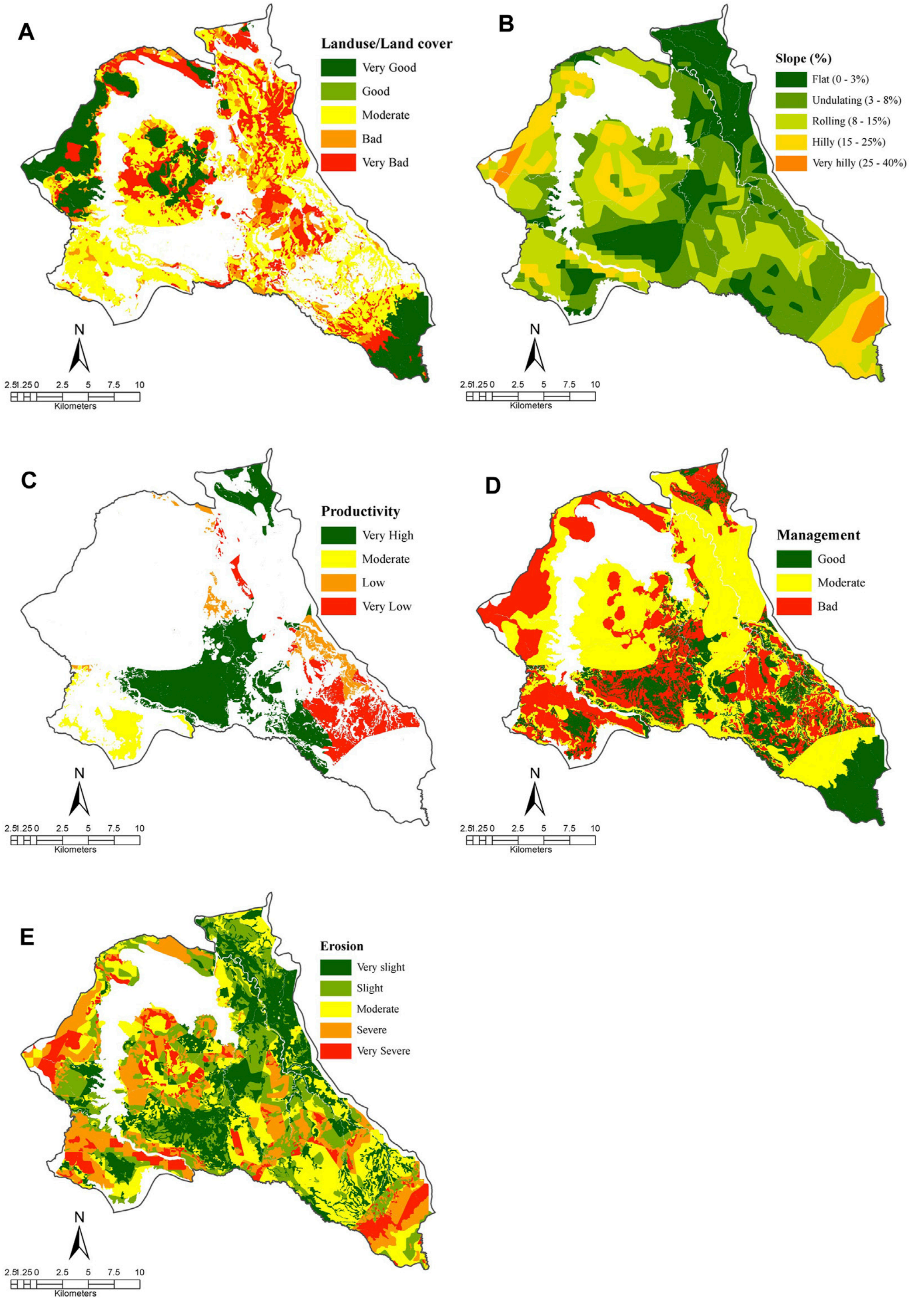

FIGURE 1 | Thematic maps for mapping LD.

the island of Lombok, Indonesia, which has similar environmental conditions. In CAPO areas, the highest weights were given to management and productivity, which can be explained by the fact that if mismanagement occurs, the result is decreased land quality and thereby degraded land. Management and productivity are thus important factors in agricultural areas, as also insisted by several researchers (Abera et al., 2020; Han et al., 2020).

TABLE 2 | Selected criteria and sub-criteria involved in the MCSW degraded land classification.

No.	Criteria	Sub-criteria	Scoring	Area	
				Ha	%
1	LULC (PFCF and PFPO)	Very good	5	8,967.86	21.56
		Good	4	15.43	0.04
		Moderate	3	18,775.45	45.14
		Bad	2	3,393.35	8.16
		Very bad	1	10,437.34	25.10
2	Productivity (CAPO)	Very high (>Rp 43 million/ha)	5	3,613.48	21.31
		High (Rp 42–43 million/ha)	4	1,563.73	9.22
		Moderate (Rp 40–41 million/ha)	3	1,660.97	9.79
		Low (Rp 38–39 million/ha)	2	—	—
		Very low (<38 million/ha)	1	10,120.34	59.68
3	Slope	Flat (0–3%)	5	12,362.19	21.11
		Undulating (3–8%)	5	21,157.41	36.14
		Rolling (8–15%)	4	16,480.14	28.15
		Hilly (15–25%)	3	7,310.80	12.49
		Very hilly (25–40%)	2	1,237.41	2.11
4	Erosion	Very slight	5	14,282.48	24.39
		Slight	4	13,096.81	22.37
		Moderate	3	14,974.56	25.58
		Severe	2	11,000.30	18.79
		Very severe	1	5,193.80	8.87
5	Management	Good	5	12,222.10	20.88
		Moderate	3	27,636.09	47.20
		Bad	1	18,689.76	31.92

AHP is a qualitative approach, in which the primary weakness is subjective and unable to incorporate uncertainty (Hong et al., 2019). The AHP is highly dependent on the selection of appropriate experts to determine the priority scale in pairwise comparisons because it will cause biased opinions among experts (Chen et al., 2021). In this study, anticipation has been done by involving five competent experts in the field of land degradation. This was proved by the consistency of the experts involved in determining the weight, which indicates that this study's results were very good. Indeed, the consistency ratio was below 10%, as displayed in **Table 1**.

3.2 Soil Erosion Predicted

The soil erosion predicted using the USLE model in the MCSW, ranging from 0.87 to 495.30 t ha^{-1} y^{-1}, is presented in **Figure 1E** and **Table 2**. The results indicated that soil erosion has a wider range than the soil erosion in Indonesia presented by Adimihardja (2008), which ranges from 35 to 220 t ha^{-1} y^{-1}. However, the results showed the same trend as recognized by Taslim et al. (2019) in 15 watersheds in Indonesia, where the erosion rate was found to range from 0 to 564 t ha^{-1} y^{-1}. The large range found in our study suggests high diversity of natural resources, especially LULC. This range of soil erosion was in the humid tropics, as reported by Labrière et al. (2015).

The results indicated that the MCSW was dominated by the light erosion class (15.20–65 t ha^{-1} y^{-1}), which accounted for 25% and was found in PFPO. The very heavy soil erosion class (>495.30 t ha^{-1} y^{-1}) was found in 7.30% of the study area and was mainly in the PFPO region. PFPO represents the largest area, occupying 65.58% of the study area. The PFPO region is mainly covered by forests, plantations, non-cultivated vegetation, and shrubs. Based on the USLE factor, the PFPO area is characterized by an *LS* factor of 1.2, a *C* factor of 0.05–0.6, and a *K* factor ranging from 0.18 to 0.21. This area is located on an 8–25% slope. The soil textures were found to be soft and medium. These physical conditions, coupled with human factors, cause this area to face all classes of soil erosion, from very severe to slight, albeit dominated by the former. As has been shown by Vahabi and Nikkami (2008), different soil characteristics influence soil erosion.

The CAPO area was found to be dominated by rice fields and settlements, located on a 0–8% slope. The CAPO area is mostly composed of metamorphic rocks with the main soil types Eutrudepts and Dystrudepts. The soil texture ranges from soft to moderately soft, while soil drainage ranges from good to poor, and soil depth from deep to very deep. This area has a *C* value of 0.05–0.75, a *K* factor ranging from 0.06 to 0.32, and an *LS* factor of 0.25. Consequently, erosion in the CAPO area tends to be quite slight. However, in this environment, which is strongly influenced by human activity, erosion can vary (Dotterweich, 2013).

The PFCF area is located on steep slopes (15–40%) with a coarse to medium-coarse soil texture, good to fast soil drainage, and shallow to deep soil depth. This area is dominated by the protected forest and conservation forest. While the USLE factor in this area is the *K* value ranging from 0.14 to 0.32, the *LS* factor was found to be 1–4.25, and the *C* factor 0.03–0.75. The main soil erosion class in this area was found to be slight. However, the very severe erosion class was found in locations bordering the PFPO area due to land clearing activities, which disturb the forest area and cause soil erosion.

3.3 Thematic Inputs Map and Degraded Land Map

The impacts of LULC, erosion, slope, productivity, and management on LD are presented in **Figure 1** and **Table 2**.

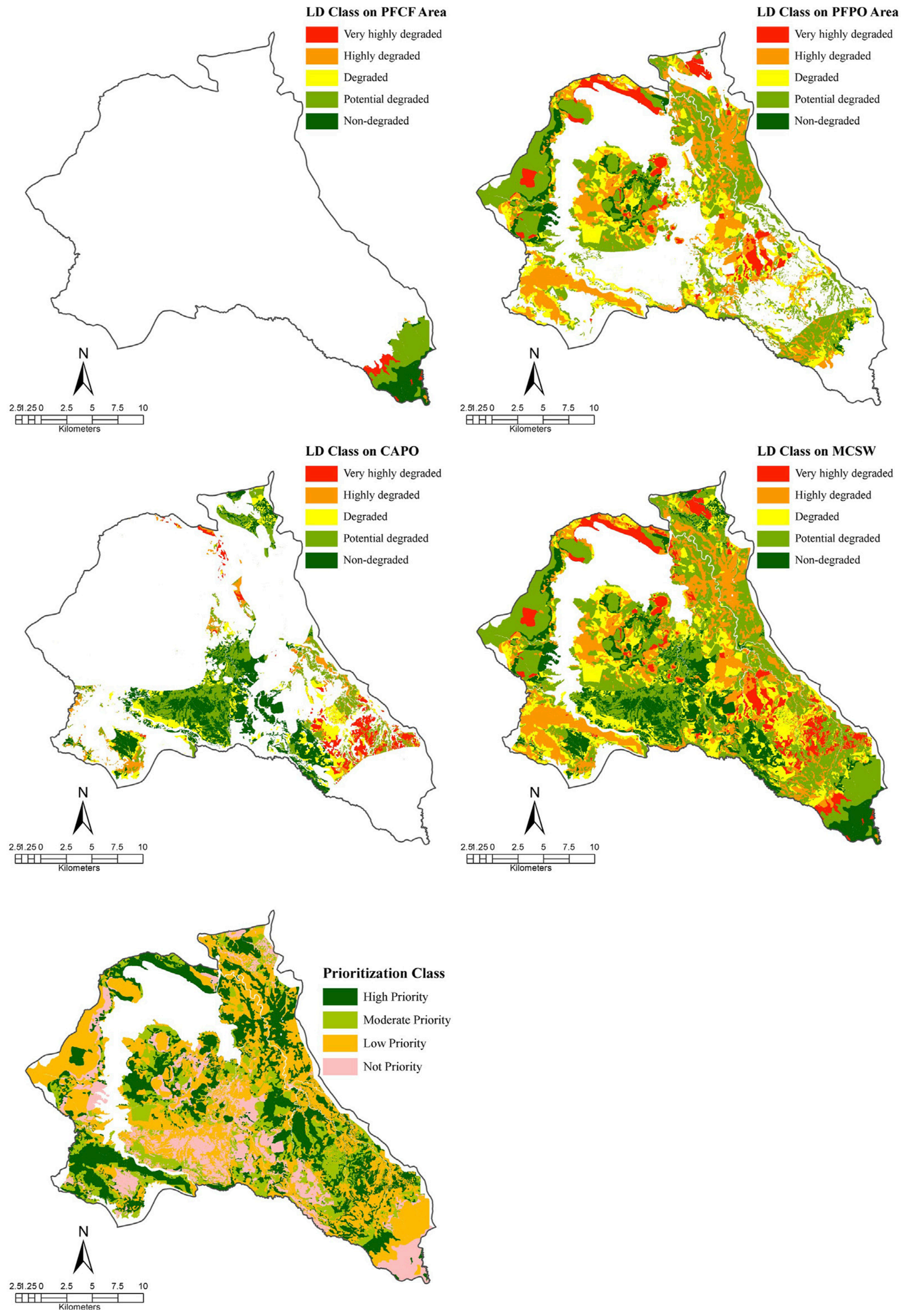

FIGURE 2 | LD class map and prioritization map.

The LULC in the MCSW was found to be dominated by plantations (24.23%) and paddy fields (24.62%). The results of the reclassification based on the LULC indicated that MCSW were dominated by the medium-degraded land classes (45%) (**Figure 1A**; **Table 2**). The impact of slopes on LD is presented in **Figure 1B** and **Table 2**. The result of the reclassification of slope

TABLE 3 | LD classes and prioritization classes for handling watershed restoration.

LD class	PFCF (ha)	PFPO (ha)	CAPO (ha)	Total area	
				Ha	%
Very highly degraded	306.91	2,482.81	1,486.08	4,275.80	7.30
Highly degraded	28.99	10,844.25	1,388.98	12,262.22	20.94
Degraded	1.44	7,463.01	3,039.47	10,503.92	17.94
Potentially degraded	1,630.96	15,593.76	4,999.78	22,224.50	37.96
Non-degraded	1,160.12	2,077.18	6,044.21	9,281.51	15.85
Total	3,128.42	38,461.01	16,958.52	58,547.95	100.00
Prioritization class					
High priority				16,538.02	28.24
Medium priority				10,503.92	17.94
Low priority				22,224.50	37.96
No priority				9,281.51	15.85

classes indicated that the MCSW is dominated by potentially degraded classes (36.14%) and slightly degraded class (28.15%) land classes. The impact of productivity on LD is presented in **Figure 1C** and **Table 2**. The productivity factor was one of the criteria used to assess LD in the CAPO areas. The reclassification of the productivity factor revealed that 59.68% of the MCSW has low productivity, meaning that it is highly degraded. The impact of management on LD is presented in **Figure 1D** and **Table 2**. The results of the management factor reclassification showed that 47.20% (CAPO areas) has moderate management factors, followed by 31.92% (PFPO areas) with poor management, rendering these places prone to LD.

3.4 Land Degradation Prioritization

In this article, certain criteria have been specified to study the degree of degradation and then create a map of prioritizing for watershed restoration. The distribution of LD in the MCSW is presented in **Figure 2** and **Table 3**. The results show that the area of degraded land in the MCSW is 29,911.95 ha (48.70%), comprising highly degraded (6.96%), degraded (24.64%), and medium-degraded classes (17.10%). The area of the non-degraded class is 31,506 ha (51.30%), divided into potentially degraded (36.19%) and non-degraded (15.11%). The degradation class map demonstrates that the area with very high LD is distributed around the forest PFPO areas. This class is mainly characterized by very severe and severe erosion classes. Non-degraded areas are concentrated in the CAPO area, distributed in the southern part of the study area and covering 15.85% of the area. The non-degraded class is dominated by paddy fields and settlements where the soil erosion can be described as either slightly eroded or moderately eroded. The dominant LD class in this study area is "potentially degraded" referring to the land that has not yet become degraded, but that has the potential to become degraded due to the changes in the LULC and inappropriate land use management. Indeed, changes in the LULC can cause severe soil erosion and LD (Firdaus et al., 2014). Inappropriate land use management is the main cause of LD, as has been discussed by Mohawesh et al. (2015). Potentially degraded land spreads into the CAPO and PFPO areas dominated by very slight and slight soil erosion.

The prioritization of watershed management was carried out to identify degraded land with high erosion activity, so that appropriate conservation measures can be taken to minimize soil erosion. The watershed was classified into three priority zones: high priority (highly degraded class and degraded class), medium priority (slightly degraded class), and low priority (potentially degraded class). The prioritization classes can be summarized as follows: 1) high priority, referring to the areas in the MCSW with high soil erosion (28.24%) and hence needing to be prioritized for soil conservation; 2) medium priority (17.94%), characterized by a moderate slope, fine texture, deep solum, good drainage, and metamorphic rocks; and 3) low priority (37.96%), referring to areas facing very light erosion as well as those that are potentially vulnerable, that is, in their existing condition, there is no significant erosion, but if they are not protected, soil erosion will occur. Therefore, soil conservation measures are required to protect against erosion. In our study, prioritization was based on the spatial unit of the LD class. This approach differed from many other studies in watershed prioritization, which have generally been based on prioritizing sub-watershed units (Choudhary et al., 2020). The two approaches are increasingly becoming complementary in watershed management.

4 CONCLUSION

Most of the study area is characterized by a low rate of soil erosion (38%), while a small part (i.e., 4,275.8 ha, or 7.30%) manifests very severe soil erosion. The resulting map indicates the degree of LD. The results demonstrated that the watershed comprises degraded land (very high, high, and moderate degraded classes), occupying 29,911.95 ha (49%), and non-degraded land (potentially degraded and non-degraded land classes), occupying 31,506 ha (51%) of the study area. The highly degraded class was found to cover an area of 7% of the total area, predominantly in the PFPO areas. The non-degraded class was found to cover an area of 15.85%, particularly in the CAPO areas. The analysis of LD classes and prioritization for handling revealed that 16,538.02 ha (28.24%) of the MCSW areas located on slopes >15% and where soil erosion is very severe or severe need to be treated with soil conservation measures. The results of this study indicate that areas with forest landcover face less LD than cultivated areas, and so forest clearing in cultivation areas needs to be accompanied by land conservation efforts. The study has found that the most significant parameter causing soil erosion–induced LD was LULC as it demonstrated the highest weights in PFCF and PFPO areas. In agricultural areas (CAPO), the highest weights were assigned to the management and productivity parameters. In terms of methodology, the results show the effectiveness of integrating the USLE model with the GIS and AHP in an MCDA environment to map LD classes and the spatial prioritization of watershed restoration. The combination of methodologies used in this study can be used as tools for prioritization management in humid tropical watersheds.

DATA AVAILABILITY STATEMENT

The original contributions presented in the study are included in the article/Supplementary Material further inquiries can be directed to the corresponding author.

AUTHOR CONTRIBUTIONS

WA and IN were the main contributors of this article, and both contributed to the conceptualization, methodology, analysis, validation, writing the draft, and article preparation. WD, JS, SL, MD, TT, YS, and DS were the contributor members. WD provided support in conceptualization, methodology and analysis, and writing the manuscript. JS, SL, MD, TT, YS, and DS contributed to the field survey and reviewed the manuscript.

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REFERENCES

AbdelRahman, M. A. E. (2014), *Assessment of Land Degradation and Land Use Planning by Using Remote Sensing and GIS Techniques in Chamarajanagar District, Karnataka*, Bangalore (India): University of Agricultural Sciences. dissertation.

AbdelRahman, M. A. E., Natarajan, A., Hegde, R., and Prakash, S. S. (2019). Assessment of Land Degradation Using Comprehensive Geostatistical Approach and Remote Sensing Data in GIS-Model Builder. *Egypt. J. Remote Sensing Space Sci.* 22, 323–334. doi:10.1016/j.ejrs.2018.03.0021110-9823

Abera, W., Assen, M., and Budds, J. (2020). Determinants of Agricultural Land Management Practices Among Smallholder Farmers in the Wanka Watershed, Northwestern highlands of EthiopiaI. *Land Use Policy*, 99, 104841. doi:10.1016/j.landusepol.2020.104841

Adimihardja, A. (2008). Teknologi Dan Strategi Konservasi Tanah Dalam Kerangka Revitalisasi Pertanian. *Pengembangan Inovasi Pertanian* 1 (2), 105–124.

Ahmad, N., and Pandey, P. (2018). Assessment and Monitoring of Land Degradation Using Geospatial Technology in Bathinda District, Punjab, India. *Solid Earth* 9, 75–90. doi:10.5194/se-9-75-2018

Alewell, C., Borrelli, P., Meusburger, K., and Panagos, P. (2019). Using the USLE: Chances, Challenges and Limitations of Soil Erosion Modelling. *Int. Soil Water Conservation Res.* 7 (3), 1–23. doi:10.1016/j.iswcr.2019.05.004

Bai, Z. G., Dent, D. L., Olsson, L., and Schaepman, M. E. (2008), *Global Assessment of Land Degradation and Improvement. 1*, Identification by remote sensing. Wageningen: Report 2008/01. ISRIC – World Soil Information.

Brevik, E. C., Cerdà, A., Mataix-Solera, J., Pereg, L., Quinton, J. N., Six, J., et al. (2015). The Interdisciplinary Nature of SOIL. *Soil* 1, 117–129. doi:10.5194/soil-1-117-2015

Chen, C.-C., McCarl, B., and Chang, C.-C. (2012). Climate Change, Sea Level Rise and rice: Global Market Implications. *Climatic Change* 110, 543–560. doi:10.1007/s10584-011-0074-0

Chen, J., Chen, S., Fu, R., Wang, C., Li, D., Jiang, H., et al. (2021). Simulation of Water Hyacinth Growth Area Based on Multi-Source Geographic Information Data: An Integrated Method of WOE and AHP. *Ecol. Indicators* 125, 107574. doi:10.1016/j.ecolind.2021.107574

Choudhary, S., Jaiswal, R. K., Nema, A. K., Thakur, A., and Gangwar, A. (2020). Watershed Prioritization for Soil Conservation Measures Using USPED Model in Nagwan Watershed. *Jour. Soi. Wat. Conser.* 19 (3), 235–243. doi:10.5958/2455-7145.2020.00032.6

Development Planning Agecy (2012). *Purwakarta Regional Spatial Plan 2011-2031.* Purwakarta: Development Planning Agecy (Bappeda) of Purwakarta Regency.

Dotterweich, M. (2013). The History of Human-Induced Soil Erosion: Geomorphic Legacies, Early Descriptions and Research, and the Development of Soil Conservation-A Global Synopsis. *Geomorphology*, 201, 1, 34. doi.org/doi:10.1016/j.geomorph.2013.07.021

Firdaus, R., Nakagoshi, N., Idris, A., and Raharjo, B. (2014). "The Relationship between Land Use/land Cover Change and Land Degradation of a Natural Protected Area in Batang Merao Watershed, Indonesia," in *Designing Low Carbon Societies in Landscapes* (Tokyo, Japan: Springer), 239–251. doi:10.1007/978-4-431-54819-5_14

García-Ruiz, J. M., Nadal-Romero, E., Lana-Renault, N., and Beguería, S. (2013). Erosion in Mediterranean Landscapes: Changes and Future Challenges. *Geomorphology* 198, 20–36. doi:10.1016/j.geomorph.2013.05.023

Gessesse, B., Bewket, W., and Bräuning, A. (2015). Model-Based Characterization and Monitoring of Runoff and Soil Erosion in Response to Land Use/land Cover Changes in the Modjo Watershed, Ethiopia. *Land Degrad. Develop.* 26 (7), 711–724. doi:10.1002/ldr.2276

Halefom, A., Teshome, A., and Sisay, E. (2019). GIS-based MCDA Model to Assess Erosion Sensitivity in Gumara Watershed, Blue Nile, Basin Ethiopia. *Asian J. Appl. Sci.* 12 (2), 61–70. doi:10.3923/ajaps.2019.61.70

Hammer, W. I. (1978). *Soil Conservation Report*. INS/78/006. Technical Note No. 7. Bogor: Soil Research Institut.

Han, J., Ge, W., Hei, Z., Cong, C., Ma, C., Xie, M., et al. (2020.) Agricultural Land Use and Management Weaken the Soil Erosion Induced by Extreme Rainstorms. *Agric. Ecosyst. Environ.*, 301, 107047. doi:10.1016/j.agee.2020.107047

Hassan, M. S., Mahmud-Ul-Islam, S., and Rahman, M. T. (2015). Integration of Remote Sensing and GIS to Assess Vulnerability of Environmental Degradation in North-Western Bangladesh. *Jgis* 07, 494–505. doi:10.4236/jgis.2015.75040

Hermassi, T., and Amami, H. (2018). "Evolution of Water and Soil Conservation Interventions in the Merguellil Catchment: Impact on the Agricultural Land protection and the Income of the Local Population," in *Recent Advances in Environmental Science from the Euro-Mediterranean and Surrounding* *regions.EMCEI 2017. Advances in Science, Technology & Innovation (IEREK Interdisciplinary Series for Sustainable Development)*. Editors A. Kallel, M. Ksibi, H. Ben Dhia, and N. Khélifi (Cham: Springer). doi:10.1007/978-3-319-70548-4_252

Hong, H., Jaafari, A., and Zenner, E. K. (2019). Predicting Spatial Patterns of Wildfire Susceptibility in the Huichang County, China: an Integrated Model to Analysis of Landscape Indicators. *Ecol. Indicators* 101, 878–891. doi:10.1016/j.ecolind.2019.01.056

Inter-agency, Expert Group on Sdg Indicators [Iaeg-SDGs] (2016). 'IAEG-SDGs - SDG Indicators'. Sustainable Development Goals. Available online at: https://unstats.un.org/sdgs/metadata/.

Indonesian Center for Agricultural Land Resources Research and Development (2017). *Soil Map Scale 1: 50.000*. Bogor: (ICALRRD Agency for Agricultural Research and Development.

Indonesian Geospatial Information Agency [IGIA] (2015). *Topographical Map Scale 1:25.000*. Bogor: Cibinong.

Indonesian Ministry of Forestry [IMF] (2013). *Regulation of the Director General of Watershed Management and Social Forestry Management No*. P. 4/V-SET/2013 concerning Technical Guidelines for Compilation of Spatial Data on Critical Land. Jakarta: Indonesian Ministry of Forestry.

Indonesian Statistics (2020). Area of Critical Land by Province and Land Criticality Level in 2006, 2011, 2013, 2018. Available online at: https://www.bps.go.id/dynamictable/2017/02/08/1231/luas-dan-penyebaran-lahan-kritis-menurut-provinsi-tahun-2005-2013-ribu-ha-.html (accessed December, , 2020).

Jang, T., Vellidis, G., Hyman, J. B., Brooks, E., Kurkalova, L. A., Boll, J., et al. (2013). Model for Prioritizing Best Management Practice Implementation: Sediment Load Reduction. *Environ. Manage.* 51 (1), 209–224. doi:10.1007/s00267-012-9977-4

Kefi, M., Yoshino, K., Zayani, K., and Isoda, H. (2009). Estimation of Soil Loss by Using Combination of Erosion Model and GIS: Case of Study Watersheds in Tunisia. *J. Arid Land Stud.* 19 (1), 287–290.

Khaledian, Y., Kiani, F., Ebrahimi, S., Brevik, E. C., and Aitkenhead-Peterson, J. (2017). Assessment and Monitoring of Soil Degradation during Land Use Change Using Multivariate Analysis. *Land Degrad. Develop.* 28, 128–141. doi:10.1002/ldr.2541

Labrière, N., Locatelli, B., Laumonier, Y., Freycon, V., and Bernoux, M. (2015). Soil Erosion in the Humid Tropics: A Systematic Quantitative Review. *Agric. Ecosyst. Environ.* 203, 127–139. doi:10.1016/j.agee.2015.01.027

Lal, R. (2015). Restoring Soil Quality to Mitigate Soil Degradation. *Sustainability* 7, 5875–5895. doi:10.3390/su7055875

Li, L., Wang, Y., and Liu, C. (2014). Effects of Land Use Changes on Soil Erosion in a Fast Developing Area. *Int. J. Environ. Sci. Technol.* 11, 1549–1562. doi:10.1007/s13762-013-0341-x

López-García, E. M., Torres-Trejo, E., López-Reyes, L., Flores-Domínguez, Á. D., Peña-Moreno, R. D., and López-Olguín, J. F. (2019). Estimation of Soil Erosion Using USLE and GIS in the Locality of Tzicatlacoyan, Puebla, México. *Soil Water Res.* 15, 9–17. doi:10.17221/165/2018-SWR

Meteorological, Climatological, and Geophysical Agency (2020). (BMKG) Online Data, Central of Database. Only in Certain Cities/Districts). Available online at online: https://dataonline.bmkg.go.id/admin/(accessed October 10).

Mohammed, A., Adugna, T., Adugna, T., and Takala, W. (2018). Morphometric Analysis and Prioritization of Watersheds for Soil Erosion Management in Upper Gibe Catchment. *J.Degrade.Min.Land Manage.* 06 (1), 1419–1426. doi:10.15243/jdmlm.2018.061.1419

Mohawesh, Y., Taimeh, A., and Ziadat, F. (2015). Effects of Land Use Changes and Soil Conservation Intervention on Soil Properties as Indicators for Land Degradation under a Mediterranean Climate. *Solid Earth* 6, 857–868. doi:10.5194/se-6-857-2015

Narendra, B. H., Widiatmaka, Kusmana, C., Karlinasari, L., and Machfud (2019), Critical Land Mapping for the Development of Biomass-Based Energy in East Lombok Regency, Indonesia, IOP Conf. 444 Series: Earth and Environmental Science, July 03-August 12, 2019. Bistok, UK: IOP Publiser, 314, 012072. doi:10.1088/1755-1315/314/1/012072

Nkonya, E., Braun, J. V., Mirzabaev, A., Le, Q., Kwon, H. Y., and Kirui, O. (2016). *Concepts and Methods of Global Assessment of the Economics of Land Degradation and Improvement Economics of Land Degradation and Improvement—A Global Assessment for Sustainable Development*. Berlin, Germany: Springer, P15–P32.

Omuto, C. T., Balint, Z., and Alim, M. S. (2014). A Framework for National Assessment of Land Degradation in the Drylands: a Case Study of Somalia. *Land Degrad. Develop.* 25, 105–119. doi:10.1002/ldr.1151

Saaty, T. L. (2008). Decision Making with the Analytic Hierarchy Process. *Ijssci* 1 (1), 83–98. doi:10.1504/ijssci.2008.017590

Shareef, M. A., Ameen, M. H., and Ajaj, Q. M. (2021). Change Detection and GIS-Based Fuzzy AHP to Evaluate the Degradation and Reclamation Land of Tikrit City, Iraq. *Geodesy and Cartography* 46 (4), 194–203. doi:10.3846/gac.2020.11616

Singh, G., Singh, R. M., Singh, S., Kumar, A. R. S., Jaiswal, R. K., Chandola, V. K., et al. (2019). Multicriteria Analytical Hierarchical Process-Based Decision Support System for Critical Watershed Prioritization of Andhiyarkhore Catchment. *Indian J. Soil Conservation* 47 (3), 263–272.

Taslim, R., Mandala, M., Mandala, M., and Indarto, I. (2019). Pengaruh Luas Penggunaan Lahan Terhadap Laju Erosi : Studi Pada Beberapa Das Di Wilayah Tapal Kuda Jawa Timur (The Effect of Land Use on Erosion Rate: a Study at Several Watersheds in Tapal Kuda Region, East Java). *Jppdas* 3 (2), 141–158. doi:10.20886/jppdas.2019.3.2.141-158

Tukura, N. G., Akalu, M. M., Hussein, M., and Befekadu, A. (2021). Morphometric Analysis and Sub-watershed Prioritization of Welmal Watershed, Ganale-Dawa River Basin, Ethiopia: Implications for Sediment Erosion. *J. Sediment. Environ.* 6, 121–130. doi:10.1007/s43217-020-00039-y

Turner, K. G., Anderson, S., Gonzales-Chang, M., Costanza, R., Courville, S., Dalgaard, T., et al. (2016). A Review of Methods, Data, and Models to Assess Changes in the Value of Ecosystem Services from Land Degradation and Restoration. *Ecol. Model.* 319, 190–207. doi:10.1016/j.ecolmodel.2015.07.017

United Nations Convention to Combat Desertification [Unccd] (2017). *Global Land Outlook*. 1st ed. Bonn. Available online at https://www.unccd.int/actions/global-land-outlook-glo.

Vahabi, J., and Nikkami, D. (2008). Assessing Dominant Factors Affecting Soil Erosion Using a Portable Rainfall Simulator. *Int. J. Sediment Res.* 23 (4), 376–386. doi:10.1016/s1001-6279(09)60008-1

Weerakoon, K. (2014). Suitability Analysis for Urban Agriculture Using GIS and Multi-Criteria Evaluation. *Int. J. Agric. Sci. Tech. (Ijast)* 2 (2), 69. doi:10.14355/ijast.2014.0302.03

Widiatmaka, A. W., Setiawan, Y., and Walter, C. (2016). Assessing the Suitability and Availability of Land for Agriculture in Tuban Regency, East Java, Indonesia. *Appl. Environ. Soil Sci.* 2016, 1–13. doi:10.1155/2016/7302148

Wischmeier, W. H., and Smith, D. D. (1978). *Predicting Rainfall Erosion Losses: A Guide to Conservation Planning (No. 537)*. Oregon US: Department of Agriculture, Science and Education Administration, Oregon State University.

Zhao, X., Dai, J., and Wang, J. (2013). GIS-based Evaluation and Spatial Distribution Characteristics of Land Degradation in Bijiang Watershed. In *SpringerPlus* 2, (1), 1–8. doi:10.1186/2193-1801-2-S1-S8

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Peatlands Are More Beneficial if Conserved and Restored than Drained for Monoculture Crops

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Peatlands are especially important but fragile tropical landscapes. The importance of peatlands is owing to their ability to 1) sequester a considerable amount of terrestrial carbon, 2) store freshwater, and 3) regulate floods during the rainy season. Nowadays, extensive peatland degradation occurs because of peatland utilization for agriculture purposes, causing severe environmental consequences such as carbon emission, loss of biodiversity, risk of flooding, and peat fire. Meanwhile, local planners and decision makers tend to overlook the long-term strategic function of peatlands for carbon storage and hydrological regulation, preferring peatland utilization for short-term economic benefits. The objective of our study is to quantify the total ecosystem services (except biodiversity) of a tropical peatland landscape in various peat-utilization scenarios to help build awareness among local planners and decision makers on the strategic tradeoff between peatland utilization and restoration. Studies on the total ecosystem services in a tropical peatland landscape involving hydrological regulation are still rare. Based on the net present value calculation, provisioning services, carbon regulation, and hydrological regulation in our study area account for 19, 70, and 11% of the total ecosystem services, respectively. Based on uncertainty analysis, at any combination of the social cost of carbon emission (within a range of USD 52.7–USD 107.4) and discount rate (within a range of 5–10%), the enrichment of peatlands with paludiculture crops (e.g., jelutong) shows superior ecosystem services compared to other peatland-utilization scenarios. Conversely, planting peatlands with monoculture crops, which are associated with peatland drainage, shows a rapid decrease in the total ecosystem services. The fluvial carbon export in our study, which is often neglected in a peatland carbon budget, increases the estimate of the total carbon budget by 8%. Restoring undrained peatlands with paludiculture crops such as jelutong contributes positively to carbon sequestration and potentially reduces carbon emissions by 11%. These quantitative findings can help local planners and decision makers in understanding the tradeoff between the long-term benefits of peatland restoration and the short-term economic benefits of peatland utilization for monoculture crops.

Keywords: carbon loss, monoculture crop, hydrological ecosystem services, paludiculture, fluvial carbon lost

INTRODUCTION

Tropical peatlands have one of the largest carbon deposits in terrestrial landscape, contributing to 65% of the global peatland carbon storage (Moore et al., 2013; Hergoualc'h and Verchot, 2014; Hapsari et al., 2017). Peatlands are especially important but fragile tropical landscapes. The importance of peatlands is related to their ability to 1) hold 525 Gt of carbon, 2) store 10% of the global freshwater, 3) host biodiversity, and 4) regulate floods during the rainy season (Joosten and Clarke, 2002; Jaenicke et al., 2008; Yule, 2010; Page et al., 2011; Acreman and Holden, 2013; Gao et al., 2016).

Tropical peatlands have become increasingly an important global issue owing to their rapid utilization for agriculture purposes. Utilizing tropical peatlands for agricultural purposes necessitates draining. Draining peatlands creates aerobic conditions and exposes organic carbon stored for over a thousand years to decomposition, resulting in CO_2 emissions and peat subsidence (Couwenberg et al., 2010; Yule, 2010; Page et al., 2011; Hooijer et al., 2012; Jauhiainen et al., 2016; Tonks et al., 2017; Moore et al., 2018). Upon peatland utilization for agriculture, the rate of emissions increases due to the drainage construction enhancing activities of aerobic microorganisms (Wösten et al., 2008; Turetsky et al., 2015; Miettinen et al., 2016).

Intensive peatland degradation caused by drainage and associated peat fires contributes to approximately 30% of the global carbon land forestry-based emissions (Couwenberg et al., 2010), whereas the other 70% is presumably associated with forest degradation and deforestation. Therefore, peatland restoration has become an essential global environmental policy issue (Ferre and Martin-Ortega, 2019), and tropical peatland management is an important focus for the UN decade of ecosystem restoration (UN Environment Programme, 2019). Peatland restoration and utilization are frequently in conflict in a landscape because peatland restoration requires waterlogged conditions, whereas peatland utilization for agriculture requires drainage. As soon as a peatland is used, its natural ecosystem changes, leading to emissions and threatening peatland sustainability (Evan et al., 2019).

Peatland restoration has become challenging because local planners and decision makers tend to undervalue the long-term strategic function of peatlands for carbon storage and hydrological regulations. Owing to a lack of understanding, they often opt for peatland utilization for short-term economic benefits. One way to improve the peatland stakeholders' understanding of the tradeoff between peatland restoration and utilization is through ecosystem service assessment (Tallis et al., 2008; Fisher et al., 2009; Goldstein et al., 2012; Macfadyen et al., 2012; Albert et al., 2014; Cohen-Shacham et al., 2015). The ecosystem service concept has become important to communicate human–environment interactions (Busch et al., 2012; Paavola and Hubacek, 2013). Understanding which ecosystem services are provided and where they are provided in a peatland ecosystem will help understand synergies and tradeoffs in peatland management (Paavola et al., 2009; Bagstad et al., 2014). Normally, provisioning and regulating ecosystem services compete in a landscape (Tilman et al., 2002; Rodríguez et al., 2006). An ecosystem service tradeoff occurs when the provision of one service is maximized at the cost of reducing another service.

The ecosystem services of tropical peatlands have often been studied (Kimmel and Mander, 2010; Sumarga et al., 2015; Suwarno et al., 2016; Uda et al., 2017), but none of these studies sufficiently included the hydrological regulation of peatlands (Yule, 2010; Ferre and Martin-Ortega, 2019). A study on boreal peatlands includes biodiversity in an ecosystem service analysis and found a strong tradeoff between biodiversity and ecosystem services in drained peatlands (Juutinen et al., 2020). Despite the very rich biodiversity in our study site, we did not yet include the biodiversity component in the total ecosystem service calculation due to a complex interaction between biodiversity and ecosystem services (Johnson et al., 2012). The objective of our study is to value the provisioning, regulation, and hydrological ecosystem services of peatlands to help build awareness among local planners and decision makers on the strategic values of peatlands. Ecosystem services have been seen as a powerful tool to understand the socioecological and economic benefits of the environment and to consider them in spatial planning and environmental policy. The result of this study can be used for two purposes: 1) planning payment for ecosystem services for those who keep peatlands intact, and 2) assist local policymakers in deciding whether to use peatlands.

METHODS

Site Description

There are three neighboring provinces in Sumatra covering approximately 42% of the Indonesian peatlands, i.e., Riau, Jambi, and South Sumatra Provinces with the peatland areas of 4.8, 0.7, and 1.2 million ha, respectively. Parts of the peatlands in these provinces had been used for plantations. This study was conducted in South Sumatra Province, Indonesia (**Figure 1**). South Sumatra Province has a size of 9 million ha. In 2015, South Sumatra Province experienced severe peat fires. Our study site belongs to Merang-Ngirawan peat hydrological units. These units have been adopted by the Indonesian government as a landscape unit for peatland restoration, and there are hundreds of peat hydrological units throughout Indonesia (BRG, 2019). A peat hydrological unit is commonly characterized by the existence of a lens-shaped peat dome with a marked peat depth around the center of the dome. The radius of peat domes in Indonesia ranges from 5 to 10 km (Dommain et al., 2010). We included hydrological ecosystem services in our analysis and used a watershed as a boundary of the analysis. The area covered by the watershed boundary is 97,286 ha, which contains mainly secondary peat forests (**Figure 1**). These secondary forests had been logged over sometime in the past, and since then, the vegetation has been regenerating naturally.

Scenarios

In our scenario, peatlands are classified as undisturbed when they are not drained. At the current condition, a part of the

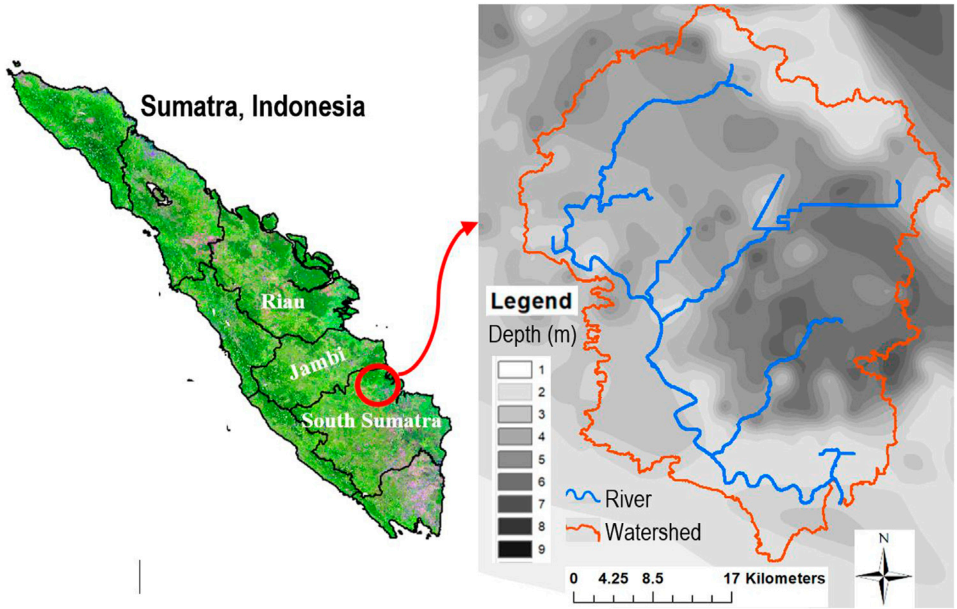

FIGURE 1 | Study site in South Sumatra Province, Indonesia with a peat-depth map.

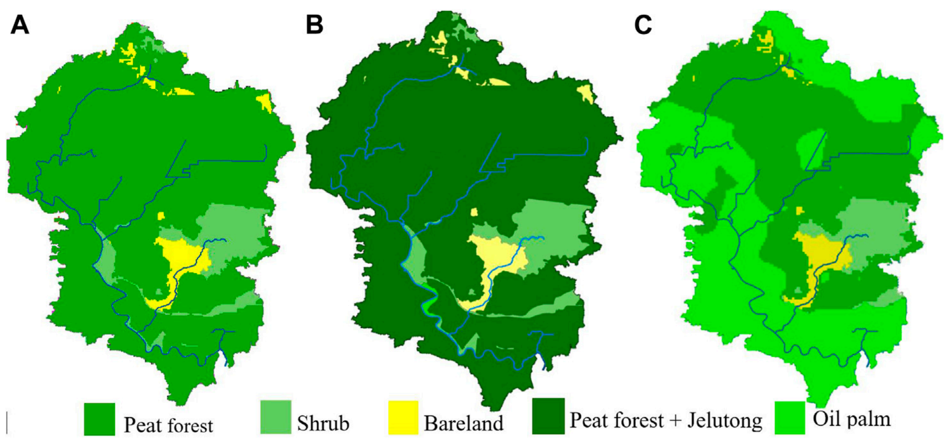

FIGURE 2 | Schematic land-use type in **(A)** Scenario 1, **(B)** Scenario 2), and **(C)** Scenario 3

study area had been drained for acacia plantation, especially in the western and eastern parts of the watershed boundaries. Further peatland utilization in the study area is expected to occur in the future. We examined ecosystem services for three land-use scenarios (**Figure 2**). Scenario 1 is a proxy for the current condition. In Scenario 1, the peatland is assumed to be unutilized. In Scenario 2, the peatland cover is enriched with jelutong crops (*Dyera* sp.). These crops can sustainably be grown in undrained conditions and do not lead to soil subsidence (Sumarga et al., 2016). Jelutong crops (*Dyera sp.*) can be planted as an intercropping enriching the current land cover aiming to increase carbon sequestration. They produce latex that can be collected and be sold in the market. In contrast to commercial plantation crops such as oil palm and acacia, jelutong crop does not require drainage. Therefore, jelutong is considered a suitable crop for degraded peatlands (paludiculture crop), but the provisioning service obtained from jelutong latex is not as profitable as those from monoculture crops (e.g., oil palm). In Scenario 3, peat with a depth of less than 3 m was converted into an oil-palm plantation. Oil palm requires peatland draining to grow optimally. According to the Indonesian regulation, only peat with a depth of less than 3 m can be used for economic benefits. It corresponds to 45,620 ha of the watershed area being studied (**Table 1**).

TABLE 1 | Land-use proportion and drainage conditions for each scenario.

Scenario	Land use (ha)				
	Forest	Oil palm	Bareland	Shrub	Water
1. Undrained peatlands	84,654	0	3,598	8,768	266
2. Undrained peatlands are enriched with paludiculture crops	84,654	0	3,598	8,768	266
3. Peatlands with a depth of less than 3 m are drained up to 90-cm groundwater level for oil palm	39,034	45,620	3,598	8,768	266

Valuation of Ecosystem Services

We analyzed provisioning and regulating ecosystem services for each scenario. Monetary benefits were assessed using net present value (NPV) with a discount rate of 10%. A period of 30 years was used for the NPV calculation, representing one cycle of oil-palm plantation. In 30 years, the oil-palm plantation must be replanted. When calculating the NPV for commodity provisioning services (e.g., oil palm and jelutong), we considered the time required from the beginning of utilization or restoration until their productive stage.

The monetary value of carbon sequestration and emission is based on the social cost of carbon emission. The social cost of carbon emission implies that if carbon is sequestered, there will be cost savings from not having to abate that carbon through other means. Based on a meta-analysis by Tol (2019), the social carbon cost was estimated at USD 106 t^{-1} C. We adopted the social cost of carbon value at USD 104.7 t^{-1} C (Interagency Working Group on Social Cost of Carbon, 2013; United States Environmental Protection Agency, 2013; Sumarga et al., 2016; Suwarno et al., 2016; Nordhaus, 2017).

Provisioning Services

The provisioning services in the study site comprise rattan collection, jelutong latex collection, and oil-palm production. When calculating the provisioning service, we consider the time required to reach a productive stage, e.g., jelutong and oil palm require 5 years after planting before it starts yielding.

The unit monetary values for commodities were derived from a study by Suwarno et al. (2016), i.e., 97.6, 98.8, and 812.8 USD ha^{-1} y^{-1} for rattan, jelutong latex, and oil-palm production, respectively.

Climate and hydrological regulations.

Climate regulations include a) carbon sequestration due to land-use changes and b) carbon emission due to peatland drainage. Meanwhile, hydrological regulations comprise a) fluvial carbon export and b) a loss of land due to subsidence and subsequent flooding.

Carbon Sequestration and the InVEST Model

We used the Integrated Valuation of Ecosystem Services and Tradeoff (InVEST) model to quantify C sequestration processes related to land cover and land-cover changes (Kareiva et al., 2011). InVEST can model and map change in ecosystem services across a watershed caused by land cover and land-management changes. It can provide monetary evaluation, which is useful to assess tradeoffs between ecosystem services and compare changes in ecosystem services under different land uses and land-use management scenarios. InVEST has been used in various applications around the world (Tallis and Polasky, 2009).

The InVEST model requires the following input: 1) land-use map, 2) carbon-pool data for each land-use type, 3) discount rate, and 4) the social cost of carbon. Land-use data, peat depth, and distribution data were derived from Kelola Sendang Project (2018) and LPPM-IPB (2019). Carbon-pool data for different land uses, i.e., forest and oil palm, were derived from Kotowska et al. (2015); Khasanah et al. (2015) and Irwan and Purwanto (2020) (**Table 2**).

Carbon Emission due to Peatland Drainage

During a prolonged dry season, even water levels in pristine peatlands can drop below 40 cm (Takahashi et al., 2002; Hirano et al., 2012; Deshmukh et al., 2021). Therefore, carbon emissions can occur both in pristine and degraded peatlands.

Several studies have been conducted since 2005 to determine the fluxes of carbon in different land uses in tropical peatlands (Page et al., 2009; Couwenberg and Hooijer, 2013; Hergoualc'h and Verchot, 2011; Hirano et al., 2012). Couwenberg and Hooijer (2013) estimated the average carbon losses amount of 18 tC ha^{-1} y^{-1} for oil palm and acacia plantations without significant differences between plantation types. Jauhiainen et al. (2016) found the total emission was as high as 22 tC ha^{-1} y^{-1} in an acacia plantation (31–46 months old). Moreover, Comeau et al. (2013) reported the carbon fluxes of 28.4, 18.5, and 16.0 tC ha^{-1} y^{-1} in oil-palm plantation, degraded peat forest, and undisturbed peat forest, respectively, using a case study in Jambi, Sumatra, Indonesia. All these studies show marked differences in the emission rate despite having the same land-use categories. The difference may be attributable to the different water-table levels during sampling periods. A new study using paired eddy covariance estimated carbon emission rates of 4.2 and 10.8 tC ha^{-1} y^{-1} in undisturbed and disturbed peatlands, respectively, in South Sumatra (Desmukh et al., 2021). In our calculation, we used

TABLE 2 | Carbon-pool inputs for the InVEST model were derived from Kotowska et al. (2015), Khasanah et al. (2015), and Irwan and Purwanto (2020).

Land use	Above ground biomass	Roots	Necromass
	(tC ha^{-1})		
Peat forest	102.7	41.6	7.3
Peat forest enriched with jelutong	111.9	66	8.5
Oil-palm plantation	40.3	13.3	5.9
Shrub	23	4	1
Bareland	3.6	0	0

TABLE 3 | Method and data description.

Analysis		Methods	Data input (sources)	Remarks
1. Creation of watershed peat-depth map (**Figure 1**)		Peat-depth map is required to model the impact of subsidence on the potential flooding and land loss. The map was created using GIS surface interpolation based on 134 field sampling points	Field sampling, including peat depth, bulk density, and water level; Soil map (BBSDLP, 2015; LPPM-IPB, 2019)	The number of field sampling points were 134 conducted in May 12th–22nd 2019
2. Creation of watershed and stream network maps (**Figure 1**)		We created the watershed and stream network from digital elevation data. The watershed map and stream network are required for calculating the fluvial carbon export	Digital elevation map with 0.5-m resolution (Kelola Sendang Project, 2018)	Watershed map is also used for the SWAT and InVEST modeling
3. Calculation of the regulation ecosystem services	Carbon sequestration using InVEST	We calculated the carbon sequestration on a watershed scale using the InVEST model. The calculations were based on the land-cover change map due to jelutong planting (Scenario 2) and oil palm (Scenario 3) on a watershed scale	1. Land-use change map (Kelola Sendang Project, 2018) 2. Carbon-pool data (see **Table 2**) 3. Social cost of carbon emission, discount rate	Example of the InVEST model input and output is submitted as Supplement Information 01
	Carbon emission (CO_2, CH_4)	We calculated CO_2 and CH_4 emissions on a watershed scale considering the peat-distribution map and different peatland utilization	1. Peat-distribution map on a watershed scale (see nr. 1) 2. Peat-utilization map (Kelola Sendang Project, 2018) 3. Yearly peatland emission rate (Deshmukh et al., 2021)	
4. Calculation of the hydrological ecosystem services	Loss of land due to permanent flooding	Peatland decomposition and erosion lead to land subsidence. GIS raster spatial analysis: a) create subsidence raster map and b) subtract peat-depth raster map with subsidence raster map. We valued the loss of land based on the opportunity cost of yield from jelutong crops	1. Peat-depth distribution in the watershed (see nr. 1) 2. Land elevation from digital elevation model 3. Peat-utilization map (Kelola Sendang Project, 2018) 4. Yearly peatland subsidence rate (Deshmukh et al., 2021)	
	Fluvial carbon export using SWAT model	We calculated fluvial discharge using the SWAT model. The discharge obtained from the SWAT model is multiplied by the concentration of fluvial total organic carbon (TOC)	Rainfall data from Chirps (2011–2020), water-table data (2015–2019), digital elevation model of 0.5-m resolution, land-use map, soil properties, river segment geometry, peat hydraulic conductivity, and bulk density (Kelola Sendang Project, 2018)	Example of the calculation is submitted as Supplement Information 02
5. Provisioning ecosystem services		Provisioning ecosystem services were calculated from commodity production (t y^{-1} ha^{-1}) and market prices (USD t^{-1})	Production and market price surveys (Suwarno et al., 2016)	
6. NPV		We valued each ecosystem service using net present value (NPV) with a discount rate of 10%. A period of 30 years was used for the NPV calculation. The monetary value of carbon sequestration and emission is based on the social cost of carbon emission (USD 107.4 t^{-1} C). The NPV was calculated using the NPV function in an Excel spreadsheet.	The social cost of carbon emission (Interagency Working Group on Social Cost of Carbon, 2013; Sumarga et al., 2016; Suwarno et al., 2016)	Example of the calculation is submitted as Supplement Information 03

the GHG balance of CO_2 and CH_4 with the rate of 15.5 and 3.3, and 39.8 and 1.9 tCO_2e ha^{-1} y^{-1} at undisturbed and degraded sites, respectively, derived from Deshmukh et al. (2021). The reason why we used the emission rate from this study are as follows: 1) it is the latest study conducted in Indonesia with relatively advanced equipment, i.e., eddy covariance tower, and 2) its study site (Riau Province) has the closest proximity to our study site (South Sumatra Province) compared to the other studies (see **Figure 1**).

Fluvial Carbon Export and the SWAT Model

The disturbance of hydrological stability in a peatland landscape either through vegetation-cover change or drainage can cause aerobic conditions, followed by a high decomposition rate of organic material in peatlands. This condition may lead to a higher production of dissolved organic carbon (DOC) and particulate organic carbon (POC) that is exported out of the peatland landscape through the fluvial system (Holden et al., 2004; Cowenberg et al., 2010). We require the river-discharge data and concentration of the total organic carbon (TOC) to calculate carbon export using fluvial systems. TOC is the sum of DOC and POC concentrations. We derived the TOC concentration from a study conducted in Indonesia by Moore et al. (2013). Based on Moore et al. (2013), the TOC concentration in fluvial systems is slightly higher in undisturbed (66.1 mg L^{-1}) than disturbed (51.1 mg L^{-1}) peatlands, but both have high TOC concentrations. The DOC from undisturbed peatlands is derived mainly from recent primary production (plant growth). Conversely, the DOC from disturbed peatlands consists mostly of much older (centuries to millennia) carbon from deep within the peat column.

Our study area does not have a discharge data record. We used the SWAT model version 2012 to simulate streamflow components. The SWAT model is a continuous model developed to simulate the impact of land cover/management changes on streamflow in a watershed with varying soil and land-use and management conditions over long periods (Neitsch et al., 2005; Arnold et al., 2012; Tarigan et al., 2016; Tarigan et al., 2017; Tarigan et al., 2020). Major model components include weather, hydrology, soil temperature and properties, plant growth, nutrients, and land management. The data source for the SWAT model in our study is described in **Table 3**.

The SWAT model simulates the quantity (volume) and quality of water on a daily time step, which can be used for assessing ecosystem services such as water yield for irrigation and hydropower, nutrient export to the river, and peak flood (Neitsch et al., 2005). In the last decade, SWAT has been applied widely around the world to model the impact of land use/land management on streamflow and nutrient export (especially N, P, and C), with hundreds of published examples (www.swat.tamu.edu).

The SWAT model can be calibrated using hard data and/or soft data (Arnold et al., 2015). Hard data are defined as measured time-series data (e.g., time-series streamflow data at a catchment outlet) commonly used in regression-based calibration (Seibert and McDonnell, 2002; Abbaspour, 2012). According to Arnold et al. (2015), soft data are defined as a signature on individual processes within a budget that may not be directly measured within the study area. Examples of soft data include the estimates of the ET/precipitation ratio, streamflow/precipitation ratio, and average annual runoff coefficients. Our study area does not have time-series discharge data, and we cannot calibrate the model using such data. White et al. (2012) developed an evaluation tool (SWAT Check program) to derive soft-data indicators (e.g., ET/precipitation ratio and average annual runoff coefficients). In our study, we employed soft data for calibration, such as the streamflow/precipitation ratio and ET/precipitation ratio derived from the SWAT Check program. Because the goal of the SWAT model in our analysis is simply to find annual water balance, not to predict daily streamflow, we consider that the soft-data calibration is sufficient for our purpose.

Loss of Land due to Permanent Flooding

Continuous peatland oxidation, compaction and shrinkage, and DOC fluvial export lead to land subsidence and the complete loss of peat layers (Hoojer et al., 2012; Ikkala et al., 2021). In the long term, the area will become relatively lower than its surroundings and potentially be permanently flooded after the rainy season. Large-scale subsidence studies conducted in oil-palm plantations on peatlands in South East Asia have reported that, at average water-table depths of 0.7 m, the subsidence rate is high at the beginning of peat draining, and after several years, it remains constant at approximately 5 cm y^{-1} (Hoojer et al., 2012). A relatively new study with a more rigorous method and comprehensive analysis reported land subsidence rates of 3.3 and 4.2 cm y^{-1} in undisturbed and degraded peatlands, respectively (Deshmukh et al., 2021).

To calculate the impact of land subsidence on the loss of land due to flooding on a watershed scale, we adopted the subsidence rate from the latest related study (Deshmukh et al., 2021). Besides, it is more rigorous, and its measurement site (Riau Province) is close to our study site (South Sumatra Province), which are both in Sumatra (**Figure 1**). The extent of the area flooded due to land subsidence was calculated using a peat contour depth in a raster form and digital elevation model with a 0.5-cm resolution, referenced to water levels in canals. We valued the loss of land based on the opportunity cost of using peatlands with paludiculture crops (e.g., jelutong crop).

RESULT

Provisioning Ecosystem Service

The provisioning services in the study area are mainly contributed by three commodity production, i.e., rattan, jelutong resin, and oil palm. Rattan is collected from peat forest. The forest areas in Scenario 1 and 2 are similar, i.e., 84,654 ha. However, in Scenario 2, jelutong latex is collected in addition to rattan collection from a similar area of forest (84,654 ha). In Scenario 3, peat with a depth of less than 3 m inside the watershed boundary was converted to oil-palm plantation. This conversion corresponds to an area of 45,620 ha. In Scenario 3, rattan and jelutong latex were also collected from the area that was not converted into oil palm. Scenario 3 has the highest provisioning ecosystem services associated with oil-palm production (**Table 4**).

Based on the SWAT modeling, the annual discharge from the undisturbed and disturbed watersheds are 1,476 and 1,500 mm, respectively (**Figure 3**).

We used soft data to calibrate the SWAT simulation. We compared four soft-data parameters obtained from the SWAT simulation of the study area and a reference area (see **Table 5**). Reference values were obtained from a SWAT study in a neighboring province (Jambi, **Figure 1**) with a Nash–Sutcliffe efficiency value of 85%, categorized as a very good result (Tarigan et al., 2020). Based on the comparison of the four soft-data parameters, the SWAT output in our study area is considered to be in an accepted range (**Table 5**).

To calculate the fluvial carbon export, we multiply annual discharge with the TOC. Based on a study by Moore et al. (2013) in Central Kalimantan, Indonesia, the DOC from undisturbed peatlands is derived mainly from the recent decomposition of vegetation, whereas the DOC from disturbed peatlands mostly comes from older and deep peat deposits. This could be the

TABLE 4 | Provisioning service in each scenario.

Scenario	Area of commodity (ha)			NPV (USD)
	Rattan	Jelutong	Oil palm	
1. Undrained peatlands	84,654	0	0	77,887,339
2. Undrained peatlands is enriched with paludiculture crops	84,654	84,654	0	107,330,443
3. Peatlands with a depth of less than 3 m are drained up to 90-cm ground water level for oil palm	39,034	39,034	45,620	288,855,879

Fluvial carbon export.

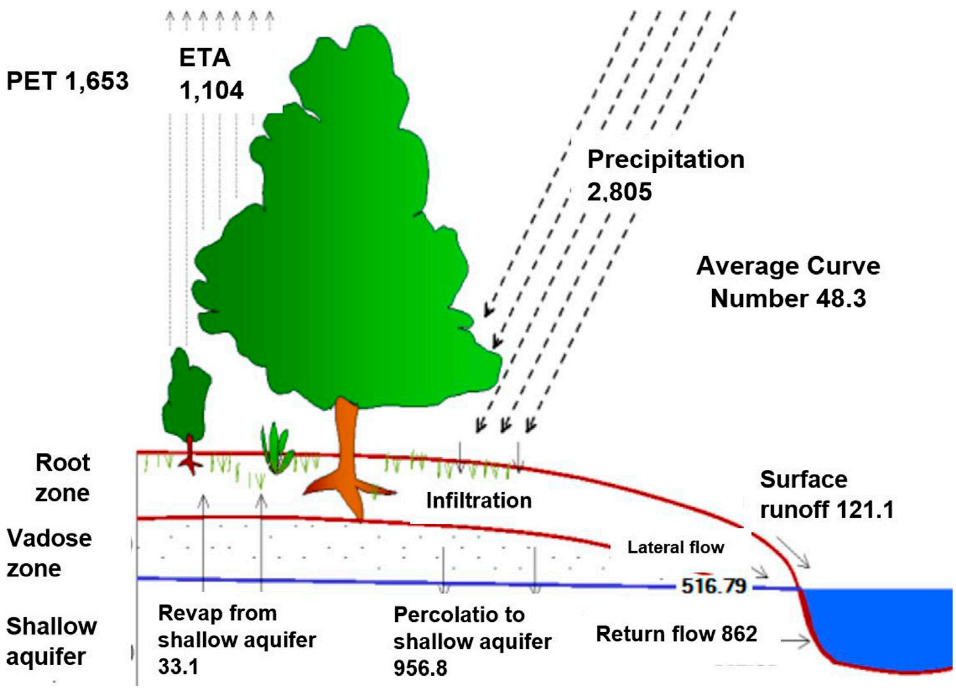

FIGURE 3 | Example of SWAT simulation outputs from our study area showing PET (potential evapotranspiration = 1,653 mm), actual evapotranspiration and transpiration (1,104 mm), total discharge (1,500 mm) comprising surface runoff (121.13 mm), lateral flow (516,79 mm), and return flow (862.03 mm).

TABLE 5 | Calibration and validation of SWAT simulation using soft data.

Soft-data type	SWAT simulation in our study area	Reverence value
Actual evapotranspiration	3.0 mm d^{-1}	3.2 mm d^{-1}
Potential evapotranspiration	4.5 mm d^{-1}	4.2 mm d^{-1}
ET and precipitation ratio	0.39	0.39
Streamflow and precipitation ratio	0.53	0.57

reason why the DOC concentration in the fluvial system was slightly higher in undrained peatlands (**Table 6**). The fluvial carbon loss in our study, which is frequently neglected in the peatland carbon budget, increases the estimate of the carbon budget by 8%.

Carbon Sequestration

Scenario 1 does not involve land use and land-cover change, and consequently, there is no carbon sequestration in Scenario 1. Carbon sequestration in Scenario 2 is associated with vegetation enrichment with jelutong (4 a). Moreover, carbon emission due to the land-use change in Scenario 3 is related to the conversion of peat forest into oil-palm plantations (4 b).

The restoration of undrained peatland with paludiculture crops such as jelutong contributes positively to carbon sequestration and potentially reduces carbon loss by 11%. The carbon sequestration or emission in Scenarios 2 and 3, 2,863,627 and −2,728,815 tC y^{-1}, correspond to the NPV of USD 95, 777, 499 and USD −91, 268, 513, respectively (−indicates emission and + sequestration).

Loss of Land due to Subsidence and Flooding

With the subsidence rates of 3.3 and 4.2 cm y^{-1} in undisturbed and disturbed peatlands, the peatland areas of 15,215 and 21,868 ha, respectively inside the watershed will irreversibly subside and be flooded in 30 years, (**Figure 4C,D**). The loss of land due to subsidence and flooding in undisturbed and disturbed peatlands

TABLE 6 | Carbon emission due to fluvial carbon export.

Scenario	Discharge (mm y^{-1})	Carbon export (tC y^{-1})	NPV (USD)
1. Undrained peatlands	1,476	82,591	−81,517,631
2. Undrained peatlands are enriched with paludiculture crops	1,476	82,591	−81,517,631
3. Peatlands with a depth of less than 3 m are drained up to 90-cm ground water level for oil palm	1,500	64,900	−64,056,432

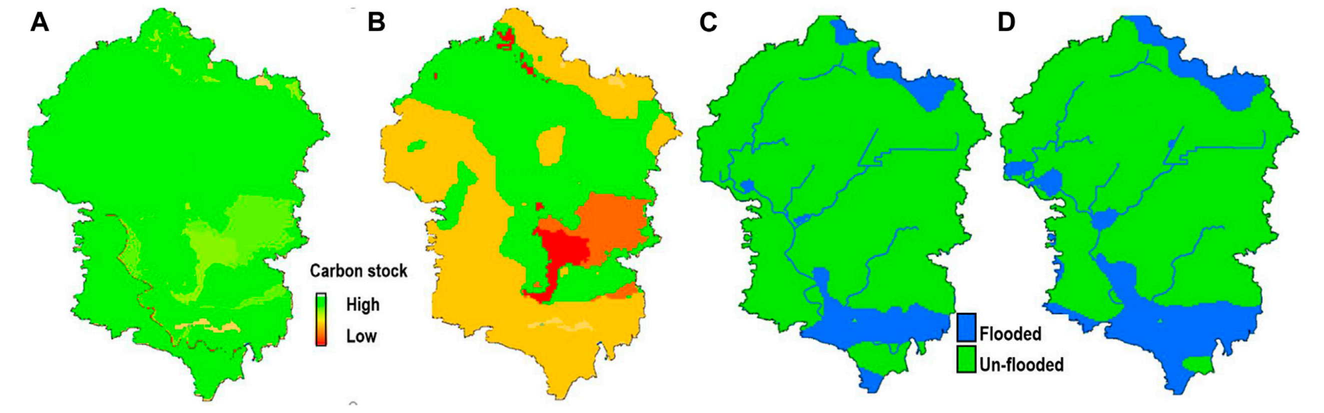

FIGURE 4 | InVEST carbon sequestration map due to the conversion of peat forest **(A)** into oil palm **(B)**; flooded area due to the land-use conversion from peat forest **(C)** into oil palm **(D)**.

corresponds to the loss of provisioning services in the future with an NPV of USD −2,703,911 and USD −3,886,239, respectively.

Carbon Emission due to Water-Table Fluctuation and Drainage

Carbon emission from tropical peatlands is mainly controlled by water-level fluctuation and drainage. The conversion of peat forest into monoculture plantations requires peatland draining. Peatland draining lowers the water table, triggering carbon emissions. However, during a prolonged dry season in Indonesia, the water table can also drop without drainage. Scenarios 1 and 2 do not involve peatland draining, and the drainage-related emissions in Scenarios 1 and 2 (undisturbed peatlands) are assumed to be similar, i.e., −431, 735 tC y^{-1}, corresponding to a negative NPV of USD −426, 121, 452. Emissions in these scenarios are mainly due to the water-level drop during the dry season and methane emissions (Deshmukh et al., 2021). The conversion of the forest area into an oil-palm plantation in Scenario 3 occurs only in a watershed area with a peat depth of less than 3 m, corresponding to 37% of the total forest within the watershed boundary. However, the emission in Scenario 3 is high, i.e., −714, 579 tC y^{-1}, corresponding to an NPV of USD −705, 288, 399.

Total Value of Ecosystem Services

The highest provisioning service is provided by Scenario 3, which is associated with oil-palm production. However, converting undisturbed peatlands into monoculture plantations (Scenario 3) contributes to the lowest value of the total ecosystem services in Scenario 3 (**Figure 5**). In Scenario 3, the social costs of carbon

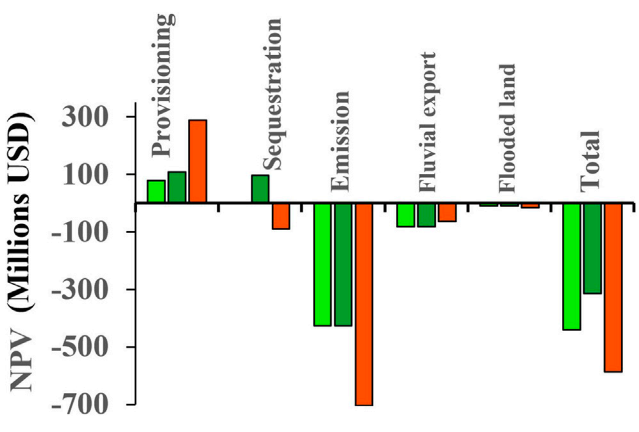

FIGURE 5 | Total ecosystem service value (USD).

emissions exceed the private benefits from oil-palm plantations in peat. These findings highlight the tradeoff between peatland restoration and utilization, posing strong challenges to peatland management in Indonesia today. Peatland utilization for oil palm provides the highest benefit to the private sector but implies long-term disadvantages related to the social cost of carbon emission and loss of land due to subsidence.

In our study area, the provisioning ecosystem service, carbon regulation ecosystem, and hydrological ecosystem service account for 19, 70, and 11% of the total ecosystem services, respectively.

DISCUSSION

Undrained peatlands still significantly contribute to carbon emissions, partly due to methane emissions and partly due to water-level fluctuation during the dry season. In all scenarios, including those without peatland drainage, carbon loss is dominated by peatland decomposition or oxidation processes. As a consequence, the total ecosystem service of all scenarios is strongly dominated by the negative contribution of carbon emission due to the peatland decomposition.

Utilizing peatlands with paludiculture crops (e.g., jelutong-*Dyera* sp.) increases carbon sequestration, which reduces the total carbon loss by 11% from peatlands. In contrast to commercial plantation crops, jelutong does not require drainage. Any peatland utilization involving draining will lead to high carbon loss. As demonstrated in our study, the land-use scenario associated with peatland drainage (e.g., monoculture plantation) rapidly reduces the total ecosystem service. Oil palm is the most profitable plantation crop in Indonesia, signifying the highest provision ecosystem service in our calculation (**Figure 5**), but it also increases carbon oxidation due to the associated peatland drainage, decreasing the total ecosystem services.

Translating the ecological impact of peatland restoration and utilization scenarios into ecosystem services provides a quantitative overview for local planners and decision makers to consider the environmental and economic benefits of alternative scenarios. However, the valuation of ecosystem service research can be highly uncertain. A valuation of nonmarketed goods (e.g., carbon cost) is an important source of uncertainty about ecosystem service analysis (Johnson et al., 2012). In our study, we find that there are three important sources of uncertainty: 1) calculating the NPV of ecosystem services involving the social cost of carbon emission and discount rate, 2) the SWAT model simulation, and 3) carbon sequestration simulation with the InVEST model.

Uncertainty Involving the Social Cost of Carbon and Discount Rate

The valuation of ecosystem services is commonly based on the social costs of carbon emission, i.e., USD 104.7 t^{-1} C and a discount rate of 10% (Sumarga et al., 2015; Suwarno et al., 2016; Tol, 2019). The main source of uncertainty in the social costs of carbon emission are the assumed abatement costs because of climate change and the selected discount rate. To assess the impact of these uncertainties on our analysis, we gradually varied the social cost of carbon emission from USD 104.7 t^{-1} C by 25% (USD 78.5) to 50% (USD 52.4) and then used different combinations of the discount rate, i.e., 5 and 10%, in our calculation (**Figure 6**).

From our calculation, we are certain that, at any combination of the social cost of carbon emission (within a range of USD 78.5–USD 107.4) and discount rate (within a range of 5–10%), Scenario 2 (planting peatland with paludiculture crops, e.g., jelutong) shows superior ecosystem services compared to other scenarios. Similarly,

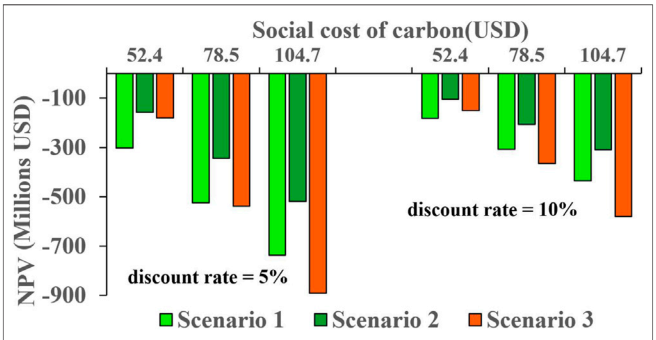

FIGURE 6 | Uncertainty analysis of the social costs of carbon emission and discount rate in the calculation of the NPV of ecosystem services in peatlands.

we are assured that, at any combination of the social cost of carbon emission (within a range of USD 78.5–USD 107.4) and discount rate (within a range of 5–10%), Scenario 3 (planting peatland with oil palm) show the lowest total ecosystem services.

Note that, in our analysis, we did not yet include an additional ecosystem service biodiversity in peatland, which will contribute positively to the total ecosystem services of Scenario 2. If we had included biodiversity ecosystem services in our analysis, the superiority of the total ecosystem services of peatland restoration (Scenario 2) over peatland utilization for monoculture crops (Scenario 3) would even be greater.

It is also important to note that, at any combination of the social cost of carbon emission (within a range of USD 52.7–USD 107.4) and discount rate (within a range 0f 5–10%), Scenario 1 (no peatland utilization) is not better than Scenario 2 (peatland utilization with paludiculture crop), signifying the benefit of peatland restoration using paludiculture crops.

Uncertainty Involving the SWAT and InVEST Models

A normal procedure to limit the uncertainty in the SWAT simulation model is through calibration and validation processes. There is a standardized procedure to calibrate and validate the SWAT model (Abbaspour, 2012). In our study, the SWAT model was calibrated using soft data due to the unavailability of sufficient time-series discharge data. The input parameter of the InVEST model is relatively simpler, involving a land-use change map, the carbon pools of different vegetation compartments, social cost of carbon emission, and discount rate. Unlike the SWAT model, the InVEST model is not a physical-based model, and there is no standardized procedure to calibrate the parameters. In our study, the carbon balance comprises 1) carbon sequestration, 2) fluvial carbon export, and 3) carbon emission due to oxidation. In any case, the total carbon budget calculation in our study area involving SWAT (Fluvial loss) and InVEST (Sequestration) account for 8 and 11%, respectively (**Figure 7**.), which is relatively smaller than the carbon

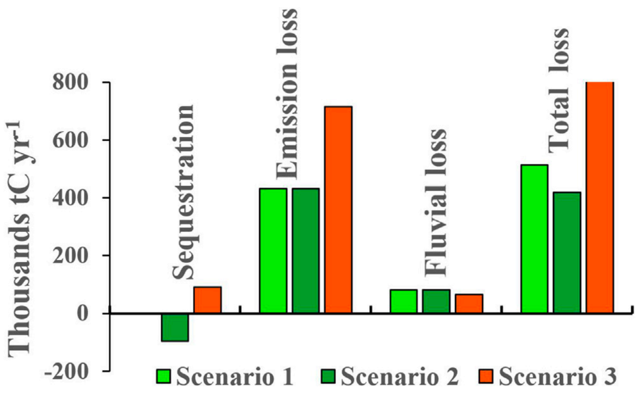

FIGURE 7 | Components of the carbon loss in our study area.

calculation involving peatland oxidation due to peatland draining (81%).

CONCLUSION

In contrast to monoculture plantation crops, paludiculture crop (e.g., jelutong) does not require drainage. Any peatland utilization involving draining will lead to a high carbon loss. Consequently, any peatland utilization associated with peat drainage reduces the total ecosystem services strongly. Peatland utilization for oil palm provides the highest benefit to the private sector but it implies long-term disadvantages related to the social cost of carbon emission and loss of land due to subsidence owing to associated peatland drainage. Our results can help local planners and decision makers realize the superiority of the long-term benefits of peatland restoration over the short-term economic benefits of peatland utilization.

DATA AVAILABILITY STATEMENT

The original contributions presented in the study are included in the article/**Supplementary Material**, further inquiries can be directed to the corresponding author.

AUTHOR CONTRIBUTIONS

ST, NZ, DB, and RK designed the research. ST wrote the manuscript. NZ, DB, RK and IZS reviewed the manuscript. YS coordinated field data collection.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fenvs.2021.749279/full#supplementary-material

REFERENCES

Abbaspour, K. C. (2012). *SWAT-CUP: SWAT Calibration and Uncertainty Programs - A User Manual*. Duebendorf, Switzerland: Eawag–Swiss Federal Institute of Aquatic Science and Technology, 106.

Acreman, M., and Holden, J. (2013). How Wetlands Affect Floods. *Wetlands* 33, 773–786. doi:10.1007/s13157-013-0473-2

Albert, C., Aronson, J., Fürst, C., and Opdam, P. (2014). Integrating Ecosystem Services in Landscape Planning: Requirements, Approaches, and Impacts. *Landscape Ecol.* 29, 1277–1285. doi:10.1007/s10980-014-0085-0

Arnold, J. G., Youssef, M. A., Yen, H., White, M. J., Sheshukov, A. Y., Sadeghi, A. M., et al. (2015). Hydrological Processes and Model Representation: Impact of Soft Data on Calibration. *Trans. ASABE* 58 (6), 1637–1660. doi:10.13031/trans.58.10726

Bagstad, K. J., Villa, F., Batker, D., Harrison-Cox, J., Voigt, B., and Johnson, G. W. (2014). From Theoretical to Actual Ecosystem Services: Accounting for Beneficiaries and Spatial Flows to Map and Quantify Ecosystem Services. *Ecol. Soc.* 19 (2), 64. doi:10.5751/es-06523-190264

BBSDLP (2015), Sumberdaya Lahan Pertanian Indonesia: Luas, Penyebaran, Dan Potensi Ketersediaan. *Balai Besar Litbang Sumberdaya Lahan Pertanian. Badan Litbang Pertanian*. Jakarta: IAARD Press.

BRG (2019), Three Years of Peatland Restoration in Indonesia Report. *Badan Restorasi Gambut Republik Indonesia (The Peatland Restoration Agency of the Republic of Indonesia)*.

Cohen-Shacham, E., Dayan, T., de Groot, R., Beltrame, C., Guillet, F., and Feitelson, E. (2015). Using the Ecosystem Services Concept to Analyse Stakeholder Involvement in Wetland Management. *Wetlands Ecol. Manage.* 23, 241–256. doi:10.1007/s11273-014-9375-1

Comeau, L. P., Hergoualc'h, K., Smith, J. U., and Verchot, L. (2013), Conversion of Intact Peat Swamp forest to Oil palm Plantation: Effects on Soil CO_2 Fluxes in Jambi, Sumatra. *Working Paper 110*. Bogor, Indonesia: CIFOR.

Couwenberg, J., Dommain, R., and Joosten, H. (2010). Greenhouse Gas Fluxes from Tropical Peatlands in South-East Asia. *Glob. Chang. Biol.* 16, 1715–1732. doi:10.1111/j.1365-2486.2009.02016.x

Couwenberg, J., and Hooijer, A. (2013). Towards Robust Subsidence-Based Soil Carbon Emission Factors for Peat Soils in South-East Asia, with Special Reference to Oil palm Plantations. *Mires Peat* 12, 1–13. Article 01.

Deshmukh, C. S., Julius, D., Desai, A. R., Asyhari, A., Page, S. E., Nardi, N., et al. (2021). Conservation Slows Down Emission Increase from a Tropical Peatland in Indonesia. *Nat. Geosci.* 14, 484–490. doi:10.1038/s41561-021-00785-2

Dommain, R., Couwenberg, J., and Joosten, H. (2010). Hydrological Self-Regulation of Domed Peatlands in South-East Asia and Consequences for Conservation and Restoration. *Mires Peat* 6, 1–17. Article 05.

Ferre, M., and Martin-Ortega, J. (2019). *A User Guide for Valuing the Benefits of Peatland Restoration*. [report]. Leeds: University of Leeds.

Fisher, B., Turner, R. K., and Morling, P. (2009). Defining and Classifying Ecosystem Services for Decision Making. *Ecol. Econ.* 68, 643–653. doi:10.1016/j.ecolecon.2008.09.014

Gao, J., Holden, J., and Kirkby, M. (2016). The Impact of Land-Cover Change on Flood Peaks in Peatland Basins. *Water Resour. Res.* 52, 3477–3492. doi:10.1002/2015WR017667

Goldstein, J. H., Caldarone, G., Duarte, T. K., Ennaanay, D., Hannah, N., Mendoza, G., et al. (2012). Integrating Ecosystem-Service Trade-Off into Land-Use Decisions. Available at: http://www.pnas.org/cgi/doi/10.1073/pnas.1201040109.

Hapsari, K. A., Biagioni, S., Jennerjahn, T. C., Reimer, P. M., Saad, A., Achnopha, Y., et al. (2017). Environmental Dynamics and Carbon Accumulation Rate of a Tropical Peatland in Central Sumatra, Indonesia. *Quat. Sci. Rev.* 169, 173–187. doi:10.1016/j.quascirev.2017.05.026

Hergoualc'h, K., and Verchot, L. V. (2011). Stocks and Fluxes of Carbon Associated with Land Use Change in Southeast Asian Tropical Peatlands: A Review. *Glob. Biogeochem. Cycles* 25, a–n. doi:10.1029/2009GB003718

Hergoualc'h, K., and Verchot, L. V. (2014). Greenhouse Gas Emission Factors for Land Use and Land-Use Change in Southeast Asian Peatlands. *Mitig. Adapt. Strateg. Glob. Change* 19, 789–807.

Hirano, T., Segah, H., Kusin, K., Limin, S., Takahashi, H., and Osaki, M. (2012). Effects of Disturbances on the Carbon Balance of Tropical Peat Swamp Forests. *Glob. Change Biol.* 18, 3410–3422. doi:10.1111/j.1365-2486.2012.02793.x

Holden, J., Chapman, P. J., and Labadz, J. C. (2004). Artificial Drainage of Peatlands: Hydrological and Hydrochemical Process and Wetland Restoration. *Prog. Phys. Geogr. Earth Environ.* 28, 95–123. doi:10.1191/0309133304pp403ra

Hooijer, A., Page, S., Jauhiainen, J., Lee, W. A., Lu, X. X., Idris, A., et al. (2012). Subsidence and Carbon Loss in Drained Tropical Peatlands. *Biogeosciences* 9, 1053–1071. doi:10.5194/bg-9-1053-2012

Ikkala, L., Ronkanen, A.-K., Utriainen, O., Kløve, B., and Marttila, H. (2021). Peatland Subsidence Enhances Cultivated lowland Flood Risk. *Soil Tillage Res.* 212, 105078. doi:10.1016/j.still.2021.105078

Interagency Working Group on Social Cost of Carbon (2013). *Technical Support Document: Technical Update of the Social Cost of Carbon for Regulatory Impact Analysis*. United States Government.

Irawan, U. S., and Purwanto, E. (2020). *Pengukuran dan Pendugaan Cadangan Karbon pada Ekosistem Hutan Gambut dan Mineral, Studi Kasus di Hutan Rawa Gambut Pematang Gadung dan Hutan Lindung Sungai Lesan, Kalimantan. Yayasan Tropenbos Indonesia*. Bogor.

Jaenicke, J., Rieley, J. O., Mott, C., Kimman, P., and Siegert, F. (2008). Determination of the Amount of Carbon Stored in Indonesian Peatlands. *Geoderma* 147 (3Äì4), 151–158. doi:10.1016/j.geoderma.2008.08.008

Jauhiainen, J., Page, S, E., and Vasander, H. (2016). Greenhouse Gas Dynamics in Degraded and Restoredtropical Peatlands. *Mires Peat* 17, 1–12.

Arnold, J. G., Moriasi, D. N., Gassman, P. W., Abbaspour, K. C., White, M. J., Srinivasan, R., et al. (2012). SWAT: Model Use, Calibration, and Validation. *Trans. ASABE* 55 (4), 1491–1508. doi:10.13031/2013.42256

Johnson, K. A., Polasky, S., Nelson, E., and Pennington, D. (2012). Uncertainty in Ecosystem Services Valuation and Implications for Assessing Land Use Tradeoffs: An Agricultural Case Study in the Minnesota River Basin. *Ecol. Econ.* 79, 71–79. doi:10.1016/j.ecolecon.2012.04.020

Joosten, H., and Clarke, D. (2002). *Wise Use of Mires and Peatlands*. International Mire Conservation Group and International Peat Society.

Juutinen, A., Tolvanen, A., Saarimaa, M., Ojanen, P., Sarkkola, S., Ahtikoski, A., et al. (2020). Cost-effective Land-Use Options of Drained Peatlands- Integrated Biophysical-Economic Modeling Approach. *Ecol. Econ.* 175, 106704. doi:10.1016/j.ecolecon.2020.106704

Kelola Sendang Project (2018). *South Sumatra Partnership for Landscape Management Support Project (KELOLA Sendang): Annual Report 2018*.

Khasanah, N., van Noordwijk, M., and Ningsih, H. (20152015). Aboveground Carbon Stocks in Oil palm Plantations and the Threshold for Carbon-Neutral Vegetation Conversion on mineral Soils. *Cogent Environ. Sci.* 1, 1119964. doi:10.1080/23311843.2015.1119964

Kimmel, K., and Mander, Ü. (2010). Ecosystem Services of Peatlands: Implications for Restoration. *Prog. Phys. Geogr. Earth Environ.* 34 (4), 491–514. doi:10.1177/0309133310365595

Kotowska, M. M., Leuschner, C., Triadiati, T., Meriem, S., and Hertel, D. (2015). Quantifying above- and Below-Ground Biomass Carbon Loss with forest Conversion in Tropical Lowlands of Sumatra (Indonesia). *Glob. Change Biol.* 21, 3620–3634. doi:10.1111/gcb.12979

LPPM-IPB (2019). *Assessing Below-Ground Carbon Stocks of Peatlands in Merang, Buring and Kepahyang Watersheds*. Bogor, Indonesia: Institut Pertanian Bogor.

Macfadyen, S., Cunningham, S. A., Costamagna, A. C., and Schellhorn, N. A. (2012). Managing Ecosystem Services and Biodiversity Conservation in Agricultural Landscapes: Are the Solutions the Same? *J. Appl. Ecol.* 49, 690–694. doi:10.1111/j.1365-2664.2012.02132.x

Miettinen, J., Shi, C., and Liew, S. C. (2016). Land Cover Distribution in the Peatlands of Peninsular Malaysia, Sumatra and Borneo in 2015 with Changes since 1990. *Glob. Ecol. Conservation* 6, 67–78. doi:10.1016/j.gecco.2016.02.004

Moore, B. D., Kaur, G., Motavalli, P. P., Zurweller, B. A., and Svoma, B. M. (2018). Soil Greenhouse Gas Emissions from Agroforestry and Other Land Uses under Different Moisture Regimes in Lower Missouri River Floodplain Soils: A Laboratory Approach. *Agroforest Syst.* 92, 335–348. doi:10.1007/s10457-017-0083-8

Moore, S., Evans, C. D., Page, S. E., Garnett, M. H., Jones, T. G., Freeman, C., et al. (2013). Deep Instability of Deforested Tropical Peatlands Revealed by Fluvial Organic Carbon Fluxes. *Nature* 493, 660–663. doi:10.1038/nature11818

Neitsch, S. L., Arnold, J. G., Kiniry, J. R., and Williams, J. R. (2005). *Soil and Water Assessment Theoretical Documentation Version 2005 Grassland, Soil and Water Research Laboratory Agricultural Research Service and Blackland Research Center*. Texas Agricultural Experiment Station. http://swatmodel.tamu.edu/media/1292/SWAT2005theory.pdf.

Nordhaus, W. D. (2017). Revisiting the Social Cost of Carbon. *Proc. Natl. Acad. Sci. USA* 114 (7), 1518–1523. doi:10.1073/pnas.1609244114

Paavola, J., and Hubacek, K. (2013). Ecosystem Services, Governance, and Stakeholder Participation: An Introduction. *E&S* 18, 4–42. doi:10.5751/ES-06019-180442

Page, S. E., Rieley, J. O., and Banks, C. J. (2011). Global and Regional Importance of the Tropical Peatland Carbon Pool. *Glob. Change Biol.* 17, 798–818. doi:10.1111/j.1365-2486.2010.02279.x

Page, S., Hosciło, A., Wösten, H., Jauhiainen, J., Silvius, M., Rieley, J., et al. (2009). Restoration Ecology of lowland Tropical Peatlands in Southeast Asia: Current Knowledge and Future Research Directions. *Ecosystems* 12, 888–905. doi:10.1007/s10021-008-9216-2

P. Kareiva, H. Tallis, T. H. Ricketts, G. C. Daily, and S. Polasky (Editors) (2011). *Natural Capital: Theoryand Practice of Mapping Ecosystem Services* (New York: Oxford Univ. Press).

Rodríguez, J. P., Beard, T. D., Bennett, E. M., Cumming, G. S., Cork, S. J., Agard, J., et al. (2006). Trade-offs across Space, Time, and Ecosystem Services. *Ecol. Soc.* 11, 28. doi:10.5751/ES-01667-110128

Seibert, J., and McDonnell, J. J. (2002). "The Quest for an Improved Dialog between Modeler and Experimentalist,". *Calibration of Watershed Models*. Editors Q. Duan, H. V. Gupta, S. Sorooshian, A. N. Rousseau, and R. Turcotte (Washington, D.C.: American Geophysical Union), Vol. 6.

Sumarga, E., Hein, L., Edens, B., and Suwarno, A. (2015). Mapping Monetary Values of Ecosystem Services in Support of Developing Ecosystem Accounts. *Ecosystem Serv.* 12, 71–83. doi:10.1016/j.ecoser.2015.02.009

Sumarga, E., Hein, L., Hooijer, A., and Vernimmen, R. (2016). Hydrological and Economic Effects of Oil palm Cultivation in Indonesian Peatlands. *E&S* 21, 2–52. doi:10.5751/ES-08490-210252

Suwarno, A., Hein, L., and Sumarga, E. (2016). Who Benefits from Ecosystem Services? A Case Study for Central Kalimantan, Indonesia. *Environ. Manage.* 57, 331–344. doi:10.1007/s00267-015-0623-9

Takahashi, H., Shimada, S., Ibie, B. I., Usup, A., Asmara, Y., and Limin, S. H. (2002). "Annual Changes of Water Table Balance and Drought index in a Tropical Peat Swamp forest of Central Kalimantan, Indonesia," in *Peatlands for People: Natural Resource Functions and Sustainable Development*. Editors J. O. Rieley, S. E. Page, and B. Setiasi (Jakarta: BPPT and Indonesian Peat Association), 63–67.

Tallis, H., Kareiva, P., Marvier, M., and Chang, A. (2008). An Ecosystem Services Framework to Support Both Practical Conservation and Economic Development. *Proc. Natl. Acad. Sci. U S A.* 105 (45), 9457–9464. doi:10.1073/pnas.0705797105

Tallis, H., and Polasky, S. (2009). Mapping and Valuing Ecosystem Services as an Approach for Conservation and Natural-Resource Management. *Ann. N. Y. Acad. Sci.* 1162, 265–283. doi:10.1111/j.1749-6632.2009.04152.x

Tarigan, S. D., Sunarti, K., Wiegand, K., Dislich, C., Slamet, B., Heinonen, J., et al. (2016). Mitigation Options for Improving the Ecosystem Function of Water

Flow Regulation in a Watershed with Rapid Expansion of Oil palm Plantations. *Sustainability Water Qual. Ecol.* 8, 4–13. doi:10.1016/j.swaqe.2016.05.001

Tarigan, S., Stiegler, C., Wiegand, K., Knohl, A., and Murtilaksono, K. (2020). Relative Contribution of Evapotranspiration and Soil Compaction to the Fluctuation of Catchment Discharge: Case Study from a Plantation Landscape. *Hydrological Sci. J.* 65 (7), 1239–1248. doi:10.1080/02626667.2020.1739287

Tarigan, S., Wiegand, K., and Slamet, B. (2017). Minimum forest Cover Required for Sustainable Water Flow Regulation of a Watershed: A Case Study in Jambi Province, Indonesia. *Hydrol. Earth Syst. Sci.* 22, 581–594. doi:10.5194/hess-22-581-2018

Tilman, D., Cassman, K. G., Matson, P. A., Naylor, R., and Polasky, S. (2002). Agricultural Sustainability and Intensive Production Practices. *Nature* 418, 671–677. doi:10.1038/nature01014

Tol, R. S. J. (2019). Social Cost of Carbon for (almost) Every Country. *Working Paper Series 0219*. Brighton: Dept. of Economics, University of Sussex Business School. doi:10.1016/j.eneco.2019.07.006

Tonks, A. J., Aplin, P., Beriro, D. J., Cooper, H., Evers, S., Vane, C. H., et al. (2017). Impacts of Conversion of Tropical Peat Swamp forest to Oil palm Plantation on Peat Organic Chemistry, Physical Properties and Carbon Stocks. *Geoderma* 289, 36–45. doi:10.1016/j.geoderma.2016.11.018

Turetsky, M. R., Benscoter, B., Page, S., Rein, G., Van Der Werf, G. R., and Watts, A. (2015). Global Vulnerability of Peatlands to Fire and Carbon Loss. *Nat. Geosci* 8, 11–14. doi:10.1038/ngeo2325

Uda, S. K., Hein, L., and Sumarga, E. (2017). Towards Sustainable Management of Indonesian Tropical Peatlands. *Wetlands Ecol. Manage.* 25, 683–701. doi:10.1007/s11273-017-9544-0

UN Environment Programme (2019). *New UN Decade on Ecosystem Restoration to Inspire Bold UN Environment Assembly Decisions*. UN story. Available at: https://www.unep.org/news-and-stories/story/new-un-decade-ecosystem-restoration-inspire-bold-un-environment-assembly (Accessed November 14, 2021)

United States Environmental Protection Agency (2013). *The Social Cost of Carbon*. Washington, D.C., USA: United States Environmental Protection Agency. Available at: http://www.epa.gov/climatechange/EPAactivities/economics/scc.html.

White, M. J., Harmel, R. D., Arnold, J. G., and Williams, J. R. (2012). SWAT Check: A Screening Tool to Assist Users in the Identification of Potential Model Application Problems. *J. Environ. Qual.* 43 (1), 208–214. doi:10.2134/jeq2012.0039

Wösten, J. H. M., Clymans, E., Page, S. E., Rieley, J. O., and Limin, S. H. (2008). Peat-water Interrelationships in a Tropical Peatland Ecosystem in Southeast Asia. *Catena* 73 (2), 212–224. doi:10.1016/j.catena.2007.07.010

Yule, C. M. (2010). Loss of Biodiversity and Ecosystem Functioning in Indo-Malayan Peat Swamp Forests. *Biodivers. Conserv.* 19, 393–409. doi:10.1007/s10531-008-9510-5

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Revisiting the Green City Concept in the Tropical and Global South Cities Context: The Case of Indonesia

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Urban areas play a key role in reaching global sustainability as they produce a high amount of waste and emissions, consume a lot of resources, and perform as the prime mover of the global economy. Unsustainable urbanization will generate multidimensional impacts on the earth's socio-ecological system that is nearly impossible to be managed. As a preemptive action, urban sustainability has been considered as one of the most important targets in the Sustainable Development Goals. Within this context, the green city has emerged as a widely adopted concept around the globe. In the Global North, the concept of a green city has been incrementally developed as efforts to mainstream green infrastructure and nature-based solutions approaches in supporting ecosystem services. Quite the contrary, in the Global South cities, due to their rapid and vast urbanization process, the green city has been fragmentally adopted. Previous studies have proposed three factors, i.e., urbanization, biophysics, and governance, underlying the different approaches of green city development between cities in the Global North and South. Still, more studies are needed to explicate these factors and how these will in turn shape a particular green city trajectory in Global South cities. This study aims to respond to these questions based on green city experimentation in Indonesia. An exploratory study was done based on a literature review and participant observation. The results exhibit that the green city program in Indonesia largely focused on the development of green open spaces which has ultimately failed to improve the urban environment and sustainability. Incorporating local socio-ecological aspects coupled with integrated multi-level and multi-actors' governance is recommended to increase the green city performances.

Keywords: green city, geographical situatedness, global south, Indonesia, urban sustainability

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INTRODUCTION

Urban areas have become a key geographical area to address sustainability issues due to their massive ecological implications, i.e., waste and emission production and ecological footprint, that have not only affected in-situ environment but also the global environment (Acuto et al., 2018; Zinkernagel et al., 2018). On the other hand, urban areas also strongly interlinked with global socio-economic dynamics as almost 60% of the world's population live in these particular areas (UNDESA, 2018) and they function as the key locus of global economic growth (Acuto et al., 2018; World Bank, 2020). Less surprisingly, urban sustainability has become an increasingly prominent topic of discussion, including in the global policy discourse where urban areas or cities are part of the United Nations Sustainable Development Goals (UN SDGs).

Amid the rising prominence of urban sustainability agendas, the green city has come to the fore as one of the most widely adopted concepts across the globe, both in the Global North and South. At the outset of green city's experimentation, more emphasis was given to minimizing environmental impact by increasing efficient use of energy, water, and land as well as reducing waste and emissions (EIU, 2012; Liendfield and Steinberg, 2012; Venkatesh, 2013).

Along with the rising global challenges, especially within the context of climate change, the socio-ecological approach has gained increased traction in the current iteration of green city's adoption. Natural processes (i.e., water, nutrient, energy, and carbon cycles) have been more carefully considered to derive ecosystem services (ES) for supporting human well-being (Elmqvist and Mcdonald, 2013). In particular, green infrastructure (GI) and nature-based solutions (NBS) have become two major approaches to operationalize ES in cities as both approaches focus on the integration between nature and artificial-human systems (Nesshöver et al., 2017; Pauleit et al., 2017).

As the intellectual origin of the green city concept, GI and NBS approaches have been developed rapidly in Global North cities. Within this particular geographical setting, developing multifunctional green spaces and their integration with grey infrastructure are increasingly required as the future cities are planned to be more compact (Hansen et al., 2019). As part of this, the development of technology is viewed to be crucial in further enhancing GI and NBS approaches. Currently, the "mixed" concept of smart-compact-green cities has emerged as a renewed initiative of green cities to achieve urban sustainability in an increasingly complex environment (Artmann et al., 2017; Richter and Behnisch, 2018).

Cities in the Global South, however, are still falling behind in implementing this newest iteration of the green city concept, especially in adopting ES, GI, and NBS in urban planning. Some researchers have conducted comparative studies of green city adoption, particularly between cities in Asia and other developing countries and cities in the Global North. Shen and Fitriaty (2018), for instance, reveal that green cities in Asia mainly focus on developing green buildings, green technology, and green open spaces without necessarily linking these initiatives to the existing urban sprawl issue. Meanwhile, the combination of green and smart cities has been partly developed in cities in Asian developed countries such as China, Japan, Korea, and Singapore. In Egypt and India, the green city was recently promoted as a new approach that is linked to urban planning (Pankaja and Nagendra, 2015; El Ghorab and Shalaby, 2016).

While previous studies have highlighted the gap in green city development and implementation between cities in the Global North and South, limited attention has been given to the factors causing this gap. In this study, our primary attention is green city experimentation in the Global South. Shen and Fitriaty (2018) and Lechner et al. (2020), in particular, have proposed three underlying factors that may strongly affect the trajectory of green city in the Global South, i.e., urbanization, biophysics, and governance. The urbanization factor highlights the issue of urban sprawl and high population density, whereas the biophysics factor focuses on the issue of climate change and its relations with the local climate condition. Meanwhile, the governance factor mainly points out the issue of fragmented governance and the lack of public participation.

Still, how these factors have affected the adoption and application of green cities remain uncovered. Further empirical investigations are therefore needed. Against this backdrop, the main objective of this paper is to explore the adoption of the green city concept in Indonesia over the last decade through the lens of urbanization, biophysics, and governance aspects. The Indonesian government formally launched the "Green City Development Program" or *Program Pengembangan Kota Hijau* (P2KH) in 2011. This program ran until 2020 with a total of 174 participating localities by the end of the program. This paper address two questions in particular: How has the green city concept been adopted in Indonesia by means of the P2KH program? And, based on this assessment and considering the context specifics of Indonesia, how should the green city concept be further developed? A thorough assessment was done to gauge the effectiveness of the program as well as to gain some valuable lessons learned from the program in achieving urban sustainability goals. This step is pivotal to defining a more refined trajectory of green cities that is attuned to the localized political and socio-ecological dynamics. This study can offer a reference for green city development in the Global South that has more or less similar characteristics to Indonesia.

GREEN CITY CONCEPTION AND ITS EVOLUTION

The origin of the green city concept can be traced back to the early 20th century, particularly rooted in the existence of urban subsistence gardens in the United States (US) metropolitan areas in the 1900s (Moore, 2006). These gardens, however, were not regarded as part of urban landscapes, but as a temporary crisis response during World War I and the Great Depression period (Kahn, 1982; Warner, 1987). Within this period, urban inhabitants were allowed to farm to secure food supply. The history of urban gardens seems to have been forgotten after the 1940s when the situation had relatively stabilized. As a result, these gardens gradually dissolved in the urban environment as they were viewed as a part of more rural landscapes (Moore, 2006).

The green city concept has revived and gained a new momentum, particularly after the unfolding of sustainable development ideas as declared by the World Commission on Environment and Development in 1987 (El Ghorab and Shalaby, 2016). The rapid pace of industrialization and urbanization has, in particular, been regarded as two major sources of global unsustainability (Lengeweg et al., 2000; Li and Lin, 2015). Therefore, various notions, notably, green city, eco-city, and livable city, have appeared as intertwining concepts to mainstream sustainable development in urban areas (Liendfield and Steinberg, 2012; Pace et al., 2016). While eco-city focuses on urban ecological health and livable city emphasizes urbanites' well-being, green city pays particular attention to the relationship between the environment and human systems within the urban context.

There are at least two main factors underlying the conceptualization of a green city. First, from the consumption perspective, the concern is to reduce the demand for natural resources and services as well as waste and emission production. Second, from the production perspective, the main attention is to invest in the improvement of nature and the environment to support a better quality of life. Combining these two perspectives, the green city concept is mainly concerned with the planning and management of green resources and materials, green community, green open spaces, green waste, green transportation, and green building in an integrated manner (Liendfield and Steinberg, 2012). However, some iterations and modifications occurred during the concept implementation, adjusting to the changes of urban issues and challenges as well as to the development of human knowledge and technology.

As the green city concept was originally produced and developed in and from European and US cities, its transformation can be clearly traced in those cities. Increasing economic prosperity in the Global North has been followed by people's increasing awareness concerning quality of life given the rising issues of environmental degradation and climate change. This has then led to a stronger commitment to developing new ways to reduce waste and emission production, generate clean energy and clean water, and increase energy and water use efficiency. Furthermore, the green city concept has been also incorporated into urban planning through developing compact city scenarios so as to reduce human mobility. The notion of a compact city implies the efforts to support efficient mobility, energy use, and reduced emission production (Artmann et al., 2017; Richter and Behnisch, 2018). However, these approaches were regarded as insufficient as recent research findings reveal that comprehending and restoring natural processes (i.e., the cycles of water, carbon, nutrient, and energy) are crucial to address environmental and climate change issues (Schwarz et al., 2017). Accordingly, the green city concept has experienced a paradigm shift. In this newest iteration, humans and their products should live in balance with nature, including all living organisms, and their habitat (Breuste et al., 2020).

The term ecosystem services (ES) has emerged as a key lens that is considered to be able to bridge the human and natural systems (MEA, 2005). Various concepts have been developed to operationalize ecosystem services in cities, in which two concepts in particular, i.e., green infrastructure (GI) and nature-based solutions (NBS), have probably become the most popular ones (Pauleit et al., 2017). GI has been integrated into urban planning through developing green hubs, ecological corridors, restored habitats, and artificial features to assist and enhance natural processes, buffer zones, and multifunctional zones (EC, 2013). Meanwhile NBS, although it is still new and underdeveloped, focuses on mainstreaming nature to deal with the environment and climate change issues by restoring or mimicking the natural process in cities (Pauleit et al., 2017). In these approaches, mainstreaming biodiversity conservation has been encouraged as living organisms are viewed to be essential in supporting and facilitating natural processes (Aronson et al., 2017; Xie and Bulkeley, 2020). Both approaches need a transdisciplinary and multistakeholder approach, focusing on landscape, biodiversity, and human well-being, and integration with urban agendas and urban-grey infrastructures. Meanwhile, environmental justice has been also taken into consideration as ecosystem services are deemed to be a part of public services that should benefit all groups of citizens at an equal basis (Derkzen et al., 2017; Garcia-Lamarca et al., 2021).

The role of technology to support multifunctional GI or NBS has also become increasingly substantial (Hansen et al., 2019). Various forms of information and communication technology (ICT) have been widely used to monitor environmental quality, including air quality, water quality, water discharge, transportation, and waste disposal. Some other technologies, including solar cell electricity, bio-fuel, and bio-energy, have also been deployed to produce clean energy. In terms of green building, some innovations in modern architecture have been developed to increase energy and water use efficiency as well as to reduce waste production. As a result, the intertwined notions of smart-compact-green cities have been promoted as a renewed concept to reach urban sustainability in the Global North (Artmann et al., 2017).

Meanwhile, the adoption of green cities in the Global South seems to be more challenging due to different socio-ecological contexts. For instance, unlike urban areas in the Global North that have developed in a gradual fashion, the urbanization process in the Global South has occurred at a much faster rate within a much shorter period of time. This urban expansion has been mainly driven by private and foreign investment, intended to propel economic development (Dahiya, 2012). On the other hand, government authorities, in most cases, have a lack of control over the implementation of spatial planning. As a result, urban physical expansion has sprawled in all directions without a clear pattern, encroaching the city's surrounding rural areas (Estoque and Murayama, 2015; Pribadi and Pauleit, 2015). This sprawling process has been accelerated by the increasing land prices coupled with declining environmental quality in the urban core, thus pushing people to live in the outskirts. As a result, commuting has become a common phenomenon that leads to a longer time of travel, thus producing emissions that continue to increase over the years.

Within this Global South cities setting, efforts to develop green open spaces, manage waste and emissions, and reduce mobility become more complex (Shen and Fitriaty, 2018). Extending green open spaces has been hampered by land scarcity and rising land prices. Waste and emission have been badly managed due to high population numbers and high density that leads to congestion and waste disposal issues. In the same token, reducing mobility has also been difficult to implement as urban sprawl continues to facilitate commuting behavior. On the other hand, issues concerning conserving biodiversity and increasing human well-being are still below the radar. It is therefore unsurprising that the new approaches such ES, GI, and NBS have not been adopted by Global South cities, except for some cities in China (Hardiman, 2020).

It is also worth noting that in the case of developing countries located in the tropical zones, the application of the concepts such as ES, GI, and NBS can become more complicated since these countries have a rich biodiversity and heterogeneous habitat. In addition to this, cities situated in these zones are also highly vulnerable to climate change, especially in terms of sea-level rise, rainstorms, flooding, landslides, and drought (Lechner et al., 2020). Coupled with uncontrolled sprawling cities and limited capacity of governance, pressures caused by climate change will put a lot of urban inhabitants at risk.

While the specific settings of Global South cities as discussed above call for a different path towards green cities, another challenge that may occur relates to the environmental visioning of green cities. As has been criticized by Garcia-Lamarca et al. (2021), the green city label in several cities, mainly in the Global North, has been also deployed with the intention to build a city's brand that can attract global actors and investment vis-à-vis improving the socio-ecological quality of a city. Such orientation can facilitate market-led greening that may result in widening socio-spatial inequalities. This "green boosterism" also applies to several Asian cities (Leducq and Scarwell, 2020), especially in Asian developed countries, where smart-green cities have been emphasized (Shen and Fitriaty, 2018) regardless of the continuation of urban sprawl, and its adverse socio-ecological impact.

Taken together, there is a need to arrive at a better understanding of the unique socio-ecological issues in the Global South cities that co-shape the different trajectories of green cities. In this paper, particular emphasis is given to uncovering three distinguished aspects: urbanization, biophysics, and governance (Lechner et al., 2020). In our view, the variegated geographical contexts should be of concern, as the failure to comprehend and manage these would lead to a failure to fully benefit from the adoption of the green city concept.

THE POLICY CONTEXT OF INDONESIA'S URBAN SUSTAINABILITY AGENDA

Indonesia's rapid economic growth combined with rapid urbanization has led to pressures on the environment (Jafari et al., 2012; Ahmed et al., 2019). Indonesia is the third largest country in Asia in terms of population and is one of the world's biggest polluters (Jambeck et al., 2015; Tarigan and Sagala, 2018). Coping with ever rising environmental challenges and being part of international communities, Indonesia has adopted a number of key global environmental agendas into its national policy circles. In this section, our intention is not to provide an exhaustive review of the global environmental (or more broadly, sustainability) agenda. Rather, we aim to show what we view as some representative examples of these agendas and how the Indonesian government has localized such agendas accordingly.

The first worldwide event that became an important milestone for environmental mainstreaming would be the 1992 Earth Summit that was held in Rio de Janeiro, Brazil. This summit was held on the occasion of the 20th anniversary of the first United Nations (UN) Human Environment Conference in Stockholm, Sweden, in 1972. Taking part in this Earth Summit, Indonesia was among the participating countries that adopted Agenda 21, a comprehensive plan of action to build a global partnership for sustainable development in the 21st century (KLH, 1997). Agenda 21 was one of the major achievements that resulted from the Earth Summit besides, among others, the United Nations Framework Convention on Climate Change (UNFCCC), the Convention on Biological Diversity, and the Declaration on the Principles of Forest Management. Following up its commitment toward this global agenda, in 1997, and Indonesia published the National Agenda 21 containing directions to insert sustainable development principles into the national development planning (KLH, 2002). This document was later followed by the Sectoral Agenda 21 which was published in 2000 (KLH, 2002). This sectoral agenda document outlines a more detailed direction for different sectors, including mining, energy, human settlement, tourism, and forestry.

Indonesia's participation continues to persist to the present-day UN SDGs. Having experience in adopting the previous global agenda, including Agenda 21 and Millennium Development Goals (MDGs), the Indonesian government has elaborated the seventeen goals of SDGs in the development planning documents through the issuance of the Presidential Regulation (*Peraturan Presiden*) 59/2017 on the Implementation of SDGs' Achievement. As mandated by this regulation, Indonesia has also published the Action Plan of SDGs at the national and provincial levels (*Rencana Aksi Nasional/Daerah Tujuan Pembangunan Berkelanjutan*).

The country's achievement of SDGs at the Southeast Asian level was, however, still relatively below the region's average overall score (Alisjahbana and Murniningtyas, 2018). Among the total seventeen goals, Indonesia had a higher score vis-à-vis the region for eight goals (**Table 1**). Meanwhile, the 11th SDG that specifically relates to urban development, i.e., "Sustainable Cities and Communities", was among those that had lower scores than the region's value. Despite not being explicitly mentioned in any government publications, some experts emphasized that sustainable cities should be among the goals of SDGs that are put as a top priority in the Indonesian context (Sachs, 2015; Alisjahbana and Murniningtyas, 2018).

In addition to SDGs, another key global environmental agenda adopted by Indonesia is the 2015 Paris Climate Agreement. The Paris Climate Agreement is the most recent agreement within the framework of the UNFCCC. One of the main aims of the agreement is to maintain the increase in the global average temperature to well below 2°C above the pre-industrial levels. Based on this, Indonesia has ratified the Paris Agreement in New York in 2016 and committed to reducing its greenhouse gas (GHG) emissions target by 29% against a 2030 business-as-usual (BAU) scenario and by up to 41% subject to international assistance for finance, technology transfer, and capacity building (Masripatin et al., 2017). To this end, five sectors have been set as priority areas for reducing this GHG emissions target, i.e., forestry and peatlands, agriculture,

TABLE 1 | Score of SDGs of Indonesia in 2015.

SDGs		Indonesia	Southeast asia (average)
1	No Poverty	76.87	83.70
2	Zero Hunger	44.82	47.10
3	Good Health and Well-being	53.79	61.24
4	Quality Education	73.83	70.06
5	Gender Equality	62.09	55.76
6	Clean Water and Sanitation	79.10	80.97
7	Affordable and Clean Energy	61.78	58.49
8	Decent Work and Economic Growth	63.71	59.55
9	Industry, Innovation, and Infrastructure	21.52	24.27
10	Reduced Inequalities	66.18	68.64
11	Sustainable Cities and Communities	48.80	56.28
12	Responsible Consumption and Production	46.41	39.86
13	Climate Action	83.96	69.84
14	Life Below Water	42.80	37.29
15	Life on Land	33.65	46.22
16	Peace, Justice, and Strong Institutions	59.51	56.37
17	Partnerships for the Goals	8.57	21.18
All SDGs		**54.40**	**54.60**

Source: Alisjahbana and Murniningtyas (2018).
The bold values (54.40 and 54.60) represent the total scores of SDGs of Indonesia (54.40) and Southeast Asian countries (54.60).

energy and transportation, industry, and waste. Cities and urban areas are not specifically mentioned here, but, to a limited extent, are partially inserted in the energy, and transportation as well as waste sectors (Masripatin et al., 2017; Wijaya et al., 2017).

Another sustainability agenda or concept brought from elsewhere into the Indonesian policy context is strategic environmental assessment (SEA). Historically, it came into existence in western countries, particularly in 1969 when the Congress of the US adopted the National Environmental Policy Act (NEPA). Later in 1985, this system of assessment was introduced in Europe with the implementation of the Environmental Impact Assessment (EIA) Directive (85/337/EEC) to a wide range of public and private projects. SEA has been used as a means to integrate environmental and social consideration into policy, plan, and program making. In Indonesia, SEA has been formally adopted through Law (*Undang-Undang*) 32/2009 on Environment Protection and Management. This umbrella regulation was later clarified through the issuance of Government Regulation (*Peraturan Pemerintah*) 46/2016 on the Procedure for the Implementation of SEA and of Ministry of Environmental Affairs and Forestry Affairs Decree (*Peraturan Menteri*) 69/2017 on the Implementation of Government Regulation 46/2016. SEA has become a mandatory assessment tool for all strategic public policies at the national, provincial, and local (*kota* and *kabupaten*) levels. SEA provisions are thus intended to overcome the void that policies, plans, and/or programs in Indonesia have mostly tended to ignore sustainable development principles (Salim and Hudalah, 2020). In this context, SEA can contribute to achieving a more sustainable urban development by means of assessing key local planning documents, notably local spatial planning (both general [RTRW] and detailed plan [RDTR]), local long-term development planning (RPJPD), and local mid-term development planning (RPJMD).

Meanwhile, in terms of urban policy, the New Urban Agenda (NUA) is deemed to be the most recent global framework promulgated by the United Nations Human Settlements Program (UN-HABITAT) to achieve a better and more sustainable future for cities across the globe. The NUA was first adopted in Quito, Ecuador, on 20 October 2016 and works as an accelerator of the SDGs, particularly the eleventh goal. Indonesia has also played an important part in developing the NUA. On 25–27 July 2016, Surabaya, Indonesia's second largest city after Jakarta, hosted the Third Preparatory Committee meeting for drafting the NUA (Salim and Hudalah, 2020). The NUA mandates that all countries must have a national urban policy. To date, there is no such policy in Indonesia. The draft of the so-called National Urban Area Development Policy and Strategy (*Kebijakan dan Strategi Pengembangan Kawasan Perkotaan Nasional* or KSPPN)' has been in progress within the Ministry of National Development Planning (BAPPENAS) for years but has not yet been officially published. The Ministry of Public Works and Housing (PUPR), on the other hand, has published a partial interpretation of the NUA for the Indonesian context (Sarosa et al., 2017). According to this publication, five areas are prioritized, i.e., access to clean water, percentage of slums, access to proper sanitation, green open space, and preservation of heritage areas.

In the absence of a national urban policy, Indonesia's central government has approached urban development through a variety of programs. One of the most pertinent programs that has a strong environmental dimension would be the "Green City Development Program" or *Program Pengembangan Kota Hijau* (P2KH). This program can be cast as an example of global policy mobility of urban ideas and planning practices in the Global South (Leducq and Scarwell, 2020). In Indonesia, the green city label was translated as a program that was anchored on the specific mandate outlined in Law 26/2007 on Spatial Planning, i.e., the obligation to provide green open space to at least 30% of a

TABLE 2 | Number of participating localities of P2KH.

Batch	Year	Number of participating localities (accumulation)
1	2013	60
2	2014	85
3	2015	112
4	2016	143
5	2017	165
6	2018	174

city's total area. Based on Article 29 of the spatial planning law, the existence of green open space is expected to provide useful ecosystem services to the city (Zain and Kencana, 2010; Joga and Ismaun, 2011).

P2KH kick-started in 2011, initiated by the Directorate General of Spatial Planning, Ministry of Public Works (PU) (Kirmanto et al., 2014). In the first year, 60 municipalities (*kota*) and regencies (*kabupaten*) participated in the program. The number of participants increased continuously. With the government institution restructuring that took place in 2015, the Directorate General of Spatial Planning was transferred to the Ministry of Agrarian Affairs and Spatial Planning (ATR). As an immediate result, P2KH was moved to the Directorate General of Human Settlements, Ministry of Public Works and Housing (PUPR). PUPR (then PU) provided financial and technical assistance to each of the participating localities for four consecutive years. This PUPR-initiated program ended in 2020 with a total of 174 participating localities (Schwarz et al., 2017) (**Table 2**). After the end of the central government's facilitation, it was expected that this initiative could be sustained by the participating local governments using their own resources. It should be noted that the P2KH program also included the development of new botanical gardens (kebun raya) in a small number of localities. However, given its minor role within the overall P2KH program, we do not discuss this further in our paper.

Inspired by green city measurements developed elsewhere, P2KH adopted eight indicators: 1) green planning and design, 2) green open space, 3) green waste, 4) green building, 5) green transportation, 6) green energy, 7) green water, and 8) green community (**Table 3**). Among the eight indicators, green planning and design is the most important one. This indicator is a mandatory indicator for localities to be selected for P2KH: the "commitment" of the local government to revise the local spatial plan (RTRW) by allocating at least 30% of a city's total area for green open space (Ministry of Public Works, 2013a). However, from 2016 to 2020 this requirement was tightened by obligating "an already revised" RTRW within which 30% of a city's total area is already allocated for green open space. Following this RTRW-related requirement, the local government was later required to prepare a green city master plan and to sign an action plan agreement to implement the P2KH program (Ministry of Public Works, 2013b; Kirmanto et al., 2014).

The other important indicators are green open space and green community. In fact, the gist of P2KH is the development of a park (green open space). Meanwhile, the green community indicator refers to the establishment of a green community forum or *forum komunitas hijau* (FKH) in each participating locality to raise public awareness toward green city development. Citizens were expected to support P2KH by maintaining and utilizing the pilot park in various ways.

The other remaining five indicators were not compulsory and were, therefore, executed differently across different localities. Indeed, not all localities have fully implemented these five indicators. It is important to note that all of these indicators were executed in a rather narrow manner, in that the development of green waste, green energy, green building, green transportation, and green water took place inside and around the pilot park, rather than having a city-wide perspective.

METHODS

This study is an exploratory study as it does not intend to provide conclusive results, but rather to observe a specific research problem that will help us to have a better understanding of this problem (Swedberg, 2020). Exploratory research tends to focus on a specific phenomenon that few or no previous studies have focused on (Brown, 2006; Swedberg, 2020). As such, this type of research often forms the basis for more conclusive studies.

The fact that the application of green cities in the context of cities in the Global South has received marginal policy and scholarly attention would make this topic fit with the nature of exploratory research. In this study, we focus on the green city development in Indonesia. As noted earlier, the green city program (P2KH) was the most obvious nationwide program run under the urban sustainability framework. Instead of being a single model that can travel seamlessly across different cities worldwide (see Leducq and

TABLE 3 | Indicators of P2KH.

No	Indicator	Description
1	Green planning and design	Commitment to revise RTRW (2011–2015); revised RTRW (2016–2020); preparing a green city master plan and signing an action plan agreement
2	Green open space	Building a pilot park in each participating locality
3	Green community	Establishing FKH in each participating locality
4	Green waste	Providing trash bins and sorting systems (organic and non-organic waste) in the pilot park
5	Green energy	Building solar cell-based lights in the pilot park
6	Green building	Constructing gazebos and park benches that have low emissions such as those made of non-metallic materials or wood in the pilot park
7	Green transportation	Providing bicycle and pedestrian paths around the park
8	Green waste	Building parks that have water elements, such as ponds or fountains

Scarwell, 2020), we considered that green cities operate, and are embedded in a specific geographical context. Within such geographical situatedness, various components have interwoven and have altogether formed a set of particular opportunities and challenges for green cities to develop. To this end, three lenses proposed by Lechner et al. (2020) were used to gain better insights into the context-specifics underpinning green city development in Indonesia.

The data and information required for the analysis using the framework mentioned above were gathered from primary and secondary sources. Primary data were collected mainly through participant observation (Laurier, 2010) that was conducted by the first author from 2011 to 2015. Participant observation is viewed as part of ethnographic methods. Participant observation has been used in qualitative research as a tool for collecting data about people, processes, and other social settings where the researcher is immersed in the day-to-day activities of people under study. The overall objective of this method is to develop a holistic understanding of the phenomenon under study (DeWalt and DeWalt, 2002). While this method has a number of advantages, it also shares some disadvantages, *inter alia* different researchers can have different interpretations about what they observe because of the key informants chosen and researcher's individual interest in a social setting (DeMunck and Sobo, 1998; DeWalt and DeWalt, 2002). To reduce such bias, secondary sources were used to triangulate the findings based on the primary data. This secondary data were collected from relevant policy documents, statistical reports, official documents, and relevant studies.

According to Gold (1958), there are four stances of "observer" in participant observation. Based on this classification, the first author acted as "the observer as participant", implying that the researcher observed and interacted closely enough with members, without participating in the core activities of the members (see Adler and Adler, 1994: 380). Within the context of our paper, the first author was involved in P2KH as part of the expert team hired by PUPR without directly influencing the core policymaking process. The researcher thus had the opportunity to interact with varying actors and parties at the national and local levels on different occasions (see **Supplementary Appendix S1** for the detailed activities of the researcher).

To keep up with the progress of green cities' development within the remaining period of P2KH implementation (2016–2020), some informal interviews were carried out with representatives of PUPR directly dealing with the implementation of the program. Meanwhile, in this case, the secondary sources were also used in concurrence not only to complement, but also to triangulate our findings based on the primary data. It should be noted here that there has been no significant change in the way the P2KH program was deployed between these two periods.

GREEN CITIES DEVELOPMENT IN INDONESIA

Urban Characteristics

As widely noted, rapidly increasing rates of population growth and urbanization in many developing countries have presented clear and pressing threats to sustainability (e.g., Hardoy et al., 2001). Such phenomena are also clearly evident in Indonesia, particularly in the context of the country's major cities and metropolitan areas. Indeed, the Indonesian urban system is characterized by the domination of the country's largest metropolises with the Jakarta metropolitan area (or Jabodetabek) sitting atop the urban hierarchy (Indraprahasta and Derudder, 2019). In this section our aim is not to capture all urban typologies in Indonesia, but rather focus on those largest urban areas where threats to environmental sustainability are more apparent.

One clear environmental implication of rapid urbanization and sprawling process has been uncontrolled land use conversion, where large tracts of conservation and agricultural areas have been converted (Firman, 2002; Pribadi and Pauleit, 2015). For instance, between 1972 and 2012, the percentage of forestland decreased from 34.4% to 10.1% in Jabodetabek (Pribadi and Pauleit, 2015). Meanwhile, within the same period, urban land use increased from 9,373 ha to 223,953 ha, with an average annual growth rate of 8.2%. Similar evidence can also be seen in the country's second largest urban agglomeration: the Surabaya metropolitan area. The percentage of built-up areas in Surabaya city and its two adjacent regions, i.e., Gresik and Sidoarjo, doubled between 1994 and 2012 (Katherina and Indraprahasta, 2019). In the case of Surabaya city, the urban land use increased from 43.2% in 1994 to 52.8% in 2003 and to 72.8% in 2012. One of the major urban characteristics resulting from this sprawling process has been the blurring of urban-rural boundaries, leading to the emergence of *desakota* areas. This term was coined by McGee (1991) to depict a densely populated rural area with more urban-like characteristics, both in terms of built environment and socio-economic landscapes.

The expansion of urban agglomeration, to a significant degree, and has been facilitated by the development of transportation networks connecting the urban core to its surrounding areas. In Jabodetabek, despite continuous improvement of the public transportation system, people daily commute between their homes and workplaces, largely by using private vehicles (BPS, 2019). Similar phenomenon also occurs in other metropolitan areas in Indonesia, including Bandung metropolitan area (Supriyatin et al., 2020). The excessive use of private vehicles in Indonesian urban agglomerations has contributed to the increase of GHG emissions production, thus deteriorating the urban environment. In the case of Jakarta city, about 70% of the city's air pollution is contributed by the emissions produced by fossil-fuel-generated cars and motorcycles (The Jakarta Post, 2018). Based on the World Air Quality Report, Jakarta is the most polluted capital city in Southeast Asia and the fifth most polluted capital city in the world in 2019 (IQAir, 2019). The air pollution level in Jakarta's surrounding regions, such as South Tangerang city and Bekasi city, is even worse than Jakarta. The 2020–2021 pandemic condition has indeed improved the air quality in the Indonesian capital due to some measures to restrict human mobility. However, no significant efforts have been made to change the trajectory of private vehicle reliance in the Jabodetabek area, let alone in Indonesia's other major metropolises. While this particular issue has caused serious environmental and health risks, it has been overlooked by the P2KH program that largely focused on green open space provision.

Given the scale and pace of urbanization, the core cities of many metropolitan areas have increasingly become a place filled with concrete, asphalt, and cement, leaving little room for green open space. Cities such as Jakarta and Medan, for instance, only have less than 10% green open space of their total area vis-à-vis 30% green open space as mandated by the spatial planning law. In general, little progress has been made to alter the fate of urban green open space. As many plots of land are already owned by individuals, the most likely option is to optimize government-owned land for green open space. This option has been also pursued by the P2KH program, although there has been no evaluation of how and to what extent this program has significantly altered the green open space provision in Indonesian cities. Some anecdotal examples, including in the case of Surabaya, have shown the importance of engaging non-government actors, including citizens, and to help increase the area for green open space. On the other hand, to demand a greater role for private actors in this matter has been a challenging task as these actors tend to capitalize their plots of land in the midst of an increasingly competitive land market, resulting in rising land prices in the urban core (Kenichiro, 2015; Leitner and Sheppard, 2018). So, while some real estate companies have developed residential complexes for middle- and upper-classes with significant areas allocated for greenery, this practice tends to encourage "green gentrification" within the city's context.

Biophysics

One of the key elements of green cities dealing with urban ecological issues is green open space. In the Indonesian context, the green city program actually originated from a mandate to provide green open space at least 30% of a city's total area in every municipality or regency. It was expected that green open space could play aesthetic and ecological functions to maintain the urban ecosystem. Both functions are important to support human well-being through the delivery of ecosystem services.

Green open space should be thoroughly planned and designed considering the city's structure and demand for ecosystem services, such as the provision of fresh water, clean air, comfort, and temperature. Every city faces different issues and challenges that should be addressed. On the other hand, each city also possesses different local ecological conditions. In such a situation, interaction between biotic and abiotic elements plays a key role since the provision of ecosystem services depends on the optimized circulation of water, energy, carbon, and nutrients. Therefore, designing green open spaces should be local site-specific where the utilization of native plant species should be enhanced as they have been long adapted to the local environment. Furthermore, maintaining green spaces should be done regularly and properly as plants' health is crucial to support environmental health.

Unfortunately, the green city program or P2KH in Indonesia was merely focused on the compliance of the minimum percentage of green open space. Instead of making integrative planning and design, the insertion of green open space into the local spatial plan was simply executed by incorporating remaining green open spaces and adding land parcels owned by the local government that could be transformed into new green open spaces. Government land was chosen to ensure the long-term maintenance of green spaces as well as to avoid land conversion. Consequently, the need for green open spaces and ecosystem services, which was local-site specific, has been less considered. The development of new green open spaces was simply done based on the aesthetic consideration as well as the proximity of government land to human settlements in order to support citizens' outdoor activities.

In practice, the P2KH program only supported the development of a single urban park as a pilot project on the local government's land. It was expected that the project could stimulate the local government to develop the remaining planned green open spaces that were already included in the local spatial plan. However, this scenario did not go as expected due to a lack of commitment to fund this program after the support from the national program was over. Certainly, green open space which was developed by the P2KH program was insufficient to support the improvement of the urban ecosystem.

As a result, there is a wide gap between P2KH and the necessities to build green open spaces that are able to enhance ecosystem services. By this, we mean not only the quantity, but also the quality of green spaces, particularly related to the capability of green spaces to support urban environmental health and sustainability. In this sense, the variety of bio-physic characteristics was barely considered.

Indonesia is a country with a very high diversity of tropical landscapes consisting of thousands of islands under different microclimate conditions and natural landscape compositions. The varied size of islands, landforms, and altitudes have given places for rich biodiversity. On the other hand, cities in Indonesia have different biophysical conditions as they are located in different geographical typologies, ranging from mountains, hills, valleys, plain areas, lowland, and coastal areas. These cities also have different sizes, ranging from small cities to mega-urban regions. This implies that the biophysical challenges vary between cities.

Still, as a tropical country located on the equator, Indonesia is characterized by heavy rainfall. Coupled with the sea level rise caused by global climate change, metropolitans, mostly located in the lowland, and coastal areas, are increasingly prone to flood. Meanwhile, cities with smaller sizes located in the upper areas are prone to landslides. Some cities in other areas are also disturbed by smoke haze caused by peatland and forest fires from their surrounding hinterlands, mainly during the dry season. Increasing temperature, pollutants, waste, and noise have also become common problems that have altogether threatened urban environmental health.

In order to cope with these issues, building urban green spaces capable of optimizing ecosystem services becomes more complicated. Restoring environmental conditions is often problematic due to the existence of various ecosystem types inhabited by rich biodiversity. It has become a great challenge when the GI and/or NBS approach as currently developed by green cities in the Global North would be fully adopted in the Indonesian green city program.

Based on this biophysical perspective, just copying the successful story of green cities in the Global North could thus be misleading. More efforts are needed to build functional green open spaces given the country's more complex environmental system. On the other hand, the lack of knowledge, as well as financial and human resources, have also hampered these efforts. A priority setting can be proposed in developing green open space, particularly focusing on the main and common problem such as water management. As argued by some (Kooy et al., 2019; Lechner et al., 2020), water management is pivotal for reaching sustainability in South and Southeast Asian cities which are affected by the monsoon climate.

Policy and Governance Context

There are several elements promoted by the green city concepts within the policy and governance context. The first and foremost element is the presence of pro-environmental policy. As has been further emphasized, the implementation of such policy is strongly encouraged to be accompanied by knowledge co-creation as well as participatory and collaborative approaches. However, the form and degree of government support for the environmental aspect and how this pro-environmental policy is implemented are influenced by, among other things, the structure of governance, government vision, political landscape, and leadership style in each country.

In Indonesia, despite the country's decentralizing and democratizing system, the central government still plays a pivotal part in the development process that, to a certain extent, relates to the country's developmentalist root (Indraprahasta et al., 2018; Warburton, 2018; Hudalah et al., 2021). The central government's role includes the production of national-wide development frameworks and directives as well as the provision of technical and substantial assistance of such national initiatives to the local governments. The development of green cities (through the P2KH program) is no exception. This most "noticeable" pro-environment urban strategy was initiated by the central government and was expected to be trialed across a large number of cities in the country.

As noted earlier, the development of green cities in Indonesia has not been backed up by a broader urban policy framework. Given this absence, Law 26/2007 on Spatial Planning has been used as the regulatory anchoring in the adoption of the green city concept, particularly the mandate to provide green open space to at least 30% of a city's total area. The utilization of this particular mandate as the main departure point, coupled with the lack of a clear sustainable cities vision, has resulted in a somewhat narrow reinterpretation of the green city conceptualization. This situation has been further aggravated by the actors involved in the policy making process at the national level. By this we mean that PUPR, being the initiator of and acting as the leading public agency in green cities development, has not invited other related ministries, including the Ministry of Environment and Forestry (KLHK), at an equal basis. This lack of inter-sectoral collaboration and institutional fragmentation has therefore dwarfed the relatively rich definition of a green city. As a result, the development of green cities in Indonesia has been deployed as a park-centered project.

From a governance perspective, PUPR's limited interpretation of the concept is also visible in the establishment of FKH at the local level. As one of the requirements for cities to be selected in the green city program, the local governments are obligated to form a green community, which is argued to ostensibly represent the participatory process of the program. The spatial planning system (Law 26/2007) promotes active participation by citizens in all stages of spatial planning process (i.e., planning, implementation, and control). However, while democracy has been given more space in Indonesia, participating citizens have been oftentimes brought on board to legitimate government-initiated purposes (Widianingsih and Morrell, 2007; Anindito et al., 2021). In the case of the green city initiative, the local government invited community leaders and environmental activists as representatives of citizens in the hope that they can maintain and use the pilot park developed by the central government. Citizens were, however, positioned more as users and were not involved in the broader policy making process, including in matters regarding where the park should be built, how the park should be designed, or, more broadly, and how the green city program would fit into the city's sustainability vision.

Recent discussion of green cities has paid great attention to the environmental "justice" issue, which also resonates with the "distributive" aspect of democratic urban governance (Anguelovski et al., 2018; Ghosh and Arora, 2021). The practice of FKH, and of green city governance in general, hardly took this distributive aspect into consideration. The selection of representatives of citizens in FKH did not consider the representativeness of different social groups of citizens. Limited attention was thus given to the social diversity of a city, addressing how the development of the pilot park would yield equal benefits to all citizens, irrespective of their social group. The absence of this distributive or justice element can be also seen from a broader policy perspective. When taking measures concerning environmental issues, many (if not most) practices of spatial planning in Indonesia have treated these as a blanket phenomenon, thus disregarding how such challenges may have different implications to different groups of citizens living in different places within a city (Padawangi, 2012; Leitner et al., 2017). Meanwhile, the spatial planning law itself solely focuses on the provision of a minimum percentage of green open space without explicitly linking this mandate to a city's unique socio-economic landscape.

Another important challenge to optimally implement pro-environmental (urban) policy, particularly in many developing countries, is that economic factors, combined with political interests (vis-à-vis environmental considerations), often have more influence on the direction of development decisions. In Indonesia, practices of land development have been largely driven by neoliberal accumulation regimes (Gellert, 2015; Leitner and Sheppard, 2018; Dale, 2021). Combined with a lack of spatial planning law enforcement, market favoring policy has facilitated the commodification of land. Private developers have assumed a larger role in transforming the country's urban landscape through the development of industrial estates, multifunctional new towns, commercial complexes, and residential areas (e.g., Firman, 2004; Firman and Fahmi, 2017). Here, we should not only be aware of

TABLE 4 | Different trajectories of developing potentialities of green cities in the global north and south.

Indicators	Green city in the global north	Green city in the global south
Urbanization	Green-compact-smart city	Interdependency of urban-rural system
	Developing various types of green spaces	Focusing on edible/productive green spaces (urban dan peri-urban agriculture)
	Reducing mobility	Facilitating mobility via mass rapid transportation
	Compact urban form	Polycentric urban from
Biophysics	Developing GI and NBS for supporting various ES	Developing GI and NBS for water resource management
	Integration of green and grey infrastructures	Integration of green and blue spaces
	High-technology and high-cost maintenance of GI and NBS	Public participation in building and maintaining GI and NBS
Governance	Participation and collaborative actions	Multi-actors and multi-level governance
	Aiming to provide ecosystem services	Aiming to deal with multidimensional issues in achieving SDGs

the undesirable consequences of such land development to a city's increasing environmental pressure, but also to the widening of environmental injustice within a city. Large-scale housing complexes constructed for middle- and upper-classes, for instance, have resulted in spatial segregation and fragmentation that have largely benefited the rich (Winarso et al., 2015). Mostly located in an area with a better natural landscape in the context of Indonesia's highly urbanized metropolises, the inhabitants living within these enclaved complexes enjoy a high value of green spaces per capita that leads to much better ecosystem services. On the other hand, on a city (and metropolitan) scale, the total area of green spaces has continued to decrease as a result of uncontrolled land use conversion.

The large-scale urban and infrastructure transformation has also been shaped by the neoliberal developmentalist vision of the state. A notable example would be the recent Trans Java Expressway development in Java's North Cost (Hudalah et al., 2020). Having the intention to spur interregional connectivity and economic growth in Indonesia's most populated island, this state-led megaproject may increase the vulnerability of cities located along the coastline to environmental pressures, most notably flooding and land subsidence (Sarah and Soebowo, 2018; Handayani et al., 2020). Such land development has also often sacrificed the everyday life of communities at the margin even though it is carried out under the environmental protection banner. The most recent example would be the climate change mitigation plan in the Jakarta Bay area that aims to protect the coast from sea level rise. This plan includes the mega-project development of a giant seawall with an integrated new town (comprising residential, commercial, and recreational spaces) branded as part of Jakarta's global city imaginary at the expense of the urban poor and fishing communities (Padawangi, 2012; Leitner et al., 2017).

While being a central government initiative, the state of policy and governance of green cities vary across cities. In decentralizing Indonesia, the fate of green cities, or of urban sustainability more broadly, depends on the initiative and efforts taken by the local governments as they have been given more room in steering their local development trajectory (Tarigan and Sagala, 2018). While many cities in Indonesia focus on reactive and short-term policy solutions, some anecdotal examples have exhibited the importance of taking continuous concrete steps to mitigate environmental challenges. A prominent example would be Surabaya, a city that is widely known for its abundant presence of green open spaces and community-based waste management. The city's current outstanding status was not achieved overnight, but is rather a constant ongoing process that started in 2002 under the leadership of mayor Bambang Dwi Hartono (Novalia et al., 2018). One of these earliest efforts was the merging of the Cleaning Department (*Dinas Kebersihan*) and Parks Department (*Dinas Pertamanan*) under Regional Law 14/2005, known there onward as *Dinas Kebersihan dan Pertamanan* (DKP). Another important aspect is the ability of the local governments to facilitate collaboration (Tarigan and Sagala, 2018). Such mode of governance creates a window of opportunity for generating more innovative ideas and solutions as well as co-sharing different resources in addressing environmental challenges. Balikpapan, for instance, has partnerships with at least 10 cities in Indonesia and with a number of national and international agencies, including Local Governments for Sustainability South-East Asia (ICLEI) and the Association of Indonesian Cities under United Nations Habitat funds (Tarigan and Sagala, 2018). In the case of Surabaya, the sister city cooperation with Kitakyushu, Japan, that was launched in 2002, has laid a learning platform for Surabaya to govern the city's sustainability path (Kurniawan et al., 2013; Novalia et al., 2018). These multiple cities and institutions' collaborations have altogether promoted knowledge sharing and co-learning processes. Meanwhile, Palembang's city government has established mutual linkages with educational institutions and private sectors to support the city's low-carbon development (Tarigan and Sagala, 2018). Many local governments in Indonesia, however, are still grappling with the practices of participatory and collaborative mode of governance (Indraprahasta et al., Forthcoming).

DISCUSSION AND CONCLUSION

The three aspects discussed above have explicated the geographical situatedness of green cities in Indonesia. In an increasingly globalized and interconnected world, the diffusion of blueprints for urban development, including green cities, across countries have escalated. As a result, various urban models and jargon have spread like wildfire: policy makers are racing to adopt a variety of global labels into the urban agenda in their respective countries and cities. In these practices of "worlding cities" (Roy and Ong, 2011), cities in the Global North have oftentimes become exemplary models to mimic. In this paper, we argue for the need to better understand the contextual specifics within which cities are located as there is no one-size-fits-all urban model.

The green city concept offers some opportunities for urban sustainability. In the Global North, the conceptualization of the green city has continued to evolve and intersected with newer environmental approaches in urban planning, including ES, GI, and NBS. It has also adopted some principles of other urban concepts such as compact city and smart city. From the case of Indonesia, it can be gleaned that the recent experimentation of green city development seems to have followed its own trajectory. Such a trajectory has been a result of the particular way the green city concept has been (re)interpreted and localized. To a certain degree, this has also been a result of the geographical situatedness of Indonesia, particularly relating to the country's urban sustainability policy. In our reading, other contextual aspects, particularly urbanization patterns and processes as well as biophysical characteristics, tend to be overlooked in the green city's adoption process.

With regard to Indonesia's reinterpretation of a green city, it is clear the concept has been adopted in a somewhat narrow manner centered around the provision of parks, a concern that is actually closer to the "garden city" concept. While the concept has ostensibly adopted the holistic vision of a green city, i.e., represented by the use of a number of green city indicators, the implementation of the concept can be seen as an urban placebo, where it has not heralded any significant intervention in terms of scale and urban, and spatial impact. Needless to say, Indonesia's adoption of a green city does not seem to be thought of as a systemic approach to the urban system, which has resulted from technocratic aspirations combined with the lack of long-term visioning of typical project-based activities.

However, in our view, the green city concept still endows some potentialities to achieve urban sustainability in the country by making some relevant adjustments. So, regardless of its limitation in terms of the concept adoption between 2011 and 2020, here we also intend to argue that further experimentation of green cities should appreciate the context specifics of Indonesian cities that may exhibit different characteristics from those of the Global North. It should be noted, however, that given the nature of our study, we did not intend to offer an archetypal model of green cities in Indonesia. Rather, from our previous discussion concerning the three aspects, we can sketch why a one-size-fits-all urban model does not exist and how and to what extent the green city concept can be contextually adopted.

First, regarding urbanization issues, green cities in the Global North were planned to be compact so as to expand areas allocated for green spaces and to reduce emissions produced by (mostly) private vehicles. However, urbanization in the Global South, including Indonesia, tends to have a sprawl pattern that form a mix of urban-rural land use, widely known as *desakota*. Following the trajectory of green cities in the Global North would, therefore, be problematic. Given this urban-rural relation, green cities as developed in the Global North can be adopted by also acknowledging the rural land uses as part of the urban system that produce ecosystem services. Most of these rural land uses are indeed farmland. With the application of good agricultural practices, farmland can also play a key role in environmental improvement as well as food security and income generation. Edible or productive green spaces could become a solution to provide ecosystem services. Another issue that should also be noted would be environmental justice as low-density settlements with larger green spaces are usually found in high-class housing complexes, mostly located in the peri-urban areas. Meanwhile, with regard to the emissions produced by private car-dependent commuting, mass public transportation with lower emissions can be developed to connect the urban core with settlements in the peri-urbans. Furthermore, encouraging polycentric urban form can be an alternative future metropolitan development to reduce commuting behavior (Hudalah and Firman, 2012). It is worth noting that, stimulated by the current pandemic situation, the change of work behavior from work from office to work from home has been proven to function well and to reduce emissions (Rendana and Komariah, 2021).

Second, mainstreaming GI and NBS has become a new trend in developing green cities in the Global North. Some ideas certainly can be adopted for developing green cities in the Global South, especially in planning and designing urban green infrastructures to provide ecosystem services for the public. However, as the bio-physic condition of cities in the Global South varies in terms of ecosystem types and biodiversity richness, priorities should be established. Particularly for Indonesia and tropical monsoon countries, water management is a key element for reaching sustainability (Kooy et al., 2019; Lechner et al., 2020). Therefore, GI and NBS approaches should be focused on storm water management, including restoring watershed quality from upstream to downstream. In parallel, waste management should be of concern as it has a close relationship with water management issues. In this case, green and blue spaces should be planned in an integrated manner and be included in the spatial planning vis-à-vis focusing on building parks or gardens to fulfill 30% green open space as mandated by the Indonesian spatial planning law. Another important aspect is developing multifunctional green and blue spaces vis-à-vis building green spaces with high technology and high-cost maintenance as in the Global North cities (e.g., roof garden, vertical garden, and green façade, etc.). In our view, this effort is crucial as most cities in the Global South have a lack of financial capacity. It is also expected that by building multifunctional spaces, communities and other participating actors would be more interested in getting involved in maintaining these green and blue spaces.

Third, participation and a collaborative approach is a prerequisite in developing green cities in the Global North. Such an approach is indeed ideal and should be adopted. However, the role of government is still very much dominant in the Global South cities. In addition to this, fragmented governance is also clearly visible in the context of the Indonesian green city program. Multi-level and multi-actor governance should be encouraged by considering horizontal and vertical boundaries of decision-making (Pauleit et al., 2021). This can be done if the green city program was not simply seen as an effort to reach 30% green open space. Rather, green cities should be positioned as an integral part of a broader urban agenda: climate change adaptation and mitigation, achieving SDGs (particularly the 11th goal of SGDs), and increasing the quality of life and urban productivities, just to name a few. Such an integrated vision is therefore expected to involve other related government institutions outside PUPR. These institutions include multi-sectors or multi-departments at the national and local levels as well as other actors outside the government at different geographical scales.

Taken together, this paper has proposed some key points that can be taken into consideration in adopting and developing green cities in the Global South through the lens of urbanization, bio-physics, and governance aspects. These key points are summarized in **Table 4**. Local characteristics and geographical situatedness of cities should be considered in this adoption process vis-à-vis simply mimicking green cities conceptualized and developed from and in the Global North. Other lenses can be also employed to expand the perspective offered by this paper. These lenses may include social, cultural, and economic aspects. Our main intention here has been to emphasize that adjustments are required to develop green cities in different socio-ecological landscapes.

DATA AVAILABILITY STATEMENT

The datasets generated during and/or analyzed during the current study are available from the corresponding author upon reasonable request.

AUTHOR CONTRIBUTIONS

All authors have contributed equally to the work and approved it for publication.

SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fenvs.2022.787204/full#supplementary-material

REFERENCES

Acuto, M., Parnell, S., and Seto, K. C. (2018). Building a Global Urban Science. *Nat. Sustain.* 1, 2–4. doi:10.1038/s41893-017-0013-9

Adler, P. A., and Adler, P. (1994). "Observation Techniques," in *Handbook of Qualitative Research*. Editors N. K. Denzin and Y. S. Lincoln (Thousand Oaks, CA: Sage), 377–392.

Ahmed, Z., Wang, Z., and Ali, S. (2019). Investigating the Non-linear Relationship between Urbanization and CO2 Emissions: An Empirical Analysis. *Air Qual. Atmos. Health* 12, 945–953. doi:10.1007/s11869-019-00711-x

Alisjahbana, A. S., and Murniningtyas, E. (2018). *Tujuan Pembangunan Berkelanjutan di Indonesia: Konsep, Target dan Strategi Implementasi*. Bandung: Unpad Press.

Anguelovski, I., Connolly, J., and Brand, A. L. (2018). From Landscapes of utopia to the Margins of the green Urban Life. *City* 22 (3), 417–436. doi:10.1080/13604813.2018.1473126

Anindito, D. B., Sagala, S. A. H., and Tarigan, A. K. M. (2021). E-musrenbang: a Digital Framework for Local Participatory Planning at the Community Level. *Int. Dev. Plann. Rev.* ahead-of-print, 1–26. doi:10.3828/idpr.2021.5

Aronson, M. F., Lepczyk, C. A., Evans, K. L., Goddard, M. A., Lerman, S. B., Macivor, J. S., et al. (2017). Biodiversity in the City: Key Challenges for Urban green Space Management. *Front. Ecol. Environ.* 15 (4), 189–196. doi:10.1002/fee.1480

Artmann, M., Kohler, M., Meinel, G., Gan, J., and Ioja, I.-C. (2019). How Smart Growth and green Infrastructure Can Mutually Support Each Other - A Conceptual Framework for Compact and green Cities. *Ecol. Indicators* 96 (June), 10–22. doi:10.1016/j.ecolind.2017.07.001

Bps [Indonesia Statistical Office] (2019). *Statistik Komuter Jabodetabek: Hasil Survei Komuter Jabodetabek 2019*. Jakarta: BPS.

Breuste, J., Artmann, M., Ioja, C., and Qureshi, S. (2020). The Green City: General ConceptMaking Green Cities Concepts, Challenges and Practices. *Springer*, 1–15. doi:10.1007/978-3-030-37716-8_1

Brown, R. B. (2006). *Doing Your Dissertation in Business and Management: The Reality of Research and Writing*. London: Sage Publications.

Dahiya, B. (2012). Cities in Asia, 2012: Demographics, Economics, Poverty, Environment and Governance. *Cities* 29, S44–S61. doi:10.1016/j.cities.2012.06.013

Dale, C. J. P. (2021). Komodo vs Pembangunanisme Kolonial. Majalah Tempo. Available at https://majalah.tempo.co/read/investigasi/162227/kolom-wisata-eksklusif-komodo-mirip-cara-belanda-mengembangkannya (Accessed 2 January 2021).

Derkzen, M. L., Van Teeffelen, A. J., Nagendra, H., and Verburg, P. H. (2017). Shifting Roles of Urban green Space in the Context of Urban Development and Global Change. *Curr. Opin. Environ. Sustainability* 29, 32–39. doi:10.1016/j.cosust.2017.10.001

DeWalt, K. M., and DeWalt, B. R. (2002). *Participant Observation: A Guide for Fieldworkers*. Walnut Creek, CA: AltaMira Press.

EC (2013). Building a green Infrastructure for Europe. [European Commission] Available at https://ec.europa.eu/environment/nature/ecosystems/docs/green_infrastructure_broc (Accessed September 10, 2021).

EIU (2012). The Green City Index. Siemens AG Corporate Communications and Government Affairs Wittelsbacherplatz. *München* 2, 80333. [Economist Intelligent Unit].

El Ghorab, H. K., and Shalaby, H. A. (2016). Eco and Green Cities as New Approaches for Planning and Developing Cities in Egypt. *Alexandria Eng. J.* 55 (1), 495–503. doi:10.1016/j.aej.2015.12.018

Elmqvist, T., and Mcdonald, R. I. (2013). *Urbanization Biodiversity and Ecosystem Services : Challenges and Opportunities*.

Estoque, R. C., and Murayama, Y. (2015). Intensity and Spatial Pattern of Urban Land Changes in the Megacities of Southeast Asia. *Land Use Policy* 48, 213–222. doi:10.1016/j.landusepol.2015.05.017

Firman, T., and Fahmi, F. Z. (2017). The Privatization of Metropolitan Jakarta's (Jabodetabek) Urban Fringes: The Early Stages of "Post-Suburbanization" in Indonesia. *J. Am. Plann. Assoc.* 83 (1), 68–79. doi:10.1080/01944363.2016.1249010

Firman, T. (2004). Major Issues in Indonesia's Urban Land Development. *Land Use Policy* 21 (4), 347–355. doi:10.1016/j.landusepol.2003.04.002

Firman, T. (2002). Urban Development in Indonesia, 1990-2001: from the Boom to the Early Reform Era through the Crisis. *Habitat Int.* 26, 229–249. doi:10.1016/s0197-3975(01)00045-5

Garcia-Lamarca, M., Anguelovski, I., Cole, H., Connolly, J. J., Argüelles, L., Baró, F., et al. (2021). Urban green Boosterism and City Affordability: For Whom Is the 'branded' green city. *Urban Stud.* 58 (1), 90–112. doi:10.1177/0042098019885330

Gellert, P. K. (2015). Palm Oil Expansion in Indonesia: Land Grabbing as Accumulation by Dispossession. *Curr. Perspect. Soc. Theor.* 34, 6599. doi:10.1108/s0278-120420150000034004

Ghosh, B., and Arora, S. (2021). Smart as (Un)democratic? the Making of a Smart City Imaginary in Kolkata, India. *Environ. Plann. C: Polit. Space*, 239965442110275. doi:10.1177/23996544211027583

Gold, R. L. (1958). Roles in Sociological Field Observations. *Social Forces* 36, 217–223. doi:10.2307/2573808

Handayani, W., Chigbu, U. E., Rudiarto, I., and Putri, I. H. S. (2020). Urbanization and Increasing Flood Risk in the Northern Coast of Central Java-Indonesia: An Assessment towards Better Land Use Policy and Flood Management. *Land* 9, 343. doi:10.3390/land9100343

Hansen, R., Olafsson, A. S., Van Der Jagt, A. P. N., Rall, E., and Pauleit, S. (2019). Planning Multifunctional green Infrastructure for Compact Cities: What Is the State of Practice. *Ecol. Indicators* 96 (November), 99–110. doi:10.1016/j.ecolind.2017.09.042

Hardiman, R. (2020). China and Natural Based Solution. Available at: https://www.regreen-project.eu/news/china-and-nature-based-solutions/(Accessed September 110, 2021).

Hardoy, J. E., Mitlin, D., and Satterthwaite, D. (2001). *Environmental Problems in an Urbanizing World: Finding Solutions in Cities in Africa, Asia and Latin America*. 2nd ed. Routledge. doi:10.4324/9781315071732

Hudalah, D., and Firman, T. (2012). Beyond Property: Industrial Estates and post-suburban Transformation in Jakarta Metropolitan Region. *Cities* 29 (1), 40–48. doi:10.1016/j.cities.2011.07.003

Hudalah, D., Octifanny, Y., Talitha, T., Firman, T., and Phelps, N. A. (2020). From Metropolitanization to Megaregionalization: Intentionality in the Urban Restructuring of Java's North Coast, Indonesia. *J. Plann. Edu. Res.*, 0739456X2096740. doi:10.1177/0739456X20967405

Hudalah, D., Talitha, T., and Lestari, S. F. (2021). Pragmatic State Rescaling: The Dynamics and Diversity of State Space in Indonesian Megaproject Planning and Governance. *Environ. Plann. C: Polit. Space*, 239965442110309. doi:10.1177/23996544211030935

Indraprahasta, G. S., Derudder, B., and Koelemaij, J. (2018). Global-local Dynamics in the Transformation of the Jakarta Metropolitan Area into a Global City-Region. *disP - Plann. Rev.* 54 (3), 52–62. doi:10.1080/02513625.2018.1525212

Indraprahasta, G. S., and Derudder, B. (2019). World City-Ness in a Historical Perspective: Probing the Long-Term Evolution of the Jakarta Metropolitan Area. *Habitat Int.* 89, 102000. doi:10.1016/j.habitatint.2019.102000

Indraprahasta, G. S., Fahmi, F. Z., and Alamsyah, P. (Forthcoming). "The Dynamics of Multi-Scalar Networks Underlying the Creative City Process: The Case of Bandung," in *Routledge Handbook of Urban Indonesia*. Editors S. Roitman and D. Rukmana (Abingdon, UK & New York: Routledge).

IQAir (2019). *World Air Quality Report*. Goldach: IQAir.

Jafari, Y., Othman, J., and Nor, A. H. S. M. (2012). Energy Consumption, Economic Growth and Environmental Pollutants in Indonesia. *J. Pol. Model.* 34, 879–889. doi:10.1016/j.jpolmod.2012.05.020

Jambeck, J. R., Geyer, R., Wilcox, C., Siegler, T. R., Perryman, M., Andrady, A., et al. (2015). Plastic Waste Inputs from Land into the Ocean. *Science* 347 (6223), 768–771. doi:10.1126/science.1260352

Joga, N., and Ismaun, I. (2011). *RTH 30 %! Resolusi (Kota) Hijau*. Jakarta: Gramedia Pustaka Utama.

Kahn, C. (1982). "Historical Roots: The History of Community Gardening," in *A Handbook of Community Gardening*. Editor S. Naimark (New York, NY: Charles Scribner's Sons), 11–16.

Katherina, L. K., and Indraprahasta, G. S. (2019). Urbanization Pattern in Indonesia's Secondary Cities: Greater Surabaya and its Path toward a Megacity. *IOP Conf. Ser. Earth Environ. Sci.* 338, 012018. doi:10.1088/1755-1315/338/1/012018

Kenichiro, A. (2015). Jakarta "Since Yesterday": The Making of the post-new Order Regime in an Indonesian metropolis. *Southeast Asian Stud.* 4 (3), 445–486.

Kirmanto, D., Ernawi, I. S., and Djakapermana, R. D. (2014). *Indonesia green city Development Program: An Urban Reform. Paper Presented at the 48th ISOCARP Congress, 10-13 September 2012*. Perm Russia.

KLH (1997). *Agenda 21 Indonesia: Strategi Nasional Untuk Pembangunan Berkelanjutan*. Jakarta: KLH. Ministry of Environment.

KLH (2002). *Dari krisis menuju keberlanjutan meniti jalan pembangunan berkelanjutan di Indonesia: Tinjauan pelaksanaan Agenda 21*. Jakarta: KLH. Ministry of Environment.

Kooy, M., Furlong, K., and Lamb, V. (2020). Nature Based Solutions for Urban Water Management in Asian Cities: Integrating Vulnerability into Sustainable Design. *Int. Dev. Plann. Rev.* 42 (3), 381–390. doi:10.3828/idpr.2019.17

Kurniawan, T. A., Puppim de Oliveira, J., Premakumara, D. G. J., and Nagaishi, M. (2013). City-to-city Level Cooperation for Generating Urban Co-benefits: the Case of Technological Cooperation in the Waste Sector between Surabaya (Indonesia) and Kitakyushu (Japan). *J. Clean. Prod.* 58, 43–50. doi:10.1016/j.jclepro.2013.08.002

Laurier, E. (2010). "Participant Observation," in *Key Methods in Geography*. Editors N. Clifford, S. French, and G. Valentine (London: Sage), 116–130.

Lechner, A. M., Gomes, R. L., Rodrigues, L., Ashfold, M. J., Selvam, S. B., Wong, E. P., et al. (2020). Challenges and Considerations of Applying Nature-Based Solutions in Low- and Middle-Income Countries in Southeast and East Asia. *Blue-Green Syst.* 2 (1), 331–351. doi:10.2166/bgs.2020.014

Leducq, D., and Scarwell, H.-J. (2020). Green-city Models as an Urban Strategy: Hanoi between International Practices and Local Assemblage. *Town Plann. Rev.* 91 (4), 437–456. doi:10.3828/tpr.2020.25

Leitner, H., Colven, E., and Sheppard, E. (2017). "Ecological Security for Whom? the Politics of Flood Alleviation and Urban Environmental justice in Jakarta, Indonesia," in *The Routledge Companion to the Environmental Humanities*. Editors U. Heise, J. Christensen, and M. Niemann (London: Routledge), 194–205.

Leitner, H., and Sheppard, E. (2018). From Kampungs to Condos? Contested Accumulations through Displacement in Jakarta. *Environ. Plan. A.* 50 (2), 437–456. doi:10.1177/0308518x17709279

Lengeweg, F., Hilderink, H., and Maas, R. (2000). *Urbanisation, Industrialisation, and Sustainable Development. Research for Man and Environment*. Tokyo: United Nations University.

Li, K., and Lin, B. (2015). Impacts of Urbanization and Industrialization on Energy consumption/CO2 Emissions: Does the Level of Development Matter. *Renew. Sustain. Energ. Rev.* 52, 1107–1122. doi:10.1016/j.rser.2015.07.185

Liendfield, M., and Steinberg, F. (2012). *Green Cities*. Mandaluyong City: ADB Publication.

Masripatin, N., Rachmawaty, E., Suryanti, Y., Setyawan, H., Farid, M., and Iskandar, N. (2017). *Strategi Implementasi NDC (Nationally Determined Contribution)*. Jakarta: Ministry of Environmental Affairs and Forestry.

McGee, T. G. (1991). "The Emergence of Desakota Regions in Asia: Expanding a Hypothesis," in *The Extended Metropolis: Settlement Transition in Asia*. Editors N. Ginsburg, B. Koppel, and T. G. McGee (Honolulu: University of Hawaii Press), 3–25.

MEA (2005). *Ecosystems and Human Wellbeing*. Washington, DC: Island Press. Millennium Ecosystem Assessment.

Ministry of Public Works (2013a). *Manual Kegiatan Program Pengembangan Kota Hijau 2013*. Jakarta: Direktorat Jenderal Penataan Ruang, Kementerian Pekerjaan Umum.

Ministry of Public Works (2013b). *Panduan Pengembangan Kota Hijau 2013. Jakarta: Direktorat Jenderal Penataan Ruang, Kementerian Pekerjaan Umum.*

Moore, S. (2006). Forgotten Roots of the Green City: Subsistence Gardening in Columbus, Ohio, 1900-1940. *Urban Geogr.* 27 (2), 174–192. doi:10.2747/0272-3638.27.2.174

Nesshöver, C., Assmuth, T., Irvine, K. N., Rusch, G. M., Waylen, K. A., Delbaere, B., et al. (2017). The Science, Policy and Practice of Nature-Based Solutions: An Interdisciplinary Perspective. *Sci. Total Environ.* 579, 1215–1227. doi:10.1016/j.scitotenv.2016.11.106

Novalia, W., Rogers, B. C., Bos, J. J., and Brown, R. R. (2018). "Diagnosing Strategic Influences of Different Agents in Governing Transition towards green city in Surabaya, Indonesia," in Paper presented at the Earth Systems Governance Conference, 5-8 November 2018, Utrecht, the Netherlands.

Pace, R., Churkina, G., and Rivera, M. (2016). *How green Is a "Green City"? A Review of Existing Indicators and Approache*s. IASS Working Paper. December 2016. doi:10.2312/iass.2016.035

Padawangi, R. (2012). "Climate Change and the north Coast of Jakarta: Environmental justice and the Social Construction of Space in Urban Poor Communities," in *Urban Areas and Climate Change (Research in Urban Sociology*. Editor W. G. Holt (Bingley, UK: Emerald), 12, 321–339.

Pankaja, M. S., and Nagendra, H. N. (2015). Green city Concept – as New Paradigm in Urban Planning. *Int. J. Eng. Sci.* 4 (10), 55–60.

Pauleit, S., Hansen, R., and Pribadi, D. O. (2021). Editorial: Transformative Urban Greening: Advancing Green Space Governance. *Front. Sustain. Cities* 3 (September), 1–3. doi:10.3389/frsc.2021.735918

Pauleit, S., Zölch, T., Hansen, R., Randrup, T. B., and Konijnendijk van den Bosch, C. (2017). "Nature-Based Solutions and Climate Change - Four Shades of Green," in *Nature-based Solutions to Climate Change Adaptation in Urban Areas, Theory and Practice of Urban Sustainability Transitions*. Editors

N. Kabisch, H. Korn, J. Stadler, and A. Bonn (Springer), 29–49. doi:10.1007/978-3-319-56091-5_3

Pribadi, D. O., and Pauleit, S. (2015). The Dynamics of Peri-Urban Agriculture during Rapid Urbanization of Jabodetabek Metropolitan Area. *Land Use Policy* 48, 13–24. doi:10.1016/j.landusepol.2015.05.009

Rendana, M., and Komariah, L. N. (2021). The Relationship between Air Pollutants and COVID-19 Cases and its Implications for Air Quality in Jakarta, Indonesia. *Jpsl* 11 (1), 93–100. doi:10.29244/jpsl.11.1.93-100

Richter, B., and Behnisch, M. (2019). Integrated Evaluation Framework for Environmental Planning in the Context of Compact green Cities. *Ecol. Indicators* 96 (2), 38–53. doi:10.1016/j.ecolind.2018.05.025

Roy, A., and Ong, A. (2011). *Worlding Cities: Asian Experiments and the Art of Being Global*. West Sussex: Blackwell Publishing.

Sachs, J. D. (2015). *The Age of Sustainable Development*. New York: Columbia University Press.

Salim, W., and Hudalah, D. (2020). "Urban Governance Challenges and Reforms in Indonesia: Towards a New Urban Agenda," in *New Urban Agenda in Asia-Pacific: Governance for Sustainable and Inclusive Cities*. Editors B. Dahiya and A. Das (Singapore: Springer), 163–181. doi:10.1007/978-981-13-6709-0_6

Sarah, D., and Soebowo, E. (2018). Land Subsidence Threats and its Management in the North Coast of Java. *IOP Conf. Ser. Earth Environ. Sci.* 118, 012042. doi:10.1088/1755-1315/118/1/012042

Sarosa, W., Isnutomo, M. D., Virant, G. E., Sarahidha, D., and Susetyo, N. A. (2017). *Panduan Praktis Implementasi Agenda Baru Perkotaan Untuk Kota Berkelanjutan Indonesia*. Jakarta: Ministry of Public Works and Housing.

Schwarz, N., Moretti, M., Bugalho, M. N., Davies, Z. G., Haase, D., Hack, J., et al. (2017). Understanding Biodiversity-Ecosystem Service Relationships in Urban Areas: A Comprehensive Literature Review. *Ecosyst. Serv.* 27, 161–171. doi:10.1016/j.ecoser.2017.08.014

Shen, Z., and Fitriaty, P. (2018). "Overview: Green City Planning and Practices in Asian Cities," in *Green City Planning and Practices in Asian Cities, Strategies for Sustainability*, 1–16. doi:10.1007/978-3-319-70025-0_1

Supriyatin, R., Pravitasari, A. E., and Pribadi, D. O. (2020). Pola Interaksi Spasial serta Karakteristik Individu dan Rumah Tangga Komuter antar Kabupaten/Kota di Bandung Raya. *Jp2Wd* 4 (2), 113–133. doi:10.29244/jp2wd.2020.4.2.113-133

Swedberg, R. (2020). "Exploratory Research," in *The Production of Knowledge*. Editors C. Elman, J. Gerring, and J. Mahoney (Cambridge: Cambridge University Press), 17–41. doi:10.1017/9781108762519.002

Tarigan, A. K. M., and Sagala, S. (2018). The Pursuit of Greenness: Explaining Low-Carbon Urban Transformation in Indonesia. *Int. Plann. Stud.* 23 (4), 408–426. doi:10.1080/13563475.2018.1513360

The Jakarta Post (2018). Air Pollution in Jakarta, an Invisible Health Threat. Available at: https://www.thejakartapost.com/adv/2018/08/15/air-pollution-in-jakarta-an-invisible-health-threat.html. (Accessed 15 August).

UNDESA (2018). *World Urbanization Prospects: The 2018 Revision*. Online Edition. United Nations, Department of Economic and Social Affairs, Population Division.

V. C. DeMunck and E. J. Sobo (Editors) (1998). *Using Methods in the Field: A Practical Introduction and Casebook* (Walnut Creek, CA: AltaMira Press).

Venkatesh, G. (2013). A Critique of the European Green City Index. *J. Environ. Plann. Manage.* 57 (3), 317–328. doi:10.1080/09640568.2012.741520

Warburton, E. (2018). A New Developmentalism in Indonesia. *Ae* 35 (3), 355–368. doi:10.1355/ae35-3c

Warner, S. B. (1987). *To Dwell Is to Garden: A History of Boston's Community Gardens*. Boston, MA: Northeastern University Press.

Widianingsih, I., and Morrell, E. (2007). Participatory Planning in Indonesia. *Pol. Stud.* 28 (1), 1–15. doi:10.1080/01442870601121320

Wijaya, A., Chrysolite, H., Ge, M., Wibowo, C., Pradana, A., Utami, A., et al. (2017). *How Can Indonesia Achieve its Climate Change Mitigation Goal? an Analysis of Potential Emissions Reductions from Energy and Land-Use Policies*. Working Paper. Jakarta, Indonesia: World Resources Institute. Available online at www.wri.org/publication/how-can-indonesia-achieve-itsclimate-goal.

Winarso, H., Hudalah, D., and Firman, T. (2015). Peri-urban Transformation in the Jakarta Metropolitan Area. *Habitat Int.* 49, 221–229. doi:10.1016/j.habitatint.2015.05.024

World Bank (2020). Urban Development. Available at https://www.worldbank.org/en/topic/urbandevelopment/overview.

Xie, L., and Bulkeley, H. (2020). Nature-based Solutions for Urban Biodiversity Governance. *Environ. Sci. Pol.* 110, 77–87. doi:10.1016/j.envsci.2020.04.002

Zain, A. M., and Kencana, I. P. (2010). "Keberagaman kota dan pembangunan perkotaan di Indonesia: Peran ruang terbuka hijau dalam lingkungan hidup," in *Ministry of Public Works, Status Perkembangan Perkotaan Indonesia 2010: Mengelola Keberagaman Dan Kesejangan Pembangunan Kota* (Jakarta: Direktorat Jenderal Penataan Ruang, Kementerian Pekerjaan Umum), 2: 34–42:48.

Zinkernagel, R., Evans, J., and Neij, L. (2018). Applying the SDGs to Cities: Business as Usual or a New Dawn. *Sustainability* 10 (9), 3201. doi:10.3390/su10093201

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Predicting Sugar Balance as the Impact of Land-Use/Land-Cover Change Dynamics in a Sugarcane Producing Regency in East Java, Indonesia

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Lamongan Regency is one of Indonesia's regencies in Indonesia that contribute to the country's sugarcane and sugar production. Land-use/land-cover changes (LULC) have happened in various areas, including Lamongan Regency in East Java, Indonesia. This study aimed to analyze the impact of land-use/land-cover change (LULCC) in 2007–2031 on the sugar balance in the Lamongan Regency in 2031. The LULC forecast in 2031 was made using R studio using the CLUE-s function in the LULCC package. Following that, the sugar requirements and sugar balance in Lamongan Regency in 2031 were forecasts. The kappa accuracy of the LULC model tested in 2019 was 0.81 when using the CLUE-s model in R studio's LULCC package. The findings of the LULC prediction in Lamongan Regency for 2031 indicated that there could be significant changes in LULC, resulting in an estimated reduction in 1,687.5 ha sugarcane, influenced by LULC competition for built-up areas. The Built-up areas in Lamongan Regency have exploded in size as a result of population. An increase in built-up areas and a reduction in sugarcane plantations as a result of LULCC are expected to diminish the sugar balance in Lamongan Regency by 10,470.76 tons in 2031 when compared to 2019. The results of this study can be utilized to prevent uncontrolled LULCC in the future to meet sugar needs in Lamongan Regency and at the national level.

Keywords: prediction, modeling, LULCC, driving factor, land competition

1 INTRODUCTION

Indonesia's current sugar production capacity is insufficient to meet the community's high demand for sugar, which exceeds six million tons per year. National sugar production in 2019 was only 2.23 million tons. Meanwhile, in 2019 sugar imports in Indonesia imported 4.09 million tons, worth up to US $1.36 billion, or approximately Rp19.3 trillion. Sugar was imported from 20 countries, with Thailand importing the most (86.53%) and Australia importing the least (13.26%) (Indonesian Statistics, 2020). Sugar production is poor as a result of sugarcane production being low. Domestic sugar production peaked in the 1930s. At that time, 179 sugar factories were operational, but today, less than half of those remain operational, totaling 64 sugar factories (Marpaung et al., 2011). One of the causes of the fall in the number of sugar factories in Indonesia is a scarcity of raw materials sufficient to match the sugar factories' production capacity.

East Java Province is Indonesia's major producer of sugarcane and sugar. East Java Province itself accounted for 51.5% or 1.11 million tons, of the total national total sugar production in 2018. (Indonesian Statistics, 2019). East Java's top three sugarcane/sugar producers are Kediri, Malang, and Lumajang Regencies. Lamongan Regency is one of the regencies that contributes as a producer of sugar cane as well as sugar in East Java. Sugar production in Lamongan Regency has been highly erratic over the last 10 years. In 2008, sugar production was 18,445 tons, and subsequently declined until 2010 when it was only 11,542 tons. After that, production improved until 2014, it reached 24,995 tons. However, it dropped again in the following year, to 23,832 tons. In 2016 production grew to 28,521 tons, then fell to 28,104 in 2017 and 24,497 tons in 2018.

The increase in population in Lamongan Regency from 2015 to 2019 increased by 31,124 people or 2.31%, from 1,342,266 to 1,373,390 people (Lamongan Statistics, 2020). This rising population has an impact on the increasing demand for sugar usage. Population growth is also pushing changes in agricultural land use, particularly the convention of the sugarcane plantation sub-sector to non-agricultural. LULCC can be driven by a variety of factors, including population growth (Mhawish and Saba, 2016), industrialization (Hatami and Shafieardekani, 2014), urbanization (Tali and Murthy, 2013), and the lower economic value of agricultural land relative to built-up areas (Rondhi et al., 2018; Peerzado et al., 2019). Changes in LULC influence a variety of areas, such as agriculture, the economy, society, and the environment.

Modeling of LULC has been widely used to forecast future trends in LULCC. Several approaches were employed to model the LULC, including CLUE (the Conversion of Land Use/Land Cover and Its Effect) (Veldkamp and Fresco, 1996), Cellular Automata (Clarke and Gaydos, 1998), Spatial Markov (Wood et al., 1997), and the Logit model (Wear and Flamm, 1993). CLUE-s (The Conversion of Land Use/Land Cover and Its Effect at Small Regional Scale) is a variant of CLUE that takes into account the interaction between current LULCs and their underlying variables as well as competition among LULCs for regional applications (Verburg et al., 2002). The CLUE-s model is commonly utilized in land-use planning research (Islam et al., 2021; Song et al., 2021; Xu et al., 2021).

Therefore, it is necessary to investigate changes in LULC that are both existent and predictable to ascertain the future pattern of LULC changes in the Lamongan Regency. This study aimed to analyze the impact of land-use/land-cover change (LULCC) in 2007–2031 on the sugar balance in the Lamongan Regency in 2031. This research used the CLUE-s model to examine the influence of LULCC from 2007 to 2031 on the sugar sufficiency balance in Lamongan Regency in 2031. The prediction year in this research is 2031, which corresponds to the end of the Lamongan Regency's 2011–2031 Regional Spatial Plan (RTRW), and thus this research is expected to contribute to science related to land-use change and its impact on future food availability and can be used as input for formulating policy strategies related to sugarcane plantations and meeting the community's sugar needs in the coming period.

2 MATERIALS AND METHODS

2.1 Study Area

This study was conducted in the Lamongan Regency, Indonesia's East Java Province (**Figure 1**) Lamongan Regency, located between 6°51′54″–7°23′06″ south latitude and 112°33′45″–112°34′45″ east longitude has a tropical climate. Lamongan Regency is divided into 27 sub-districts had a population of 2019 of 1,373,390 people. The Java Sea forms the northern boundary of Lamongan Regency; Gresik Regency forms the eastern border boundary; Mojokerto and Jombang Regencies from the southern boundary, and Bojonegoro and Tuban Regencies form the western boundary.

2.2 Data

The orthorectified SPOT satellite images from 2007 to 2013 and 2019 were collected by the National Institute of Aeronautics and Space. The 2007 image was taken by a SPOT 4 satellite on 22 June 2007. The 2013 image was obtained by a SPOT 5 satellite on 12 June 2013, and the 2019 image was taken by a SPOT 5 satellite on 10 June 2019. The slope data is coming from NASA's Digital Elevation Model; the distance between the road and administration boundary comes from the Geospatial Information Agency; and the climate data from the Meteorology, Climatology, and Geophysics Council. The final resolution of all spatial data is processed is 30 × 30 m. Indonesia statistics provided statistical data on population and sugarcane productivity. GPS was used to conduct field checks, ArcGIS was utilized to rectify spatial data, R Studio was used for modeling, and Microsoft Office was used for other analyses.

2.3 Data Processing and Analysis

2.3.1 Analysis of Land-Use/Land-Cover Change in 2007, 2013, and 2019

The land use and cover of Lamongan Regency were investigated visually (Andries et al., 2021; Wei et al., 2021) using SPOT satellite images of 2007, 2013, and 2019. The analyzed LULC was classified using a one-level classification scheme into nine categories, including forest, field/moor, built-up area, open land, sugarcane plantation, rice field, shrubs, fishpond, and water body. The classification findings were validated in the field utilizing a purposive sample technique specific to the LULC type. The status of LULC in the past was determined through interviews with residents who knew the condition of the location in the past (Debeko et al., 2018) and also integrated with high-resolution time-series imagery on Google Earth (Getachew and Meten, 2021). Each type of LULC was assigned eight sample points, for a total of 72 points, taken under limited mobility during the COVID-19 pandemic. These sample points are then used as a reference for evaluating the classification accuracy of LULCs. The kappa accuracy test was used to obtain the accuracy. The greater the kappa accuracy score, the more precise the results of the LULC prediction. The kappa accuracy test has the following equation (Cohen, 1960; Foody, 2002; Adhiatma, 2020):

$$K_i = \frac{N\sum_{i=1}^{r} X_{ii} - \sum_{i=1}^{r} (X_{i+} + X_{+i})}{N^2 - \sum_{i=1}^{r} (X_{i+} + X_{+i})}$$

where Ki denotes the kappa accuracy index; xii: denotes the simulation result of the i LULC type which corresponds to the i LULC the observed; xi: the simulation result of the i LULC; x + i denotes the observed i LULC type; N denotes the total number of observation points, r denotes the total type of LULC.

2.3.2 Prediction Model of Land Use/Land Cover in Lamongan Regency in 2031

Model Description

The CLUE-s model was utilized in this research to simulate and predict changes in LULCs by considering the empirical conditions of a place, spatio-temporal dynamic interactions, and competition between LULCs at that location (Verburg, 2010). Elasticity is a measure of competition among LULCs; the greater the elasticity, the more competitive the LULC is in comparison to other LULCs (Liao et al., 2010). After determining the relationship between LULC and its driving factor through neighborhood analysis using binary logistic regression analysis the CLUE-s model can be employed to stimulate LULCC (Setiawan and Kunihiko, 2020). The binary logistic regression formula is:

$$ln\left[\frac{p}{1-p}\right] = \beta_0 + \beta_1 x_1 + \beta_2 x_2 + \beta_3 x_3 + \ldots + \beta_n x_n$$

where the dependent variable is the probability of occurrence of nonoccurrence; β0 is the intercept; and β1, β2, β3,.., βn: the coefficients of the driving factor as independent variables (x1, x2, x3,.., and xn) (Siagian et al., 2019). The slope, distance from the road, rainfall, and temperature are all considered driving considerations in this study (**Supplementary Figure S1**).

Following that, LULCC modeling can be carried out. The stages of LULCC modeling began with a simulation in 2007, and then the model was validated to predict LULC in 2013. The initial elasticity value for each LULC class was determined based on a previous study (Sinurat et al., 2015) then adjusted based on the model's performance in 2013 so that better modeling results are obtained. After that, the simulation is continued by predicting and validating LULC in 2019. If the validation results indicate that the model is accurate, LULC predictions for 2031 can be produced. In this investigation, the procedure of modeling and predicting LULC using the CLUE-s model was performed using the LULC package in the R studio software (Moulds et al., 2015), because it lacks cell boundaries and thus enables high-resolution analysis.

Model Validation

Model is validated is by comparing the LULC in 2019 obtained the simulation and the LULC in 2019 resulting from the categorization (actual map) (Leta et al., 2021; Yu et al., 2021). The accuracy of the model validation results was illustrated by the kappa index value which ranges from 0 to 1. A kappa index worth one means that there is a perfect agreement or the model is doing better, while the kappa index closer to 0 means there is a bad agreement (poor agreement) or the resulting model is not good (Landis and Koch, 1977).

2.3.3 Calculation of Sugar Balance in 2031

Before establishing the sugar balance, the sugar requirements for Lamongan Regency in 2031 were calculated by population projections. The population in 2031 is forecast by first estimating the rate of population growth over the previous 5 years (2015–2020). The population growth rate and population predictions are measured as follows (Indonesian Statistics, 2010; Mekonnen, 2018; Al-Eideh and Al-Omar, 2019) is:

$$r = \frac{1}{t} ln\left(\frac{P_t}{P_0}\right)$$

$$P_t = P_0 e^{rt}$$

where r donates the population growth rate, Pt is the total population in year t (projected year); P0 denotes the population in year 0 (base year); t denotes the time between years 0 and year t. Sugar needs may be obtained after determining the population in 2031. Sugar needs in Lamongan Regency are calculated as follows:

$$Sn = Sc*Pt$$

where Sn denotes sugar needs (ton); Sc denotes sugar consumption per capita, which is 9.25 kg/capita/year (Food Security Agency, 2017); and Pt is the total population in year t (people) (Murdaningsih et al., 2017). following measuring of sugar requirements in 2031, the next step was to compute the sugar balance in 2031 using Rejekiningrum's (2013) formula:

$$S/D = SP_t - SC_t$$

where S/D stands for surplus/deficit; SPt stands for sugar production at time t (tons), which is calculated by multiplying the area of sugarcane plantations (ha) resulting from the classification and projection of LULC by the average productivity value of sugarcane plantations in Lamongan Regency, which is 70.66 tons/ha (Lamongan Statistics, 2020); and SCt stands for sugar consumption at time t (tons).

3 RESULTS

3.1 Land-Use/Land-Cover Classification of 2007, 2013, and 2019

The classification of LULC was carried out to ascertain the state of the current LULC in the Lamongan Regency. The classification of LULC in Lamongan Regency using SPOT images in 2007, 2013, and 2019 demonstrated that LULC has comprised of forest, fields/moors, built-up areas, open land, sugarcane plantations, rice fields, shrubs, fishponds, and water bodies. The accuracy procedure was conducted to test the accuracy of the findings of the LULC classification. The accuracy test was conducted using 72 points. Each LULC type had eight sample points. The kappa accuracy-test

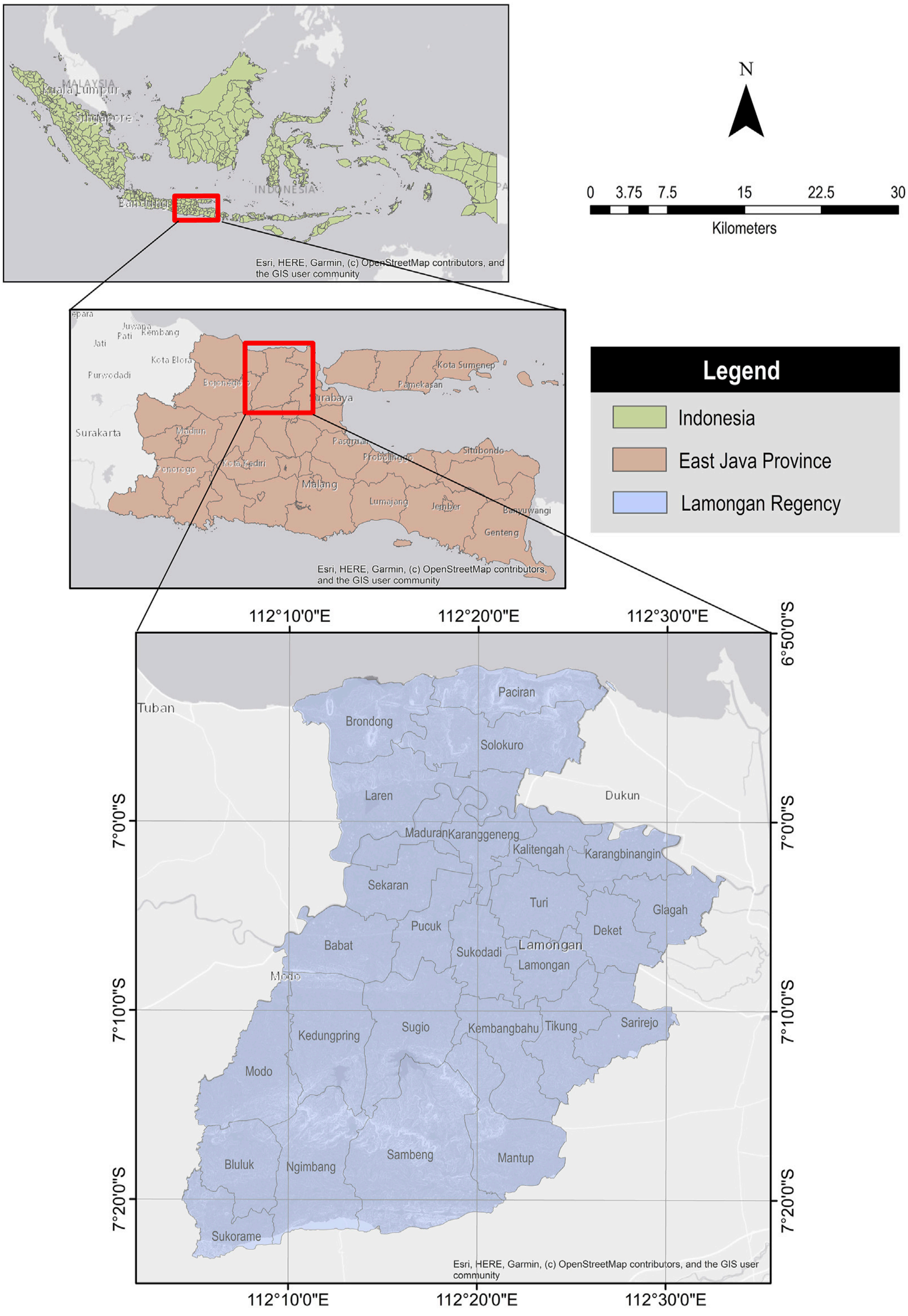

FIGURE 1 | The study area map.

values were above 0.84 (2007), 0.82 (2013), and 0.86 (2019) (**Supplementary Table S1**), illustrating that the LULC classification was acceptable because the kappa accuracy-test values were above the minimum level of nearly perfect agreement (very high accuracy) of 0.81 (Rwanga and Ndambuki, 2017).

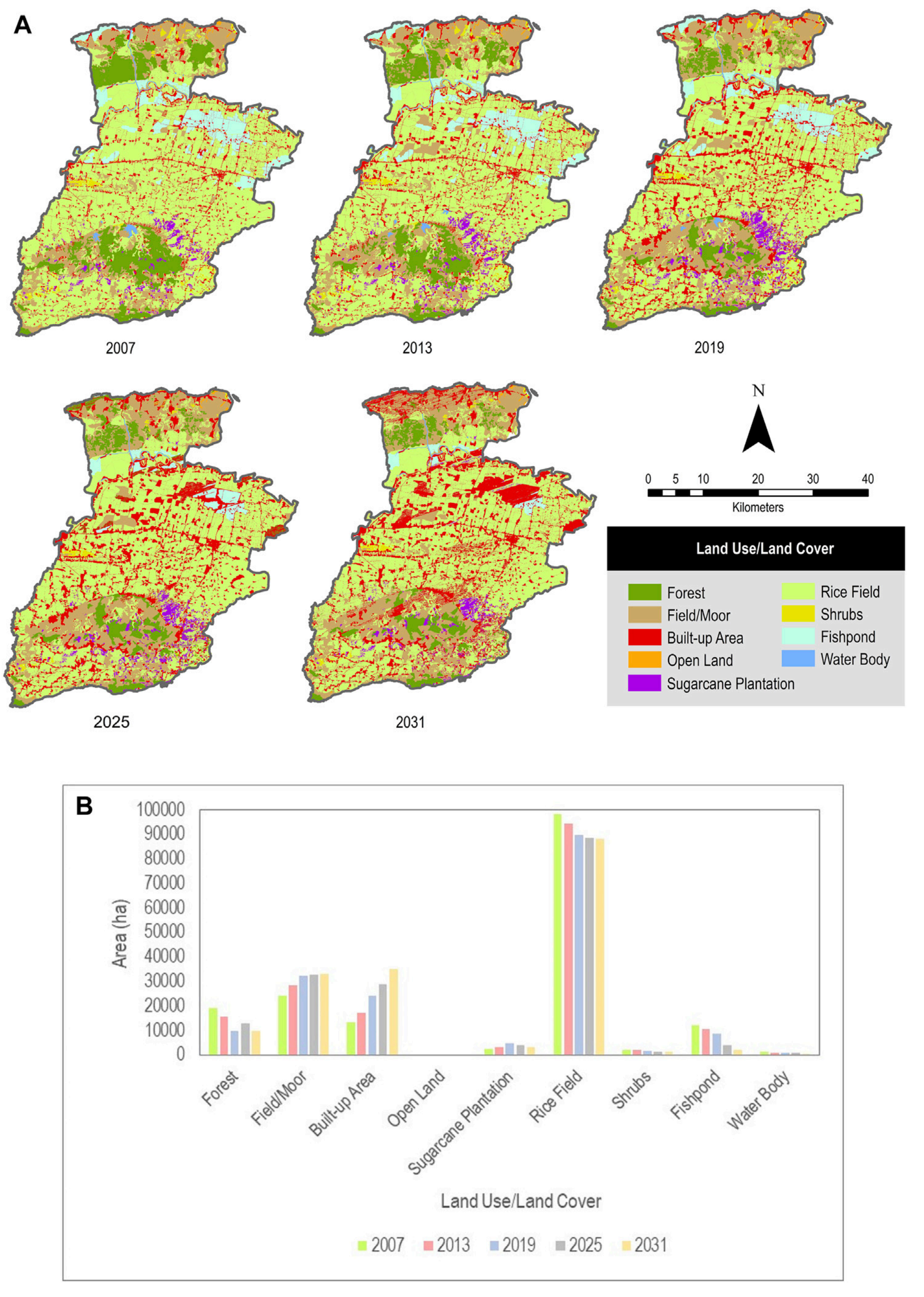

FIGURE 2 | The distribution of LULC in Lamongan Regency in 2007, 2013, 2019, and the anticipated results for LULC in Lamongan Regency in 2025 and 2031 **(A)**; graph of change of each LULC **(B)**.

From 2007 to 2019, there was a change in the LULC classification, as demonstrated by the annual changes in the area of each every year (**Figure 2**). Land uses/land cover that has reduced in the area from 2007 to 2019 include forests, rice fields, shrubs, fishponds, and water bodies. The greatest reduction occurred in the forest area, which decreased by 9,080.4 ha (47.41%). Between 2007 and 2019, the forest turned into fields/moor at most, which was 9,429.8 ha, but there was also an increase in forest area from the previous fields/moor, which totaled 757.2 ha (**Supplementary Table S2**). In general, Indonesian forests are decreasing as agricultural land is converted (Austin et al., 2019). Meanwhile, the area built-up land increased the most, which totaled 10,780.2 ha. The increase in the built-up land was mostly due to the addition of 7,508.5 ha of rice fields. Rice fields were converted to built-up land as a result of the development of infrastructure, housing, and economic activity centers in numerous regions, all of which were a result of Lamongan Regency's population growth. Sugarcane plantations increased by 2,298.9 ha because in 2015 there was a new sugar factory built in Lamongan Regency, but starting in 2020 the plantation area began to decline again (Lamongan Statistics, 2020).

3.2 Prediction Model of Land Use/Land Cover in Lamongan Regency in 2031

Land-Use/Land-Cover Change Model

The LULCC model in this research incorporates input in the form of driving factors affecting land-use/land-cover changes in the modeling. Natural variables such as slope, climate, elevation, hydrological effects, construction costs, and forest distribution play a major role in LULCC (Bajocco et al., 2016; Zhao et al., 2018; Zhu et al., 2021). According to the analysis's results, the driving factor, as expressed as slope, has a substantial impact on all types of LULC. The slope is positively associated with forest, filed/moor, open land, sugarcane plantation, and shrubs, but negatively associated with the built-up area, rice field, fishpond, and water body. Distance from the road has a considerable negative influence on the field/moor and rice fields and fishpond, but a large favorable effect on the fishpond. Rainfall has a large positive influence on the forest, sugarcane plantation, and water bodies but has a significant negative effect on field/moor, built-up area, and open land. The temperature has a significant positive impact on the forest, sugarcane plantation, rice fields, fishpond, and water body but has a significant negative effect on field/moor, open land, and shrubs (**Supplementary Table S3**).

Model Validation

The kappa value generated from this study's validation of the LULC prediction model was 0.81 (**Supplementary Table S4**). As a result of the categorization and range of kappa value classes, the model generated in this study has a near-perfect level of similarity with the current LULC conditions in 2019 as a result of the classification (Alkaradaghi et al., 2018). Therefore, the finding model can be used to project LULC over the next years. In other words, this research is capable of predicting the distribution of LULC in 2031 in Lamongan Regency.

Land-Use/Land-Cover Prediction in 2031 in Lamongan Regency

The LULC prediction for 2031 is based on a "business as usual" scenario. Along with 2031, a projection for 2025 was created to obtain changes during a 6-year interval. LULC maps from 2007 to 2013, and 2019 were utilized as inputs. The LULC modeling of CLUE-s conversion demands that the elasticity of LULC conversion be specified. The elasticity value is a characteristic that indicates whether or not the land may be used for another LULC. The elasticity value of LULC conversion has a range of 0–1. The closer the value to zero, the easier it is to convert the land; on the other hand, the closer the value to one, the more difficult it is to convert the land (Luo et al., 2010). Each form of LULC has an elasticity value that is adjusted to the conditions of land-use/land-cover changes that occur in a given area (Murdaningsih et al., 2017). The elasticity of 0.7 for the forest, 0.5 for fields/moor, 1.0 for built-up areas, 0.3 for open land, 0.7 for sugarcane plantations, 0.6 for rice fields, 0.6 for shrubs, 0.5 for fishponds, and 0.9 for water bodies were utilized in this study. The distribution of LULC in Lamongan Regency in 2007, 2013, 2019, as well as prediction, projected findings for LULC in Lamongan Regency in 2025 and 2031 are depicted in **Figure 2**.

Based on the prediction results, between 2019 and 2031, Lamongan Regency is expected to experience an increase in the area of field/moor and built-up areas. The area covered by built-up areas increase the most by 10,881.6 ha. Meanwhile, it is anticipated that there is the extent of forest, open land, sugarcane plantations, rice fields, shrubs, ponds, and bodies of water. Rice fields will lose most of the area, totaling 1,687.5 ha. The reduced rice fields are predicted to be converted to built-up land covering an area of 2,632.7 ha. The area and percentage of each LULC in 2019–2031 and their changes are shown in **Table 1**.

3.3 Sugar Balance of Lamongan Regency in 2031

Based on the results of the LULC classification and predictions, the area of sugarcane plantations between 2019 and 2031 decreased by 1,687.5 ha which may have contributed to the loss in the sugar balance in Lamongan Regency, as the population was predicted to increase while the output of sugarcane decreased. The total population of Lamongan Regency total population in 2019 was 1,373,390 and is anticipated to reach 1,419,843 in 2031. Lamongan Regency's sugar production in 2019 was 27,357.39 tons. According to the projections, the value of sugar production in the Lamongan Regency in 2031 will reach 17,316.32 tons (**Figure 3**). This means that sugar production in Lamongan Regency in 2031 compared to production in 2019 is expected to fall by 10,470.76 tons. Sugar balance projections for 2031 showed that Lamongan Regency will have a total sugarcane surplus of 4,182.77 tons. According to the analysis's results, Lamongan Regency had a sugar surplus in 2007, 2013, 2019, 2025, and 2031 but the surplus are expected to decrease in 2025 and 2031 due to the decline in the area of sugarcane plantations, which is influenced by changes in land use from sugarcane plantations to other land-uses, particularly built-up areas. Sugarcane plantations, it is predicted, will eventually be unable to compete with built-up areas in the future. This is because the built-up area in Lamongan Regency has an elasticity value of one, making it more competitive than sugarcane plantations with an elasticity value of 0.7.

4 DISCUSSIONS

4.1 Sugar Balance as the Impact of Land-Use/Land-Cover Change Dynamics

In this study, there was a land-use/land-cover change in Lamongan Regency in 2007–2019 which has caused the greatest loss in the forest. Human activities and population growth influence land-use/land-cover change (Awotwi et al.,

TABLE 1 | Area and percentage of each land-use/land-cover in 2019–2031 and their changes.

No	LULC	2007		2013		2019	
		Area (ha)	Percentage (%)	Area (ha)	Percentage (%)	Area (ha)	Percentage (%)
1	Forest	19,153.2	11.02	15,627.9	8.99	10,072.8	5.80
2	Field/Moor	24,257.2	13.96	28,358.9	16.32	32,437.1	18.67
3	Built-up Area	13,518.0	7.78	17,405.4	10.02	24,298.2	13.98
4	Open Land	310.2	0.18	316.9	0.18	334.0	0.19
5	Sugarcane Plantation	2,571.2	1.48	3,476.3	2.00	4,870.1	2.80
6	Rice Field	98,346.5	56.60	94,419.0	54.34	89,935.0	51.76
7	Shrubs	2,217.1	1.28	2,168.7	1.25	1,909.3	1.10
8	Fishpond	12,091.0	6.96	10,840.0	6.24	8,818.3	5.07
9	Water Body	1,301.8	0.75	1,153.1	0.66	1,091.3	0.63
Total		173,766.1	100.00	173,766.1	100.00	173,766.1	100.00
No	**LULC**	**2025**		**2031**			
		Area (ha)	**Percentage (%)**	**Area (ha)**	**Percentage (%)**		
1	Forest	12,925.6	7.44	9,785.3	5.63		
2	Field/Moor	32,916.9	18.94	33,174.8	19.09		
3	Built-up Area	28,787.0	16.57	35,179.8	20.25		
4	Open Land	274.9	0.16	238.8	0.14		
5	Sugarcane Plantation	4,151.6	2.39	3,182.6	1.83		
6	Rice Field	88,603.0	50.99	88,097.9	50.70		
7	Shrubs	1,346.8	0.78	1,292.3	0.74		
8	Fishpond	3,920.9	2.26	2,187.1	1.26		
9	Water Body	839.3	0.48	627.5	0.36		
Total		173,766.1	100.00	173,766.1	100.00		

2019; Hailu et al., 2020; Mengist et al., 2021). The population growth stimulates the conversion of LULC from forests to agricultural land as the population grows, so will the diverse needs. This encourages the community, especially farmers, to encroach on the forest and convert it into agricultural land to suit their needs. Meanwhile, the built-up area tends to continue to develop. The built-up area increase the most, from 10,780.2 ha in 2007–2019. The phenomenon of urbanization and population growth contributes to the expansion of built-up areas and changes in spatial structure in suburban areas (Surya et al., 2021).

Based on these conditions, we have modeled the LULCC, including driving factors and elasticity settings, to forecast LULC in 2031. Based on the results of the investigation, it is predicted that the built-up areas will see the most development, reducing the area of sugarcane plantations to 1,687.5 ha. Although sugarcane plantations increased from 2007 to 2019, the annual increase rate of sugarcane plantation areas is still substantially lower than the increase rate of built-up area. The built-up area increased by 898.4 ha on average in 2007–2019, whereas sugarcane plantations increased by only 191.6 ha on average. Due to the built-up area's huge growth, areas that previously had the potential to see a rise in sugarcane plantations finally overlapped with built-up areas. The built-up areas have been expanded to sugarcane plantations because the area is close to the inter-district road so it is very massive to experience development. Furthermore, there has been competition among LULCs. The built-up area has an elasticity of one, makes it difficult to convert to other LULCs, whereas sugarcane plantations have an elasticity of 0.7, allowing sugarcane plantations to be converted to other LULCs, and allowing conversion from sugarcane plantations to built-up areas to be possible in 2031.

According to the predicted drop in sugarcane plantations in 2031 as a result of competition from built-up areas, sugar production in Lamongan Regency is still quite susceptible to falling in the future. Lamongan Regency had a sugar surplus in 2031 of 4,182.77 tons but the surplus is expected to decrease compared to previous years due to the decline in the area of sugarcane plantations which caused a reduction in sugar production. If the reduction is not countered by prevention initiatives, it is believed that sugar self-sufficiency in Lamongan Regency will be jeopardized in the future. As a result of this, the growth of sugarcane plantations is critical in Lamongan Regency. The development of sugarcane plantations in Lamongan Regency is expected to continue to support sugar self-sufficiency in Lamongan Regency and contribute to the fulfillment of sugar needs in East Java and throughout the country.

4.2 Managerial Implication

The regional spatial plan of Lamongan Regency makes sugarcane the main plantation commodity to be developed, but currently, there is no specific map for sugarcane, but only for plantations in general (**Supplementary Figure S3**). This study is expected to be an input for the government in formulating policies for developing sugarcane plantations and controlling LULCC in the future. Apart from promoting regional and national self-sufficiency in sugarcane and sugar, the development of sugarcane plantations has the potential to improve people's welfare. According to Anam and Qibtiyah (2018), sugarcane plantations are possible to cultivate in Lamongan based on the R-C ratio analysis. The existence of sugarcane plantations in Lamongan Regency has also facilitated the establishment of the Sugarcane Farmer Cooperative (KPTR) which can support economic development (Perdana and Nasution, 2016). The establishment of sugarcane farmer cooperatives the acquisition of

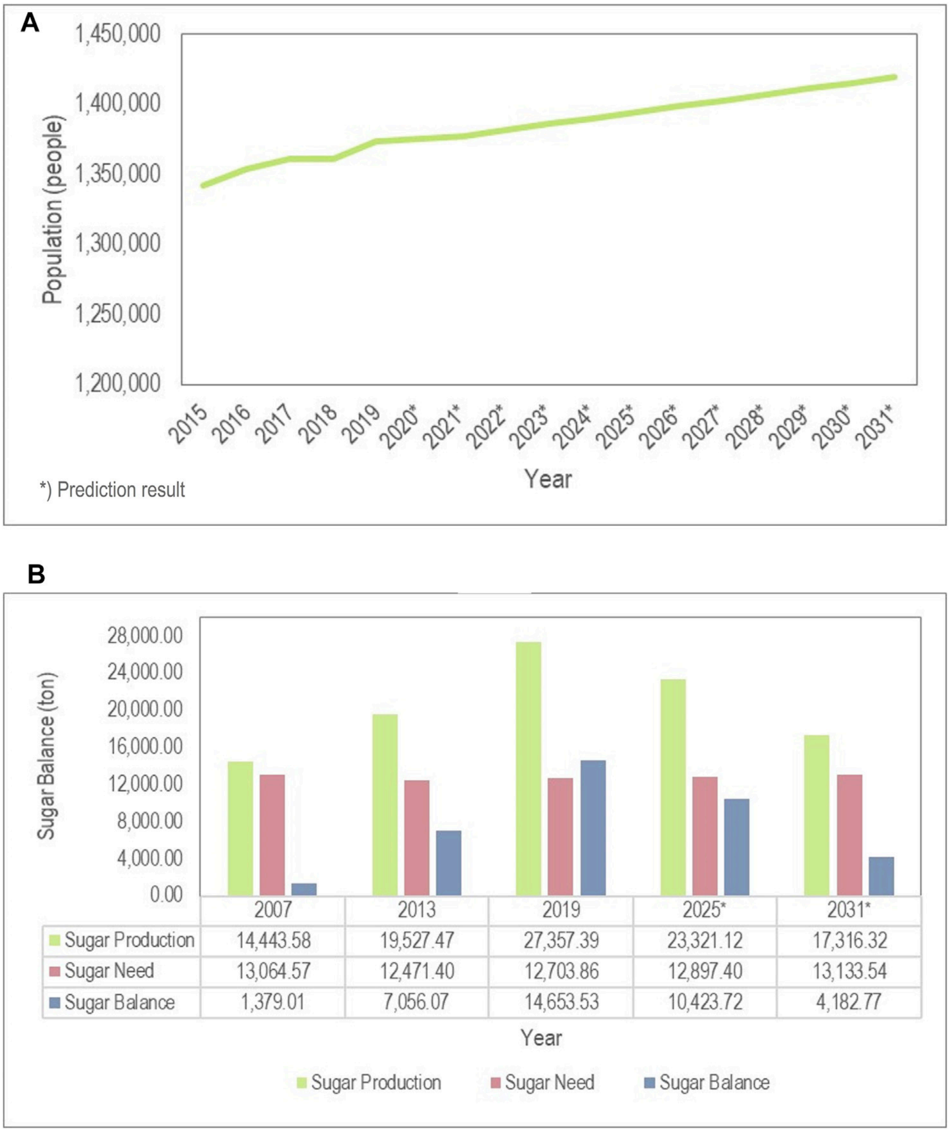

FIGURE 3 | The population projection result in Lamongan Regency until 2031 **(A)**; the sugar balance in 2007, 2013, 2019, 2025, and 2031 **(B)**.

capital and production facilities, the conduct of savings and loans activities, and the improvement of coordination among sugarcane farmers.

5 CONCLUSIONS

The land-use/land-cover patterns in Lamongan Regency were highly diversified from 2007 to 2019. Changes in LULC are predicted to lead to a drop in sugarcane plantation areas in 2025 and 2031 as a result of competition between built-up areas and sugarcane plantation growth. The built-up area has grown rapidly in line with the population growth of Lamongan Regency in 2031, resulting in the built-up area being the most competitive when compared to other LULC, such as sugarcane plantations. The elasticity of LULC conversion demonstrates the conflict between built-up areas and sugarcane. The built-up area is elastic of one, while sugarcane plantations are elastic of 0.7.

The estimated area of sugarcane plantation in 2031 can be transformed into sugarcane production in 2031, allowing for the determination of the sugar balance in 2031. The dynamics of land-use/land-cover change in 2007–2031 and population growth are projected to have an impact on the sugar balance in Lamongan Regency in 2031. The population is expected to grow, but sugarcane production is expected to decline, leading to the sugar balance in 2031 being predicted to decrease by 10,470.76 tons compared to 2019. This may jeopardize regional and national sugar sufficiency, so a strategy is required to secure sugar sufficiency in the future, including the development of sugarcane plantations and control of the built-up area in the Lamongan Regency.

DATA AVAILABILITY STATEMENT

The original contributions presented in the study are included in the article/**Supplementary Material**, further inquiries can be directed to the corresponding author.

AUTHOR CONTRIBUTIONS

SDA collected the data and wrote the initial article. Widiatmaka is the project leader who provided funding for the research, as well as provided the methods used. YS provided land-use/land-cover modeling methods. Meanwhile, Marimin provided the discussion and revision.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fenvs.2022.787207/full#supplementary-material

REFERENCES

Adhiatma, R., Widiatmaka, fnm., and Iskandar Lubis, fnm. (2020). Perubahan penggunaan/ tutupan lahan dan prediksi perubahan penggunaan/ tutupan lahan di Kabupaten Lampung Selatan. *Jpsl* 10 (2), 234–246. doi:10.29244/jpsl.10.2.234-246

Al-Eideh, B. M., and Al-Omar, H. O. (2019). Population Projection Model Using Exponential Growth Function with a Birth and Death Diffusion. *Eur. J. Sci. Res.* 151 (3), 271–276. Available at: : http://www.europeanjournalofscientificresearch.com.

Alkaradaghi, K., Ali, S. S., Al-Ansari, N., and Laue, J. (2018). Evaluation of Land Use & Land Cover Change Using Multi-Temporal Landsat Imagery: A Case Study Sulaimaniyah Governorate, Iraq. *Jgis* 10 (03), 247–260. doi:10.4236/jgis.2018.103013

Anam, C., and Qibtiyah, M. (2018). Research on the Potential of Sugarcane Plantations on Community Welfare in Lamongan Regency. *Praja Lamongan* 1 (1), 46–54.

Andries, A., Murphy, R. J., Morse, S., and Lynch, J. (2021). Earth Observation for Monitoring, Reporting, and Verification within Environmental Land Management Policy. *Sustainability* 13, 9105–9134. doi:10.3390/su13169105

Austin, K. G., Schwantes, A., Gu, Y., and Kasibhatla, P. S. (2019). What Causes Deforestation in Indonesia. *Environ. Res. Lett.* 14 (2), 1–9. doi:10.1088/1748-9326/aaf6db

Awotwi, A., Anornu, G. K., Quaye-Ballard, J. A., Annor, T., Forkuo, E. K., Harris, E., et al. (2019). Water Balance Responses to Land-Use/land-Cover Changes in the Pra River Basin of Ghana, 1986-2025. *Catena* 182, 104129. doi:10.1016/j.catena.2019.104129

Bajocco, S., Ceccarelli, T., Smiraglia, D., Salvati, L., and Ricotta, C. (2016). Modeling the Ecological Niche of Long-Term Land Use Changes: The Role of Biophysical Factors. *Ecol. Indicators* 60, 231–236. doi:10.1016/j.ecolind.2015.06.034

Clarke, K. C., and Gaydos, L. J. (1998). Loose-coupling a Cellular Automaton Model and GIS: Long-Term Urban Growth Prediction for San Francisco and Washington/Baltimore. *Int. J. Geographical Inf. Sci.* 12, 699–714. doi:10.1080/136588198241617

Cohen, J. (1960). A Coefficient of Agreement for Nominal Scales. *Educ. Psychol. Meas.* 20 (1), 37–46. doi:10.1177/001316446002000104

Debeko, D., Angassa, A., Abebe, A., Burka, A., and Tolera, A. (2018). Human-climate Induced Drivers of Mountain Grassland over the Last 40 Years in Sidama, Ethiopia: Perceptions versus Empirical Evidence. *Ecol. Process.* 7 (34), 2–16. doi:10.1186/s13717-018-0145-5

Food Security Agency (2017). *Food Security Statistics 2017*. Jakarta: Food Security Agency.

Foody, G. M. (2002). Status of Land Cover Classification Accuracy Assessment. *Remote Sensing Environ.* 80, 185–201. doi:10.1016/S0034-4257(01)00295-4

Getachew, N., and Meten, M. (2021). Weights of Evidence Modeling for Landslide Susceptibility Mapping of Kabi-Gebro Locality, Gundomeskel Area, Central Ethiopia. *Geoenviron Disasters* 8 (6), 2–22. doi:10.1186/s40677-021-00177-z

Hailu, A., Mammo, S., and Kidane, M. (2020). Dynamics of Land Use, Land Cover Change Trend and its Drivers in Jimma Geneti District, Western Ethiopia. *Land use policy* 99, 105011. doi:10.1016/j.landusepol.2020.105011

Hatami, M., and Shafieardekani, M. (2014). The Effect of Industrialization on Land-Use Changes; Evidence from Intermediate Cities of Iran. *Int. J. Curr. LIFE Sci. ISSN* 4 (12), 11899–11902.

Indonesian Statistics (2010). *Guidelines for Calculating Population and Labor Force Projections*. Jakarta: Indonesian Statistics.

Indonesian Statistics (2019). *Indonesian Sugarcane Statistics 2018*. Jakarta: Indonesian Statistics.

Indonesian Statistics (2020). *Indonesian Sugarcane Statistics 2019*. Jakarta: Indonesian Statistics.

Islam, S., Li, Y., Ma, M., Chen, A., and Ge, Z. (2021). Simulation and Prediction of the Spatial Dynamics of Land Use Changes Modelling through CLUE-S in the Southeastern Region of Bangladesh. *J. Indian Soc. Remote Sens.* 49, 2755–2777. doi:10.1007/s12524-021-01402-w

Lamongan Statistics (2020). *Lamongan Regency in Numbers*. Lamongan: Lamongan Statistics.

Landis, J. R., and Koch, G. G. (1977). The Measurement of Observer Agreement for Categorical Data. *Biometrics* 33 (1), 159–174. doi:10.2307/2529310

Leta, M. K., Demissie, T. A., and Tränckner, J. (2021). Modeling and Prediction of Land Use Land Cover Change Dynamics Based on Land Change Modeler (LCM) in Nashe Watershed, Upper Blue Nile Basin, Ethiopia. *Sustainability* 13, 3740. doi:10.3390/su13073740

Liao, Q., Li, M., Chen, Z., Shao, Y., and Yang, K. (2010). "Spatial Simulation of Regional Land Use Patterns Based on GWR and CLUE-S Model," in 2010 18th International Conference on Geoinformatics, Beijing, China, 18-20 June 2010. doi:10.1109/geoinformatics.2010.5567963

Luo, G., Yin, C., Chen, X., Xu, W., and Lu, L. (2010). Combining System Dynamic Model and CLUE-S Model to Improve Land Use Scenario Analyses at Regional Scale: A Case Study of Sangong Watershed in Xinjiang, China. *Ecol. Complexity* 7 (2), 198–207. doi:10.1016/j.ecocom.2010.02.001

Marpaung, Y. T. F., Hutagaol, P., Limbong, W., and Kusnadi, N. (2011). The Development of the Indonesian Sugar Industry and the Urgence of National Self-Support. *Indones J. Agric. Econ.* 2 (1), 1–14.

Mekonnen, Y. A. (2018). Population Forecasting for Design of Water Supply System in Injibara Town, Amhara Region, Ethiopia. *Civ Environ. Res.* 10, 54–65. Available at: www.iiste.org.

Mengist, W., Soromessa, T., and Feyisa, G. L. (2021). Monitoring Afromontane forest Cover Loss and the Associated Socio-Ecological Drivers in Kaffa Biosphere reserve, Ethiopia. *Trees, Forests and People* 6, 100161. doi:10.1016/j.tfp.2021.100161

Mhawish, Y. M., and Saba, M. (2016). Impact of Population Growth on Land-Use Changes in Wadi Ziqlab of Jordan between 1952 and 2008. *Int. J. Appl. Sociol.* 6 (1), 7–14. doi:10.5923/j.ijas.20160601.02

Moulds, S., Buytaert, W., and Mijic, A. (2015). An Open and Extensible Framework for Spatially Explicit Land Use Change Modelling: the Lulcc R Package. *Geosci. Model. Dev.* 8 (10), 3215–3229. doi:10.5194/gmd-8-3215-2015

Murdaningsih, W., Munibah, K., and Ambarwulan, W. (2017). Spatial Analysis of Changes in Agricultural Land Use in Indramayu Regency. *Maj Ilm Globë.* 19 (2), 175–184. doi:10.24895/mig.2017.19-2.604

Peerzado, M. B., Magsi, H., and Sheikh, M. J. (2019). Land Use Conflicts and Urban Sprawl: Conversion of Agriculture Lands into Urbanization in Hyderabad, Pakistan. *J. Saudi Soc. Agric. Sci.* 18 (4), 423–428. doi:10.1016/j.jssas.2018.02.002

Perdana, Y., and Nasution, N. (2016). Lamong Jaya Sugarcane Farmer's Cooperative 1999 – 2004. *Avatara* 4 (1), 29–35.

Rejekiningrum, P. (2013). Rice Surplus Optimization Model to Determine the Level of National Food Security. *Pros Semin. Nas Mat Sains, Dan Teknol.* 4, 62–75.

Rondhi, M., Pratiwi, P., Handini, V., Sunartomo, A., and Budiman, S. (2018). Agricultural Land Conversion, Land Economic Value, and Sustainable Agriculture: A Case Study in East Java, Indonesia. *Land* 7, 148. doi:10.3390/land7040148

Rwanga, S. S., and Ndambuki, J. M. (2017). Accuracy Assessment of Land Use/land Cover Classification Using Remote Sensing and GIS. *Ijg* 08, 611–622. doi:10.4236/ijg.2017.84033

Setiawan, Y., and Yoshino, K. (2020). Spatial Modeling on Land Use Change in Regional Scale of Java Island Based-On Biophysical Characteristics. *Jpsl* 10 (3), 511–523. doi:10.29244/jpsl.10.3.511-523

Siagian, D. R., Shrestha, R. P., Shrestha, S., and Kuwornu, J. K. M. (2019). *Factors Driving rice Land Change 1989 – 2018 in the Deli Serdang Regency*. Indonesia: Agriculture 9, 186. doi:10.3390/agriculture9090186

Sinurat, T. P., Munibah, K., and Baskoro, D. P. T. (2015). Pemodelan Perubahan Penggunaan Lahan Kabupaten Humbang Hasundutan Menggunakan Clue-S. *J. Tanah, Lingk,* 17 (2), 75–82. doi:10.29244/jitl.17.2.75-82

Song, K., Choi, Y.-E., Han, H.-J., and Chon, J. (2021). Adaptation and Transformation Planning for Resilient Social-Ecological System in Coastal Wetland Using Spatial-Temporal Simulation. *Sci. Total Environ.* 789, 148007. doi:10.1016/j.scitotenv.2021.148007

Surya, B., Salim, A., Hernita, H., Suriani, S., Menne, F., and Rasyidi, E. S. (2021). Land-use Change, Urban Agglomeration, and Urban Sprawl: a Sustainable Development Perspective of Makassar City, Indonesia. *Land* 10, 1–31. doi:10.3390/land10060556

Tali, A. J., and Murthy, K. (2013). Influence of Urbanization on the Land-Use Change: A Case Study of Srinagar City. *Am. J. Res. Commun.* 1 (17), 271–283.

Veldkamp, A., and Fresco, L. O. (1996). CLUE: a Conceptual Model to Study the Conversion of Land Use and its Effects. *Ecol. Model.* 85, 253–270. doi:10.1016/0304-3800(94)00151-0

Verburg, P. H., Soepboer, W., Veldkamp, A., Limpiada, R., Espaldon, V., and Mastura, S. S. A. (2002). Modeling the Spatial Dynamics of Regional Land Use: The CLUE-S Model. *Environ. Manage.* 30 (3), 391–405. doi:10.1007/s00267-002-2630-x

Verburg, P. (2010). *The CLUE Model.* Amsterdam: University Amsterdam.

Wear, D. N., and Flamm, R. (1993). Public and Private forest Disturbance Regimes in the Southern Appalachians. *Nat. Resource Model.* 7 (4), 379–397. doi:10.1111/j.1939-7445.1993.tb00159.x

Wei, H., Lu, C., and Liu, Y. (2021). Farmland Changes and Their Ecological Impact in the Huangshui River Basin. *Land* 10, 1082–1115. doi:10.3390/land10101082

Wood, E. C., Lewis, J. E., Tappan, G. G., and Lietzow, R. W. (1997). "The Development of a Land Cover Change Model for Southern Senegal," in *Land Use Modeling Workshop* (Sioux Falls, SD. Santa Barbara, CA: National Center for Geographic Information and Analysis).

Xu, K., Chi, Y., Ge, R., Wang, X., and Liu, S. (2021). Land Use Changes in Zhangjiakou from 2005 to 2025 and the Importance of Ecosystem Services. *PeerJ* 9, e12122. doi:10.7717/peerj.12122

Yu, J., Hagen-zanker, A., Santitissadeekorn, N., and Hughes, S. (2021). Calibration of Cellular Automata Urban Growth Models from Urban Genesis Onwards - a Novel Application of Markov Chain Monte Carlo Approximate Bayesian Computation. *Comput. Environ. Urban Syst.* 90, 101689. doi:10.1016/j.compenvurbsys.2021.101689

Zhao, X., Pu, J., Wang, X., Chen, J., Yang, L., and Gu, Z. (2018). Land-use Spatio-Temporal Change and its Driving Factors in an Artificial forest Area in Southwest China. *Sustainability* 10, 4066. doi:10.3390/su10114066

Zhu, B., Liao, J., and Shen, G. (2021). Spatio-Temporal Simulation of Mangrove Forests under Different Scenarios: A Case Study of Mangrove Protected Areas, Hainan Island, China. *Remote Sensing* 13, 4059–4113. doi:10.3390/rs13204059

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Policy Allocation for Settlement Development Using Simple Allocation Matrix Rules and Geographic Information System

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The settlement growth is potentially increased by the rural-to-urban perspective change due to the new era of aerotropolis in Kulon Progo. Land-use planning evaluation is required, especially settlement, which has a significant impact on the environment. However, land-use evaluation studies are currently focused on conforming analysis of official land-use planning (OLUP) toward existing or predicted land use partially or in combination with the performance assessment unit. Consequently, it affects the quality of policy products by disregarding crucial considerations of diverse conditions at points of time and aspects of ideality, reality, and regulation. Therefore, the objective of this study was to design a comprehensive policy allocation for settlements using a matrix allocation rule that integrates conformity and performance analysis in three aspects of planning simultaneously. Land allocation was proposed using a geographic information system (GIS) of land capability, settlement suitability, and agricultural and forest land protection. The current land use/land cover (LULC) was classified using visual interpretation of SPOT 7 satellite imagery and a multi-layer perceptron neural network (MLPNN) to predict the LULC in 2035. The result indicated that the stock of land allocation for settlement development is sufficient to meet the demands in 2035. However, there is a problem in the settlement distribution pattern in which 64.3% of existing settlements are located in non-recommended allocation. That number is predicted to increase by 1,145.8 ha. Land-use control instruments need to be conducted to prevent extensive settlement growth in non-recommended allocation. Conversely, zoning allocation should be directed to trigger the growth of settlements in recommended allocation.

Keywords: evaluation assessment, simple decision matrix, spatial planning, urban growth, zoning regulation

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1 INTRODUCTION

Kulon Progo faces an aerotropolis era of regional development due to the construction of Yogyakarta International Airport (YIA). Aerotropolis is a concept of city development centered on the airport (airport city) and emphasizes the surrounding aviation-linked business and residential areas (Kasarda, 2019). The emergence of new airports creates accessibility (Tveter, 2017; Aguirre et al., 2019; Fernandes et al., 2019; Hubbard et al., 2019) and offers added value to the region

(Blonigen and Cristea, 2015). Accessibility could trigger development and attract population growth (Baum-Snow, 2010; Chi, 2012; McGraw, 2020). The increasing activity intensity and the growing population stimulate built-up growth, especially settlements (He and Xie, 2019); land use accommodates urban activities such as trade, housing, and services.

The progress in the dynamics of urban the growth of build-up area potentially causes sprawl, which is a big challenge in realizing sustainable development (Bovet et al., 2018). Urban sprawl brings physical impact such as increased emissions (Han, 2020), mass consumption of resources (He et al., 2017), numerous changes in the structure and functioning of landscape (Solon, 2009), conversion of agricultural and forests (Kurnianti et al., 2015; Rustiadi et al., 2020), and socioeconomic problems (Giyarsih, 2010) such as the emergence of slum areas, poverty, and unemployment (Wang and Maduako, 2018).

The evaluation of land-use planning, especially settlements, should be conducted in the early phase of the aerotropolis, considering there is perspective change on regional development to mitigate the emergence of urban sprawl. Land-use planning is an important instrument to prevent urban sprawl (Wilson and Chakraborty, 2013; Bovet et al., 2018). However, the current study (Kassis et al., 2021; Mallma, 2021) has focused its evaluation only on the official land-use plan (OLUP) using performance indicators and on the existing land use (Lyu et al., 2022) to provide recommendations on land-use planning, or evaluated predicted land use in every scenario (Shi et al., 2021; Nijhum et al., 2021) separately. The study of Widiatmaka et al. (2016a) combines the evaluation of OLUP and presents land use using performance analysis. However, the analysis has not included future considerations and has no access to the planning instruments to conduct the policy. The detached analysis has shortcomings that affect the quality of policy product since it did not consider key information at different points in time. Moreover, the policy recommendation of the study that is not linked to the land-use planning instruments is probably difficult to operationalize. Consequently, the evaluation should integrate performance and conformance analysis of the present and future conditions, as well as regulation to provide a policy that accommodates the ideality, reality, and regulation aspect.

A simple decision matrix provides an integrated analysis of policy generations. The simple decision model has the robustness to conduct a policy that involves multiple stakeholders on frequently repeated decisions compared to the complex ones (Katsikopoulos et al., 2018). It is a suitable tool to generate recommendations on spatial planning policy in Indonesia, which is regularly conducted every 5 years with the participatory paradigm involving multiple stakeholders. The geographic information system (GIS) helps analyze the data input by arranging the policy spatially. GIS application has been widely used in spatial planning (AbuSada and Thawaba, 2011; Omar and Raheem, 2016; Ustaoglu and Aydınoglu, 2020). Therefore, the objective of this study was to design policy allocation for settlement development comprehensively using a matrix allocation rule and a geographic information system that integrates unit analysis of evaluation, existing and predicted land use, and the OLUP simultaneously to conduct remediation in OLUP.

2 MATERIALS AND METHODS

2.1 Study Area

The study was located in Kulon Progo Regency, Province of Yogyakarta, Indonesia, with a land area of 57,326.1 ha and a total population of 436,395 in 2020 (Statistic Bureau of Kulon Progo, 2021). The airport development is under the national strategic projects mentioned in Presidential Regulation Number 3 of 2016 and 98 of 2017. The airport construction entered its first phase in 2016–2021, which could accommodate 10 million passengers per year. The ultimate capacity can accommodate 20 million passengers per year, by 2031–2041 (Minister of Transportation, 2013).

Kulon Progo is classified into three types of development areas. First, the north cluster that has an undulating topography with the existence of Menoreh Hills. The Bedah Menoreh project that aims to develop geotourism and create a nature-based tourism village (Local Government Regulation No 2 of 2021) is indicated by the yellow line in **Figure 1**. Second, the east cluster has a hilly topography accentuated by the new development of the Sentolo industrial estate according to Kulon Progo Regional Spatial Plan 2012–2032, as indicated by the shaded area in **Figure 1**. The third is the south cluster of Kulon Progo, which is an area with flat topography. The cluster has become the center of settlement growth and infrastructure development.

2.2 Data Collection

The data of SPOT 4 of 2010, SPOT 6 of 2015, and SPOT 7 of 2020 satellite imageries were obtained through Indonesia's National Institute of Aeronautics and Space. Furthermore, base maps and infrastructure facilities were obtained from the Geospatial Information Agency. OLUP data for Kulon Progo 2012–2032 and forest area were downloaded through http://geoportal.kulonprogokab.go.id/. The soil map with a 1:50,000 scale was acquired from the Indonesian Center for Agricultural Land Resources Research and Development. Social, economic, and population data were collected through Statistics Indonesia, while the primary data were obtained through ground check and AHP questionnaire.

2.3 Methods

The research dissects three processes: analysis of present and future condition of land uses, analysis of the land allocation using multi-criteria decision analysis and a geographic information system, and arrangement of the policies using a matrix allocation rule.

2.3.1 Existing and Land-Use Prediction

Visual interpretation by on-screen digitizing based on the interpretation key (Lillesand et al., 2015) was conducted to build land use/land cover (LULC) at three time points: 2010, 2015, and 2020. Meanwhile, land use was classified using SNI

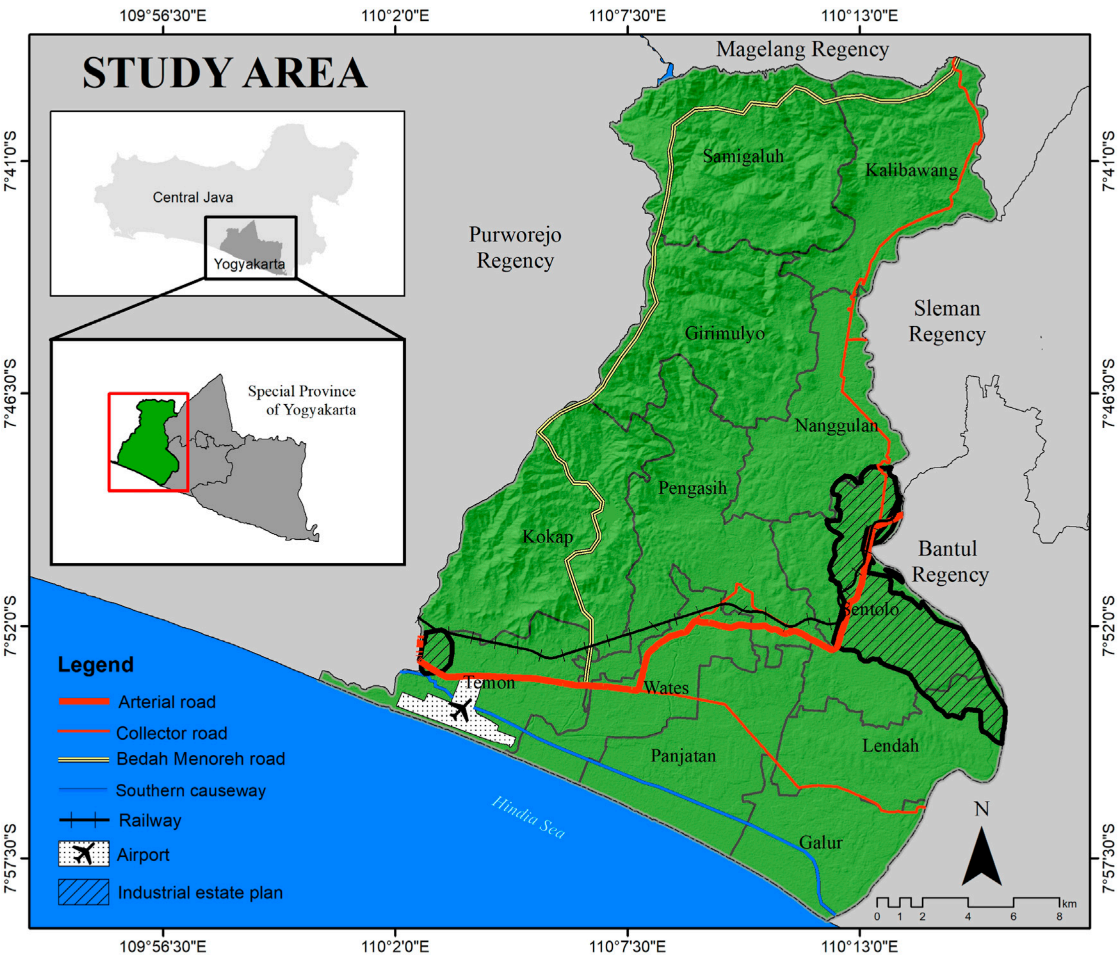

FIGURE 1 | Study area.

7645–1:2014. The ground check verification of 96 spots was held on December 3–10, 2020. The Kappa accuracy of 2010, 2015, and 2020, which validates LULC with the field data, was 81.2%, 81.5%, and 81.9%, respectively.

Several land-use prediction models have been used to model forecasting LULC, such as Cellular Automata (CA) (Aljoufie, 2014; Aljoufie et al., 2016), Cellular Automata-Markov (CA-Markov) (Mitsova et al., 2011; Kurnianti et al., 2015; Gong et al., 2019), SLEUTH (Saxena and Jat, 2020), and Artificial Neural Network (Tayyebi et al., 2011; Morgado et al., 2014; Gong et al., 2014; Bhatti et al., 2015). The CA and ANN combination model is provided by the multi-layer perceptron neural network (MLPNN) found in Modules for Land Use Change Simulation (MOLUSCE) (https://plugins.qgis.org/plugins/molusce/) and has been used by Hossain et al. (2021), Hossain and Moniruzzaman (2021), and Kafy et al. (2021). The forecasting LULC in this research used MOLUSCE in QGIS software 2.18.23 version. This module provided transitional potential modeling using an artificial neural network algorithm that has a high accuracy in building transitional models compared to other processes (Hossain et al., 2021), cellular automata for prediction, and Kappa accuracy for validation.

The prediction used the year 2035 to accommodate the gap of year implementation of the plan, which will be reviewed as an input in the next spatial plan. The prediction was carried out using two models: the simulation model used to validate LULC in 2020 and the simulation model to project the LULC of 2035. The LULC of 2010 and 2015 was selected to generate the first model. The modeling used a 5×5 pixel neighborhood to set maximum iteration and pixels for modeling. The iteration of data learning was used 10,000 times to gain stability (Eastman, 2012). The validation process for modeling predictions was conducted by comparing the simulation model LULC of 2020 and the actual land use using the validation panel. The model is said to have good accuracy when it has a Kappa value of at least 0.80 (Eastman, 2012). Furthermore, after validation, it can be used to build projected land use of 2035 with four iterations. One iteration represents 5 different time points (years) considering the trend of LULC change.

2.3.2 Land Allocation for Settlement Development

The allocation area for settlement development is constructed by considering the component of land capability as a factor of carrying capacity and suitability, which indicates the suitable strategic location of the settlement. Land capability and suitability are useful tools for building management units (De Feudis et al., 2021). Furthermore, paddy fields and forest areas are counted as limiting factors for settlement development. The island of Java, including Kulon Progo, is a national food barn. However, Kulon Progo has not delineated an official food security area and has been faced with massive urbanization, interrupting agricultural land sustainability. On the other hand, settlements are prohibited in the protected area under the Ministry of Environment and Forestry authorities.

Land capability is assessed by matching criteria of land characteristics including topography, drainage, soil texture, erosion, adequate soil depth, and particular factors such as rocks and flooding (Hardjowigeno and Widiatmaka, 2007). It is derived from USDA that was quantified by Arsyad (2012). The land capability class is divided into eight based on limiting factors and conditions in every class. The high to medium classes consist of classes I to IV, which can support cultivation activity. Classes V to VIII belong to the moderate to poor classes, which are prioritized for conservation due to severe inhibiting factors (Arsyad, 2012; Widiatmaka et al., 2015).

Land suitability analysis was conducted using GIS to carry out multi-criteria decision analysis (MCDA) and analytical hierarchy process (AHP) to weight the criteria (Widiatmaka et al., 2016b; Saxena and Jat, 2020). The criteria used include biophysical parameters such as slope, elevation, distance to rivers, distance to the coast, rainfall, and socioeconomic parameters. This consists of distance to road access, city center, trade facilities, transportation facilities, and health and education facilities. Weighting assessment involves experts to achieve an objective approach indicated by a consistency ratio (CR) of less than 10% (Saaty, 2008). Furthermore, the criteria that the expert has scored are measured to determine their influence on other criteria. This is conducted to obtain a relative ranking between criteria through pairwise comparison (Saaty, 2008). The scale of the relative ranking between variables ranged from 1 to 9. The overlay analysis using ArcGIS was calculated following the weight for every criterion to obtain suitability scores. The suitability of settlement is divided into four classes, namely, S1 (very suitable), S2 (fairly suitable), S3 (marginally suitable), and N (not suitable) using natural break.

Land allocation for settlement development is categorized into recommended and non-recommended allocations. Recommended allocation is the area that satisfies the following prerequisites: has high to medium class (I–IV) of land capability, is located in a suitable class (S1, S2, and S3), and is located in non-agricultural land (wetland agriculture) and forest area. Recommended allocation consists of Allocation I, II, and III, determined by their suitability class. Allocation I is the area that has S1 suitability class, and so on. Meanwhile, the area that does not meet the prerequisites is included in the non-recommended allocation.

2.3.3 Policy Allocation for Settlement Development

Policy allocation is decided by a simple decision matrix, considering the land allocation of settlement development, existing LULC of 2020, LULC prediction of 2035, and OLUP of 2012–2032. Furthermore, the policies are prepared using a simple decision matrix rule to be easily applied by stakeholders (Katsikopoulos et al., 2018; Pérez et al., 2020). Land allocation of settlement development represents the ideal area for developing settlements. This allocation has functioned as a performance unit analysis that will evaluate existing, predicted land use, and OLUP. Land uses are divided into three major classes: settlement (S), cultivated (C), and protected area (P), depending on the degree of environmental influence. The evaluation of predicted land use focused on settlement. The matrix integrated with conformance analysis between existing, predicted land use, and OLUP, which represents the regulation aspect. The review of OLUP was dependent on the conformance and performance analysis.

The policy allocation for settlement development is divided into two rules: the rule in the recommended area indicated by **Supplementary Figure S1** in which the main policy is to trigger settlement development, and the rule in the non-recommended area indicated by **Supplementary Figure S2** involving settlement growth prevention because of the shortage in allocation requirement. The rule follows the combination condition of existing (S/C/P), predicted (S), and OLUP (S/C/P). The combination condition link to the instrument of planning control policy is provided in Local Government of Kulon Progo Regency no. 1 of 2012, including zoning regulation, disincentive and incentive, land permit, and law enforcement. The alternative policy is to review the OLUP due to the irrelevant allocation or the potential threat to the environment.

3 RESULTS AND DISCUSSION

3.1 Existing and Land-Use Prediction

The Kappa accuracy of the 2020 simulation map that validates the simulation map 2020 with LULC of 2020 is 0.95. The prediction results in 2035 showed that the settlement area would increase from 4,129.3 ha to 5,863.5 ha, indicated by the bar of settlement in **Figure 2A**. The newly built area indicated by settlement in this research tends to grow near the existing one in accordance with the study of Luo and Wei (2009) and Zhang et al. (2011). It caused the neighborhood land use to have a more significant conversion probability. The mixed garden has a spatial distribution correlation with the settlement that causes the more extensive conversion of the mixed garden. The conversion of wetland agriculture also increased due to the low land rent. Rondhi et al. (2018) reveal that agricultural land rent is lower than housing rent, especially in the peri-urban area that encourages agricultural land conversion. The southern part of Kulon Progo is experiencing peri-urban conditions with the perspective change. The number of conversions tends to increase in the second period shown in **Figure 2B**, indicating the process of settlement expansion. However, the massive urbanization in

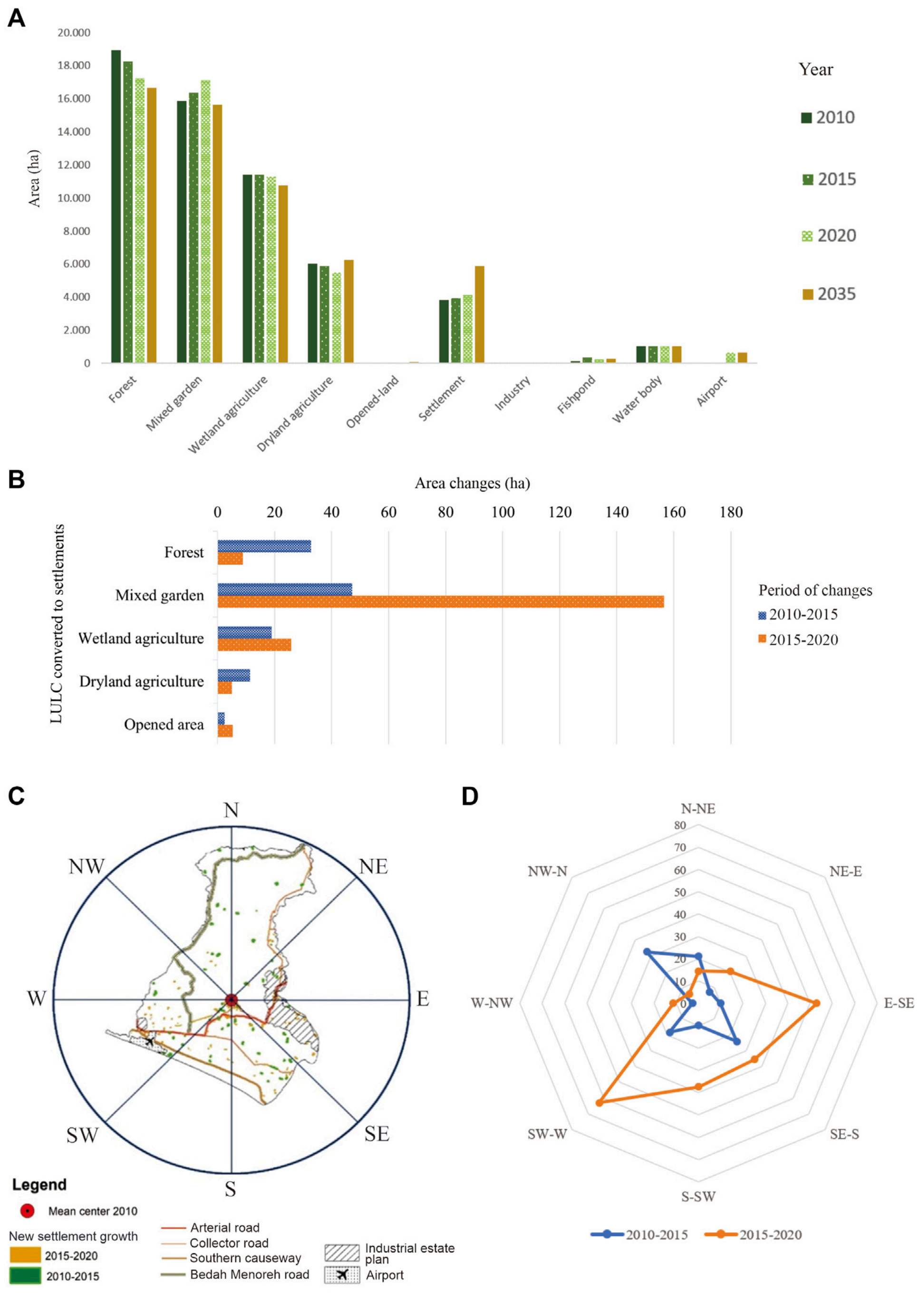

FIGURE 2 | The LULC changes and prediction.

the early phase of aerotropolis from the perspective of settlement growth rate remains unnoticed. The growth rate in the first period was only 113 ha and was increased to 168.6 ha in the second one. Compared with the immediate mature stage urban area, the Yogyakarta Urbanized Area (YUA) settlement growth was 329 ha per year in 2003–2013 (Wijaya and Umam, 2016).

Even though the growth rate of settlement is still small, the process of regional perspective change can be identified by the deviation direction of new settlement location growth. The new settlements in 2015–2020 are primarily distributed between W-SW (West-Southwest) or the surrounding airport area. The E-SE (East-Southeast), S-SW (South-Southwest), and SW-W (Southwest-West) areas also experienced an increase, although

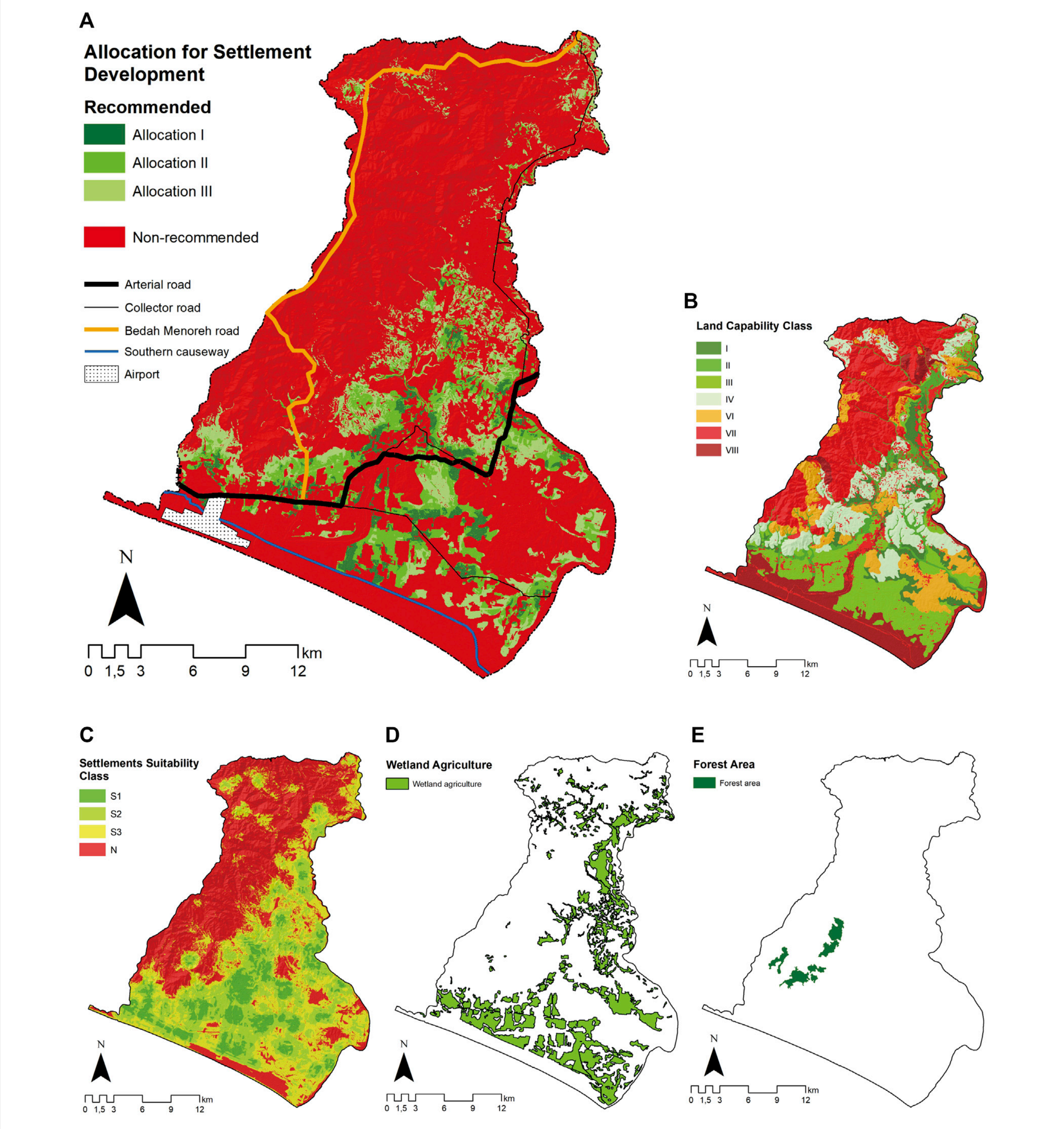

FIGURE 3 | Land allocation for settlement development.

not as much as the W-SW. Generally, the area on the southern part of settlement's mean center [indicated by the red dot in **Figure 2C**, which was adopted from Litasari et al. (2022)] experienced an increase during the aerotropolis period shown by the orange net in **Figure 2D** and spatially can be seen in **Figure 2C**. The result is in accordance with the study of Litasari et al. (2022), which applied perspective change using spatial mean center in three time points of settlement development. Pratiwi and Rahardjo (2018) also found increasing land price around the airport area, especially the area with good accessibility near the Daendeles and National road. It indicated that the emergence of the airport as a new growth center potentially triggered the activity shown by the deviation in the settlement's growth location. The study conducted by Wang et al. (2019) shows that the emergence of transportation infrastructure (in the case of the high-speed railway) could deviate the spatial structure.

3.2 Land Allocation for Settlement Development

Kulon Progo is dominated by non-recommended allocation covering 79% of the total area, which is distributed mainly in Menoreh Hills. According to the study of Hadmoko et al. (2010), the area of Menoreh covering 26,100 ha (45.5% total area of Kulon Progo) is prone to landslide (moderate to high risk) and, thus, is not suitable for settlement. The Bedah Menoreh project shown with a yellow line in **Figure 3A** will provide access, potentially triggering settlement growth. This area is grouped in Class VII and VIII with the slope as the main obstacle and an area with N suitability class shown in **Figures 3B,C**.

The remaining percentage is contributed by other obstacles like coarse soil texture in the southern part of Kulon Progo. Soil texture affects the ability of the soil to hold and pass water (Arsyad, 2012). Water can easily pass (and thus difficult to store) in soil with a coarse texture (Lu et al., 2021). In addition, soil with a coarse texture is a poor filtering material and can cause groundwater pollution (Hardjowigeno and Widiatmaka, 2007). Such conditions raise the potential for sanitation problems and water pollution. It is a big challenge due to the development of aerotropolis that stimulates the activities in the southern part of Kulon Progo. The wetland agriculture (**Figure 3D**) and forest area (**Figure 3E**) promote to be protected and included in non-recommended allocation. Meanwhile, the areas of recommended allocation I, II, and III are 1,856.11 (3.23%), 4,796.01 (8.37%), and 5,386.36 (9.40%) ha, respectively. The recommended area is sufficient to meet the demands of the settlement in 2035, which covers 5,863.5 ha according to the LULC prediction.

3.3 Policy Allocation for Settlement Development

The evaluation indicates that 2,656.53 ha (64.33%) of existing settlement belong to non-recommended allocation and are predicted to increase by 1,145.83 ha in 2035. Widodo et al. (2015) also pointed out that only 22.73% of villages have good land resource carrying capacity in YUA. The number of areas with good carrying capacity in YUA smaller than Kulon Progo is probably affected by the degree of urban growth. Consequently, the development of settlements needs to be conducted carefully according to carrying capacity. The development in Kulon Progo is in the early stages; the supply land is adequate and can still meet the demand. The main problem of settlement development lies in spatial distribution.

It is difficult to demolish or relocate settlements located in non-recommended allocation. The implementation of the policies requires data such as land ownership, cost–benefit analysis, and multidimensional analysis to measure social and economic effects. Demolition may be the alternative when there is a public interest regulated in Law Number 2 of 2012, which provides appropriate and fair compensation. The law can be enforced in the area defined as a protected area in the OLUP following the S-S-P rule in **Figure 4A**. The rule shows a good performance of the OLUP that conforms to land allocation. However, the existing land use does not conform to land allocation and OLUP. The condition represents land-use infringement. The area included 458.83-ha settlements distributed in Menoreh Hills shown in red color in **Figure 4B**.

The growth of the rest of the existing settlement outside the protected area is supposed to be limited using the disincentive mechanism. The rule follows the condition of S-S-S/C in **Figure 4A**. The condition shows conformity between the existing, predicted, and OLUP, but it has a bad performance due to the location that distribute in non-recommended allocation. It is strengthened by the study of Shen et al. (2019), which stated that good conformity does not always represent good performance. The disincentives, in this case, can be given in the form of the imposition of high tax levies, adjusted to the number of costs required to overcome the land-use impacts, restrictions on infrastructure provision, imposition of compensation, and penalties.

The disincentive mechanism can be implemented to the cultivated area potentially converted into settlements. Even though the OLUP and existing land use have conformity and good performance, it potentially can be converted into settlement in the future. The rule follows the condition of C-S-C/P. A study of Widiatmaka et al. (2016) concludes that the condition is fairly without problem because the study obviates the dynamic change of future conditions. The settlement does not exist but potentially will grow in non-recommended allocation, so the policy is required to prevent conversion. Dadashpoor and Malekzadeh (2020) found that disincentives affect the spatial structure by preventing the growth of metropolitan areas through social welfare, rule, and regulation. The mechanism that can be used is to reject the proposed permit of settlement utilization and restrict the infrastructure provision. The area recommended proposed a disincentive policy covering the entire settlement in Kulon Progo indicated by dark purple in **Figure 4B** with a cover area of 2,730.5 ha.

The lack of a spatial plan cannot be tackled due to the changing condition that needs to be evaluated and revised (Faludi, 2000). The policy to revise the zoning allocation is required when the existing condition has a good performance, whereas the OLUP has a poor performance. It is indicated by the condition of C-S-S or P-S-S/C. Indeed, it has a high probability that it will be converted into settlement in the future. The settlement allocation adjusted in that area will burden the environment due to the exceeded land capacity. Meanwhile, the condition of P-S-P has good conformance and performance for both the OLUP and the existing condition, although it has potentially converted into settlement. Thus, it required incentive policies to preserve the allocation of land use. The incentive is an effective tool to protect the landscape and is better applied in privately owned lands (Wainaina et al., 2021).

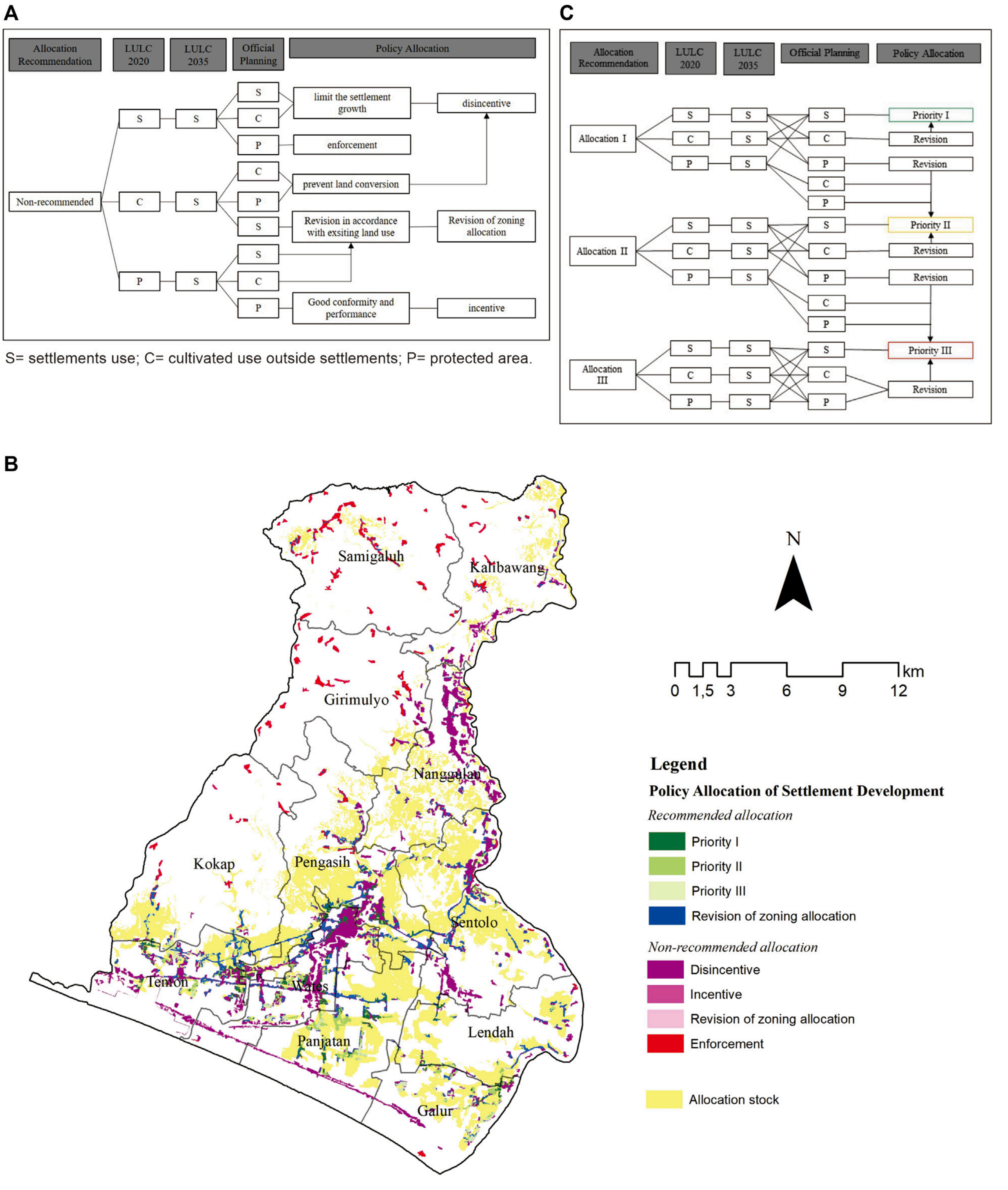

FIGURE 4 | Policy allocation for settlement development.

Using this allocation matrix rule, approximately 168.5 ha of protected area and 400.8 ha of wetland agriculture can be preserved. The settlement that potentially grows in the non-recommended allocation can be triggered to grow in the stock area. The amount of allocation stock until 2035 covers 9.968,08 ha, indicated by the yellow polygon in **Figure 4B**.

Conversely, the existing and projected settlements located in the recommended allocation are supposed to be stimulated to grow. The zoning allocation is targeted to an area of 1,053.47 to trigger the settlement growth. It only needs details on the priority hierarchy to follow the rule in **Figure 4C**. The hierarchy of prioritization utilization in recommended allocation is divided into the following:

- Priority I: located in Allocation I and prioritized for the designation of settlements with high density.
- Priority II: distributed in Allocation II and allocated for settlements with medium to low density because of some obstacles that need some management.
- Priority III: areas that can be used for settlement expansion that will be used when the area in Allocations I and II has been exceeded. This area is located in Allocation 3.

The existing settlements that have been regulated in zoning allocation cover an area of 751.24 ha distributed in Allocation I, II, and III. However, 721.53 ha is not adjusted to settlements in OLUP. On the other hand, the total predicted settlement located in recommended allocation is 595.34 ha, most of which is distributed in Allocation II. However, only 292.23 ha have been regulated in zoning allocation. Although this area has a good performance in supporting activity, the policy to allocate using the stipulation settlement zoning allocation is required. A zoning allocation is a tool that can manage the distribution of settlements to prevent sprawling (Chadchan and Shankar, 2012).

4 CONCLUSION

The evaluation reveals that the present settlements are dominantly distributed in non-recommended allocation and are predicted to increase in 2035. The stock of land allocation for settlement development is adequate to fill the settlement demand in 2035, which is represented by the area of predicted settlements. The problem lies in the distribution pattern rather than the lack of appropriate area for settlement development. The policy allocation matrix rule fills the gap linking policy recommendation to the planning control instruments by managing the distribution of settlement growth. The rule integrates the performance and conformance between the land allocation for settlement development accommodating the ideality aspect, existing and projected settlement that represents the reality aspect, and OLUP as an aspect of regulation. Using this method, the potential conversion of the area with low carrying capacity, wetland agriculture, and forest area can be mitigated and triggered to grow in recommended allocation. The research confined the evaluation to build a policy allocation only on the settlement. A broader analysis and framework can be developed to accommodate all land-use evaluation in OLUP.

DATA AVAILABILITY STATEMENT

The original contributions presented in the study are included in the article/**Supplementary Material**, further inquiries can be directed to the corresponding author.

AUTHOR CONTRIBUTIONS

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SUPPLEMENTARY MATERIAL

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REFERENCES

AbuSada, J., and Thawaba, S. (2011). Multi Criteria Analysis for Locating Sustainable Suburban Centers: A Case Study from Ramallah Governorate, Palestine. *Cities* 28 (5), 381–393. doi:10.1016/j.cities.2011.05.001

Aguirre, J., Mateu, P., and Pantoja, C. (2019). Granting Airport Concessions for Regional Development: Evidence from Peru. *Transport Policy* 74, 138–152. doi:10.1016/j.tranpol.2018.12.003

Aljoufie, M., Brussel, M., Zuidgeest, M., van Delden, H., and van Maarseveen, M. (2016). Integrated Analysis of Land-Use and Transport Policy Interventions. *Transport. Plann. Techn.* 39 (4), 329–357. doi:10.1080/03081060.2016.1160578

Aljoufie, M. (2014). Toward Integrated Land Use and Transport Planning in Fast-Growing Cities: The Case of Jeddah, Saudi Arabia. *Habitat Int.* 41, 205–215. doi:10.1016/j.habitatint.2013.08.010

Arsyad, S. (2012). *Konservasi Tanah Dan Air*. Editor H. Siregar. 2th ed (Bogor: IPB Press).

Baum-Snow, N. (2010). Changes in Transportation Infrastructure and Commuting Patterns in US Metropolitan Areas, 1960-2000. *Am. Econ. Rev.* 100 (2), 378–382. doi:10.1257/aer.100.2.378

Bhatti, S. S., Tripathi, N. K., Nitivattananon, V., Rana, I. A., and Mozumder, C. (2015). A Multi-Scale Modeling Approach for Simulating Urbanization in a

Metropolitan Region. *Habitat Int.* 50, 354–365. doi:10.1016/j.habitatint.2015.09.005

Blonigen, B. A., and Cristea, A. D. (2015). Air Service and Urban Growth: Evidence from a Quasi-Natural Policy Experiment. *J. Urban Econ.* 86, 128–146. doi:10.1016/j.jue.2015.02.001

Bovet, J., Reese, M., and Köck, W. (2018). Taming Expansive Land Use Dynamics - Sustainable Land Use Regulation and Urban Sprawl in a Comparative Perspective. *Land Use Policy* 77, 837–845. doi:10.1016/j.landusepol.2017.03.024

Chadchan, J., and Shankar, R. (2012). An Analysis of Urban Growth Trends in the post-economic Reforms Period in India. *Int. J. Sustain. Built Environ.* 1, 36–49. doi:10.1016/j.ijsbe.2012.05.001

Chi, G. (2012). The Impacts of Transport Accessibility on Population Change across Rural, Suburban and Urban Areas: A Case Study of Wisconsin at Sub-county Levels. *Urban Stud.* 49 (12), 2711–2731. doi:10.1177/0042098011431284

Dadashpoor, H., and Malekzadeh, N. (2020). Driving Factors of Formation, Development, and Change of Spatial Structure in Metropolitan Areas: A Systematic Review. *J. Urban Manage.* 9, 286–297. doi:10.1016/j.jum.2020.06.005

De Feudis, M., Falsone, G., Gherardi, M., Speranza, M., Vianello, G., and Vittori Antisari, L. (2021). GIS-based Soil Maps as Tools to Evaluate Land Capability and Suitability in a Coastal Reclaimed Area (Ravenna, Northern Italy). *Int. Soil Water Conserv. Res.* 9 (2), 167–179. doi:10.1016/j.iswcr.2020.11.007

Eastman, J. R. (2012). *IDRISI Selva Tutorial*. Worcester: Clark Labs-Clark University.

Faludi, A. (2000). The Performance of Spatial Planning. *Plann. Pract. Res.* 15 (4), 299–318. doi:10.1080/713691907

Fernandes, V. A., Pacheco, R. R., Fernandes, E., and da Silva, W. R. (2019). Regional Change in the Hierarchy of Brazilian Airports 2007-2016. *J. Transport Geogr.* 79, 102467. doi:10.1016/j.jtrangeo.2019.102467

Giyarsih, S. R. (2010). Urban Sprawl of the City of Yogyakarta, Special Preference to the Stage of Spatial Transformation. *Indones. J. Geogr.* 42 (1), 47–58. doi:10.22146/ijg.1576

Gong, J., Chen, W., Liu, Y., and Wang, J. (2014). The Intensity Change of Urban Development Land: Implications for the City Master Plan of Guangzhou, China. *Land Use Policy* 40, 91–100. doi:10.1016/j.landusepol.2013.05.001

Gong, X., Bian, J., Wang, Y., Jia, Z., and Wan, H. (2019). Evaluating and Predicting the Effects of Land Use Changes on Water Quality Using SWAT and CA-Markov Models. *Water Resour. Manage.* 33 (14), 4923–4938. doi:10.1007/s11269-019-02427-0

Hadmoko, D. S., Lavigne, F., Sartohadi, J., Hadi, P., and Winaryo, W. (2010). Landslide Hazard and Risk Assessment and Their Application in Risk Management and Landuse Planning in Eastern Flank of Menoreh Mountains, Yogyakarta Province, Indonesia. *Nat. Hazards* 54, 623–642. doi:10.1007/s11069-009-9490-0

Han, J. (2020). Can Urban Sprawl Be the Cause of Environmental Deterioration? Based on the Provincial Panel Data in China. *Environ. Res.* 189, 109954. doi:10.1016/j.envres.2020.109954

Hardjowigeno, S., and Widiatmaka, W. (2007). *Evaluasi Kesesuaian Lahan Dan Perencanaan Tataguna Lahan*. Yogyakarta: Gadjah Mada University Press.

He, Y., and Xie, H. (2019). Exploring the Spatiotemporal Changes of Ecological Carrying Capacity for Regional Sustainable Development Based on GIS: A Case Study of Nanchang City. *Technol. Forecast. Soc. Change* 148, 119720. doi:10.1016/j.techfore.2019.119720

He, Q., Song, Y., Liu, Y., and Yin, C. (2017). Diffusion or Coalescence? Urban Growth Pattern and Change in 363 Chinese Cities from 1995 to 2015. *Sustain. Cities Soc.* 35, 729–739. doi:10.1016/j.scs.2017.08.033

Hossain, F., and Moniruzzaman, D. M. (2021). Environmental Change Detection through Remote Sensing Technique: A Study of Rohingya Refugee Camp Area (Ukhia and Teknaf Sub-district), Cox's Bazar, Bangladesh. *Environ. Challenges* 2, 100024. doi:10.1016/j.envc.2021.100024

Hossain, M. T., Zarin, T., Sahriar, M. R., and Haque, M. N. (2021). Machine Learning Based Modeling for Future Prospects of Land Use Land Cover Change in Gopalganj District, Bangladesh. *Phys. Chem. Earth*, 103022. doi:10.1016/j.pce.2021.103022

Hubbard, J. R., Mitra, S., and Miller, C. R. (2019). Analyzing Human Capital as a Component of the Aerotropolis Model. *J. Air Transport Manage.* 78, 63–70. doi:10.1016/j.jairtraman.2019.04.008

Kafy, A. A., Dey, N. N., Al Rakib, A., Rahaman, Z. A., Nasher, N. M. R., and Bhatt, A. (2021). Modeling the Relationship between Land Use/land Cover and Land Surface Temperature in Dhaka, Bangladesh Using CA-ANN Algorithm. *Environ. Challenges* 4, 100190. doi:10.1016/j.envc.2021.100190

Kasarda, J. D. (2019). "Aerotropolis," in *The Wiley Blackwell Encyclopedia of Urban and Regional Studies*, 1–7. New York: John Wiley & Sons Inc. doi:10.1002/9781118568446.eurs0436

Kassis, G., Bertrand, N., and Pecqueur, B. (2021). Rethinking the Place of Agricultural Land Preservation for the Development of Food Systems in Planning of Peri-Urban Areas: Insights from Two French Municipalities. *J. Rural Stud.* 86, 366–375. doi:10.1016/j.jrurstud.2021.07.003

Katsikopoulos, K. V., Durbach, I. N., and Stewart, T. J. (2018). When Should We Use Simple Decision Models? A Synthesis of Various Research Strands. *Omega* 81, 17–25. doi:10.1016/j.omega.2017.09.005

Kurnianti, D. N., Rustiadi, E., and Baskoro, D. P. T. (2015). Land Use Projection for Spatial Plan Consistency in Jabodetabek. *Indones. J. Geogr.* 47, 124–131. doi:10.22146/ijg.9249

Lillesand, T. M., Kiefer, R. W., and Chipman, J. W. (2015). *Remote Sensing and Image Interpretation*. United States: Petra Recter.

Litasari, U. C. N., Widiatmaka, W., Munibah, K., and Machfud, M. (2022). Spatial Pattern Changing Analysis of Built-Up Due to the New Era of Aerotropolis in Kulon Progo, D. I. Yogyakarta. *IOP Conf. Ser. Earth Environ. Sci.* 950, 012100. doi:10.1088/1755-1315/950/1/012100

Lu, J., Shao, G., Gao, Y., Zhang, K., Wei, Q., and Cheng, J. (2021). Effects of Water Deficit Combined with Soil Texture, Soil Bulk Density and Tomato Variety on Tomato Fruit Quality: A Meta-Analysis. *Agric. Water Manage.* 243, 106427. doi:10.1016/j.agwat.2020.106427

Luo, J., and Wei, Y. H. D. (2009). Modeling Spatial Variations of Urban Growth Patterns in Chinese Cities: The Case of Nanjing. *Landscape Urban Plann.* 91, 51–64. doi:10.1016/j.landurbplan.2008.11.010

Lyu, Y., Wang, M., Zou, Y., and Wu, C. (2022). Mapping Trade-Offs Among Urban Fringe Land Use Functions to Accurately Support Spatial Planning. *Sci. Total Environ.* 802, 149915. doi:10.1016/j.scitotenv.2021.149915

Torres Mallma, S. F. (2021). Mainstreaming Land Use Planning into Disaster Risk Management: Trends in Lima, Peru. *Int. J. Disaster Risk Reduct.* 62, 102404. doi:10.1016/j.ijdrr.2021.102404

McGraw, M. J. (2020). The Role of Airports in City Employment Growth, 1950–2010. *J. Urban Econ.* 116, 103240. doi:10.1016/j.jue.2020.103240

Minister of Transportation (2013). Decree of the Minister of Transportation Number: KP.1164 of 2013 Concerning Location Determination of New Airport in Kulon Progo Regency, Yogyakarta Province. Jakarta: Minister of Transportation.

Mitsova, D., Shuster, W., and Wang, X. (2011). A Cellular Automata Model of Land Cover Change to Integrate Urban Growth with Open Space Conservation. *Landscape Urban Plann.* 99 (2), 141–153. doi:10.1016/j.landurbplan.2010.10.001

Morgado, P., Gomes, E., and Costa, N. (2014). Competing Visions? Simulating Alternative Coastal Futures Using a GIS-ANN Web Application. *Ocean Coastal Manage.* 101, 79–88. doi:10.1016/j.ocecoaman.2014.09.022

Nijhum, F., Westbrook, C., Noble, B., Belcher, K., and Lloyd-Smith, P. (2021). Evaluation of Alternative Land-Use Scenarios Using an Ecosystem Services-Based Strategic Environmental Assessment Approach. *Land Use Policy* 108, 105540. doi:10.1016/j.landusepol.2021.105540

Omar, N. Q., and Raheem, A. M. (2016). Determining the Suitability Trends for Settlement Based on Multi Criteria in Kirkuk, Iraq. *Open Geospatial Data Soft. Stand.* 1 (1), 1–9. doi:10.1186/s40965-016-0011-2

Pérez, W., Arias, R., Taipe, A., Ortiz, O., Forbes, G. A., Andrade-Piedra, J., et al. (2020). A Simple, Hand-Held Decision Support Designed Tool to Help Resource-Poor Farmers Improve Potato Late Blight Management. *Crop Prot.* 134, 105186. doi:10.1016/j.cropro.2020.105186

Pratiwi, S. E., and Rahardjo, N. (2018). Pemodelan Spasial Harga Lahan Dan Perubahannya Akibat Pembangunan Bandara New Yogyakarta International Airport Di Sekitar Area Bandara. *J. Bumi Indones.* 7 (3), 1–12.

Rondhi, M., Pratiwi, P. A., Handini, V. T., Sunartomo, A. F., and Budiman, S. A. (2018). Agricultural Land Conversion, Land Economic Value, and Sustainable Agriculture: A Case Study in East Java, Indonesia. *Land* 7 (4), 1–19. doi:10.3390/land7040148

Rustiadi, E., Pravitasari, A. E., Setiawan, Y., Mulya, S. P., Pribadi, D. O., and Tsutsumida, N. (2020). Impact of Continuous Jakarta Megacity Urban Expansion on the Formation of the Jakarta-Bandung Conurbation over the Rice Farm Regions. *Cities* 111, 103000. doi:10.1016/j.cities.2020.103000

Saaty, T. L. (2008). Decision Making with the Analytic Hierarchy Process. *IJSSCI* 1 (1), 83–98. doi:10.1504/ijssci.2008.017590

Saxena, A., and Jat, M. K. (2020). Land Suitability and Urban Growth Modeling: Development of SLEUTH-Suitability. *Comput. Environ. Urban Syst.* 81, 101475. doi:10.1016/j.compenvurbsys.2020.101475

Shen, X., Wang, X., Zhang, Z., Lu, Z., and Lv, T. (2019). Evaluating the Effectiveness of Land Use Plans in Containing Urban Expansion: An Integrated View. *Land Use Policy* 80, 205–213. doi:10.1016/j.landusepol.2018.10.001

Shi, X., Zhou, F., and Wang, Z. (2021). Research on Optimization of Ecological Service Function and Planning Control of Land Resources Planning Based on Ecological Protection and Restoration. *Environ. Techn. Innovation* 24, 101904. doi:10.1016/j.eti.2021.101904

Solon, J. (2009). Spatial Context of Urbanization: Landscape Pattern and Changes between 1950 and 1990 in the Warsaw Metropolitan Area, Poland. *Landscape Urban Plann.* 93, 250–261. doi:10.1016/j.landurbplan.2009.07.012

Statistic Bureau of Kulon Progo (2021). Kulon Progo Regency in Figures 2020. Available at: https://kulonprogokab.bps.go.id/publication.html (Accessed June 20, 2021).

Tayyebi, A., Pijanowski, B. C., and Pekin, B. (2011). Two Rule-Based Urban Growth Boundary Models Applied to the Tehran Metropolitan Area, Iran. *Appl. Geogr.* 31 (3), 908–918. doi:10.1016/j.apgeog.2011.01.018

Tveter, E. (2017). The Effect of Airports on Regional Development: Evidence from the Construction of Regional Airports in Norway. *Res. Transport. Econ.* 63, 50–58. doi:10.1016/j.retrec.2017.07.001

Ustaoglu, E., and Aydınoglu, A. C. (2020). Suitability Evaluation of Urban Construction Land in Pendik District of Istanbul, Turkey. *Land Use Policy* 99, 104783. doi:10.1016/j.landusepol.2020.104783

Wainaina, P., Minang, P. A., Nzyoka, J., Duguma, L., Temu, E., and Manda, L. (2021). Incentives for Landscape Restoration: Lessons from Shinyanga, Tanzania. *J. Environ. Manage.* 280, 111831. doi:10.1016/j.jenvman.2020.111831

Wang, J., and Maduako, I. N. (2018). Spatio-temporal Urban Growth Dynamics of Lagos Metropolitan Region of Nigeria Based on Hybrid Methods for LULC Modeling and Prediction. *Eur. J. Remote Sensing* 51 (1), 251–265. doi:10.1080/22797254.2017.1419831

Wang, S., Wang, J., and Liu, X. (2019). How Do Urban Spatial Structures Evolution in the High-Speed Rail Era? Case Study of Yangtze River Delta, China. *Habitat Int.* 93, 102051. doi:10.1016/j.habitatint.2019.102051

WidiatmakaAmbarwulan, W., Setiawan, Y., Purwanto, M. Y. J., Taryono, T., and Effendi, H. (2015). Land Use Planning for Brackish Water Shrimp Ponds in the North Coast of Tuban, Indonesia. *Indones. J. Geogr.* 47 (2), 194–211. doi:10.22146/ijg.9268

Widiatmaka, W., Ambarwulan, W., Ambarwulan, Y., and Walter, C. (2016a). Assessing the Suitability and Availability of Land for Agriculture in Tuban Regency, East Java, Indonesia. *Appl. Environ. Soil Sci.* 2016, 1–13. doi:10.1155/2016/7302148

Widiatmaka, W., Ambarwulan, W., and Sudarsono, S. (2016b). Spatial Multi-Criteria Decision Making for Delineating Agricultural Land in Jakarta Metropolitan Area's Hinterland: Case Study of Bogor Regency, West Java. *Agrivita* 38 (2), 105–115. doi:10.17503/agrivita.v38i2.746

Widodo, B., Lupyanto, R., Sulistiyono, B., Harjito, D. A., Hamidin, J., Hapsari, E., et al. (2015). Analysis of Environmental of Carrying Capacity for the Development of Sustainable Settlement in Yogyakarta Urban Area. *Proc. Environ. Sci.* 28, 519–527. doi:10.1016/j.proenv.2015.07.062

Wijaya, M. S., and Umam, N. (2016). Permodelan Spasial Perkembangan Fisik Perkotaan Yogyakarta Menggunakan Model Cellular Automata Dan Regresi Logistik Biner. *Maj. Ilm. Globe* 17, 165–172. doi:10.24895/MIG.2015.17-2.227

Wilson, B., and Chakraborty, A. (2013). The Environmental Impacts of Sprawl: Emergent Themes from the Past Decade of Planning Research. *Sustainability* 5 (8), 3302–3327. doi:10.3390/su5083302

Zhang, Q., Ban, Y., Liu, J., and Hu, Y. (2011). Simulation and Analysis of Urban Growth Scenarios for the Greater Shanghai Area, China. *Comput. Environ. Urban Syst.* 35 (2), 126–139. doi:10.1016/j.compenvurbsys.2010.12.002

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OPINION
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Making Sustainable Forest Development Work: Formulating an Idea for a More Appropriate Green Policy Paradigm

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INTRODUCTION

There is increasing pressure on the conversion of forest areas for food, energy, development, and other purposes. Development that is driven by green growth is being proposed as a solution. The problem, however, is that many ideas driving the notion of "green" growth are trapped in a contradiction that overwhelmingly focuses on the "green" while neglecting the growth, even if there is a compelling case to be made that "green" and "growth" are not two competing paradigms (World Bank, 2012).

Freedom should be the primary element of development. It means that the only acceptable evaluation of human progress is primarily and ultimately enhancement of freedom, and the achievement of development depends on people's free agency (Sen, 1999). Peacock (2021) argues that the freedom proposed by Sen (1999) is a combination of rationality and commitment. Commitment is essential in development strategy, in addition to the mainstream rational choice theory (Peacock, 2021).

Sustainable development is a global issue and forest regime is currently dominates global sustainability discourses on the management of the world's forests (Giessen, 2013; Sahide et al., 2015). However, such discourses also pay less attention to the different situations in developed and developing countries. For instance, reducing emission and net-zero deforestation has been promoted as policy ideals to be implemented according to the Paris Agreement (UNFCCC, 2015) is less considering the countries' social, economic, and ecological differences. Thus, promoting more objective sustainable development principles, i.e., reversibility or limits (Letey, 1973), adaptability or tolerance (Letey, 1973), rational commitment (Kuznets, 1955; Sen, 1999; Peacock, 2021), and legitimacy (Nurrochmat et al., 2016b; Peacock, 2021), is necessary to build more appropriate global perspectives in sustainable development.

The article aims to promote an idea for establishing a more appropriate set of principles of sustainable forests. The argument provides insight into sustainable forest development in ways that are not only determined by the temporary disturbance of forests (environmental degradation) but also the potential for economic growth, which considers the ability of forests to recover (see Silva, 2021) using the concepts of limits and tolerance (Letey, 1973), rational commitment (Sen, 1999; Peacock, 2021) and legitimacy (Nurrochmat et al., 2016b). Scientific arguments support the idea of the article, for instance, Environmental Kuznets Curve (EKC) theory (1955), limits and tolerance of ecology (Letey, 1973), and also the green development strategies of the World Bank (2012). The argument is laid out for the purposes of practical implementation.

ECOSYSTEM DEGRADATION AND RECOVERY PERIODS OF DEVELOPMENT

Economic growth and environmental sustainability are like two sides of a coin inseparable from the sustainable development concept. Studies of contemporary popular natural resource economics tend to apply the *Environmental Kuznets' Curve* (EKC) –a modification of the *inverted U hypothesis* (Kuznets, 1955) that explains the interrelation between economic growth as a function of per capita income and the level of environmental degradation (**Figure 1**).

It is commonly found in the development approaches in developing countries that rely on intense pressures on land and natural resources, where increasing per-capita income (PCI) is compensated by the depletion of natural resources and environmental degradation. The model then suggests that after achieving a certain level of human welfare, a turning point will occur whereby the higher levels of welfare will initiate broader interests, policies, and outcomes toward environmental sustainability. Environmental degradation thus decreases along with increasing PCI and broadly reflects conditions found in developed countries.

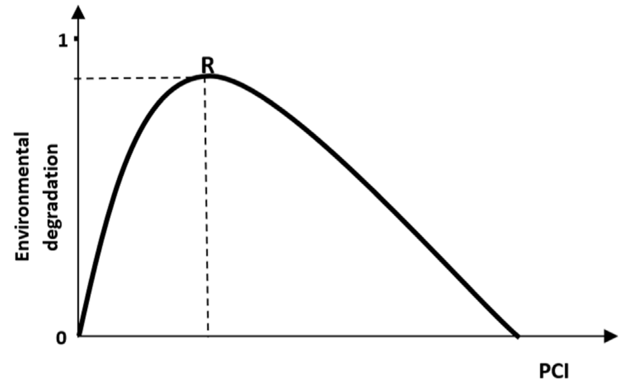

FIGURE 1 | Environmental Kuznets' Curve.

The Reversibility Principle

In the early stages of development, environmental degradation is part of the economic development strategy, as long as the degradation does not exceed the limits or ability of the environment to recover (Letey, 1973) or the natural regeneration's ability of degraded forests (Silva, 2021). Thus, limits on the exploitation of natural resources must be considered in the development strategy.

A universal illustration of the policy strategy for using natural resources and environmental recovery is presented as a metaphor of choosing a flyover road development strategy. The first policy option is to close the old road temporarily with the consequence that people who want to go through cannot pass. Meanwhile, the second option is to open the road halfway so that people can still pass, but results in more extended traffic jams. The completion of the flyover project can be considered a turning point in achieving smoother traffic patterns, which is similar in this case to support the turning points of environmental recovery. Theoretically, faster transitions to environmental recovery are possible through the first option, whereby total closure of roads to make flyovers are applied. However, this can only succeed with particular prerequisites, namely that there are alternative access points to important places such as hospitals, transportation hubs, and other vital facilities. Still, the benefits of the flyover are quickly completed so that smoother traffic patterns will recover with better quality (Nurrochmat et al., 2017).

In line with this illustration, economic development that causes environmental disturbances, for instance forest degradation or temporary deforestation, can still be considered an option of sustainable development strategies if the disturbance is laid within the limits of being reversible and encourages the acceleration of environmental recovery as welfare increases. In this context, it is clear that the concept of sustainability is bound to and cannot be separated from human welfare.

The Adaptability Principle

Every living thing has limits in facing environmental changes, which Letey, 1973 called "tolerance," which underpins the adaptability principles. Humans can adapt to different extreme environmental conditions that change over generations. Many other living things such as plants and animals also adjust to different situations, including toxic ones (Letey, 1973) and extreme environments of warming and cooling temperatures. Thus, the concept of sustainability should not (always) mean exactly bequeathing natural resources such as the current state of the generations to come. Likewise, the idea of restoration does not have to be interpreted as restoring conditions to an "original" ecosystem because: first, there is no definition of time considered as the original natural state, and second, human beings and other living things may have adapted to new environments and new human-environment relations, so that restoring environmental conditions can even have the potential of presenting new dangers.

The film *"Jurassic Park"* describes a powerful critique of people who seek to restore prehistoric beings on Earth. Successfully reviving dinosaurs, in this case, can create dangerous new ecosystems for other species and human life. Thus, the concept of sustainability should be seen in the context of the human and other living things' ability in the present situation to adapt to environmental changes. Sustainability is a dynamic concept whose conditions may differ according to the context of time and place. Therefore, restoring the environment does not have to become a "zero environmental degradation" principle. Environmental recovery needs to be carried out up to the limits of natural adaptability (**Figure 2**).

Many of these questions of environmental change are at the forefront of our understanding and policies around climate change. Climate change results in environmental change at time scales that challenge the notions of irreversibility and adaptability. Thus, many aspects of adaptation should be integrated with mitigation in forest and environmental policy, both at the national and local levels (Di Gregorio et al., 2016; Nurrochmat et al., 2020; Nurrochmat et al., 2021). In this context, restoration for climate mitigation needs to be done at the limit of humans to adapt and not necessarily exceed reasonable social costs. The calculus of this has changed with the increasing

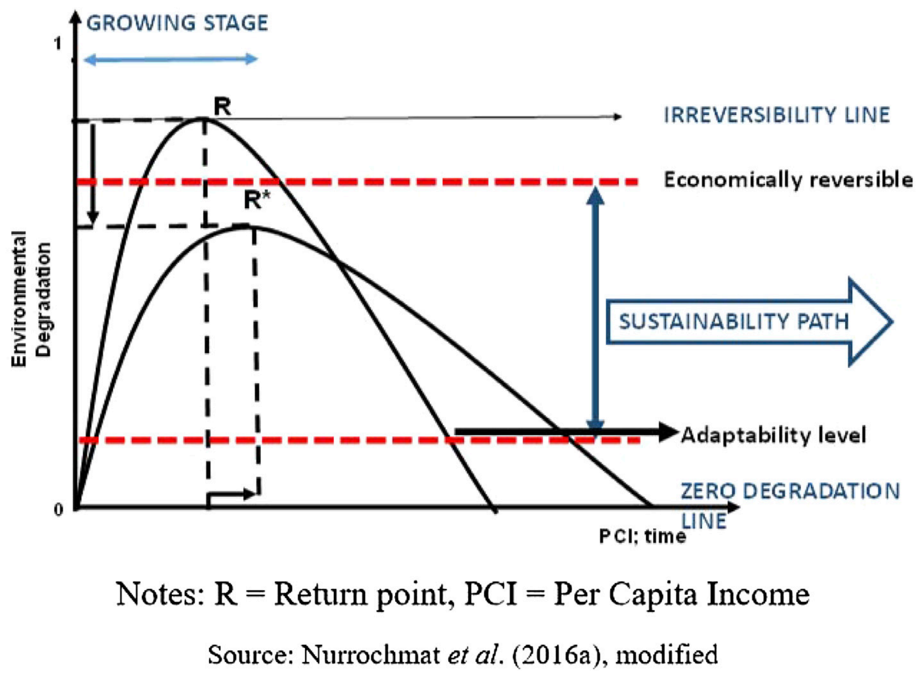

FIGURE 2 | The Concept of reversibility and adaptability in environmental management policy strategies.

immediacy of the detrimental effects of climate change. For this reason, restoration must also consider species adaptation alongside the adaptability of the surrounding social environment. Replacing the existing vegetation with native species or restoring the state of initial vegetation is not always necessarily appropriate to carry out in search of sustainability principles–in terms of ecological, social, economic, technological, and legitimacy aspects (see Ekayani et al., 2015; Sukwika et al., 2016; Nurrochmat et al., 2017; Yovi and Nurrochmat, 2018; Rahmani et al., 2021).

GLOBAL FOREST-REGIMES, NATIONAL POLICIES, AND LOCAL INTERESTS ON SUSTAINABLE DEVELOPMENT

The strategy for managing forest and environmental resources must be in line with the norms of environmental management and social and economic rationality. In some cases, global regime intervention is an obstacle in implementing the best forest resource and environmental management policy (see Nurrochmat et al., 2016a; Erbaugh and Nurrochmat, 2019). Therefore, land and natural resources management strategy should always consider the principles of people's prosperity and national sovereignty, which relies on national and local interests (Pribadi et al., 2020; Nurrochmat et al., 2021).

Not all policies outlined by the government manifests as a reality in the field. In particular, the issue of legality is not always congruent with legitimacy. For example, many regulations attempt to determine forest management outcomes. Still, too many regulations are being violated or deemed illegitimate at different governing scales or other notions of authority. Legitimacy starts with trust in someone or something. Credibility, efficiency, and fairness will be enforced if the implementation of policies is in line with the corresponding characteristics and purposes.

International regimes, especially in the forest sector, influence national legal systems. The hegemony of interpretations of interest groups or certain countries on forest boundaries impact the selection and application of policy strategies and benchmarks for the sustainability of forest resources and environmental performance (see Giessen 2013; Nurfatriani et al., 2015; Sahide et al., 2015, 2015a; Nurrochmat et al., 2016a). In choosing a strategy for environmental policy management, in addition to paying attention to economic linkages, leakages and effectiveness must also be considered (see Nurrochmat et al., 2016b; Sheriffdeen et al., 2020).

The case of oil palm plantations expansion is salient. On a local or national scale, there may be some examples that oil palm causes deforestation, but these dynamics are very different (see Sahide et al., 2015; Nurrochmat et al., 2021). The environmental benefits of oil palm plantations are never equal compared to natural forest ecosystems, and therefore, comparing these two conditions is irrelevant. A much fairer comparison is the benefits and environmental damage caused by various world vegetable oil-producing plants when seen from a broader context. The world's demand for vegetable oils continues to grow. This market is contested by different vegetable oils derived from oil palm, soybeans, rapeseeds, sunflower seeds, and several other vegetable oils (World Growth, 2011). Suppose vegetable oil production from palm oil is halted and substituted by soybean oil, rapeseed, or sunflower seeds whose productivity per land area is only about one-tenth of palm oil (Palm Oil Research, 2017). What are the implications for global deforestation and market demands for vegetable oils? It is by no means an endorsement of palm oil over others, given the different ecologies and economies for which these agricultural products are produced. Instead, this article is trying to highlight the broader context of forest management and sustainability at differing scales of governance.

Contestation of interests has not necessarily created a conflict (Roslinda et al., 2012; Budiono et al., 2017; Astuti et al., 2020). Nurrochmat et al., 2017 promote a concept of changing contestation into collaborative management of forest resources and the environment. Actors or groups of actors each have different interests in resources and the environment. It is impossible to generalize all actors' diverse interests at the global, national, and local scales in natural resource management (see Pribadi et al., 2020; Nurrochmat et al., 2021). However, it is possible to identify the slices of various interests by exploring the intrinsic interests of each actor–which cannot be determined from the question of what the actors are doing. Instead, it becomes evident of they have done or are doing. A land-swap policy may be an excellent example for compromising interests and goals in local and national development (Nurrochmat et al., 2020).

DISCUSSIONS

Sustainable forest development can be implemented effectively if and only if a more appropriate sustainability standard is normative and more reasonable, measurable, and implementable. The condition of "sustainability" in natural resource management is thus limited by the ability of nature to recover and the human ability to adapt, which will form a new dimension of human-nature relations in future generations. Therefore, sustainability should not be defined as transferring resources and the environment to future generations with the same conditions as the initial ones. The situation should be based on what enables future generations to adapt and create ways that allow for a better quality of life.

It concludes that the successful implementation of a forest development policy is also largely determined by the rationality of socio-economic-ecological arguments. The rationality is often measured by the benefits received by a particular community, generally determined by the level of legitimacy of a particular policy rather than its legal aspects. The different interests of actors in managing forest resources do not always cause conflict. Identifying the intrinsic interests of various actors in managing resources and the environment is the key to shifting potential conflicts and transitioning them into opportunities for cooperation and legitimate consensus. Thus, it is imperative to improve the green policy paradigm by considering the principles of reversibility, adaptability, rationality, and legitimacy.

AUTHOR CONTRIBUTIONS

DN discussed the concept; DN collected data and contributed to the analyses; DN and MF drafted the manuscript; DN, MS and MF contributed to revising and refining it.

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REFERENCES

Astuti, E. W., Hidayat, A., and Nurrochmat, D. R. (2020). Community forest Scheme: Measuring Impact in Livelihood Case Study Lombok Tengah Regency, West Nusa Tenggara Province. *Jurnal Manajemen Hutan Tropika*, 26 (1), 52–58. doi:10.7226/jtfm.26.1.52

Budiono, R., Nugroho, B., Hardjantoand Nurrochmat, D. R. (2017). The Village forest as a Counter Territorialization by the Community forest Village in Kampar Peninsula Riau. *Jurnal Manajemen Hutan Tropika* 24 (3), 115–125. doi:10.7226/jtfm.24.3.115

Di Gregorio, M., Nurrochmat, D. R., Paavolaa, J., Sari, I. M., Fatorellia, L., Pramova, E., Locatelli, B., Brockhaus, M., and Kusumadewi, S. D. (2016). Climate Policy Integration in the Land-Use Sector: Mitigation, Adaptation and Sustainable Development Linkages. *Environ. Sci. Pol.* 67, 35–43.

Ekayani, M., Nurrochmat, D. R., Nurrochmat, D. R., Saharjo, B. H., and Erbaugh, J. T. (2015). Assessing Conformity of Scientific Voices and Local Needs to Combat forest Fire in Indonesia. *jtfm* 21 (2), 83–92. doi:10.7226/jtfm.21.2.83

Erbaugh, J. T., and Nurrochmat, D. R. (2019). Paradigm Shift and Business as Usual through Policy Layering: Forest-related Policy Change in Indonesia (1999-2016). *Land Use Policy* 86, 136–146. doi:10.1016/j.landusepol.2019.04.021

Giessen, L. (2013). Reviewing the Main Characteristics of the International forest Regime Complex and Partial Explanations for its Fragmentation. *Int. For. Rev.* 15 (1), 2013. doi:10.1505/146554813805927192

Kuznets, S. (1955). Economic Growth and Income Inequality. *Am. Econ. Rev.* 45 (1995), 1–28.

Letey, J. (1973). "Reversible and Irreversible Ecological Processes," in *Environmental Administration. Edmund & Letey* (Toronto: McGraw-Hill Book Company).

Nurfatriani, F., Darusman, D., Nurrochmat, D. R., Yustika, A. E., and Muttaqin, M. Z. (20152015). Redesigning Indonesian Forest Fiscal Policy to Support forest Conservation. *For. Pol. Econ.* 61, 39–50. doi:10.1016/j.forpol.2015.07.006

Nurrochmat, D. R., Boer, R., Ardiansyah, M., Immanuel, G., and Purwawangsa, H. (2020). Policy Forum: Reconciling palm Oil Targets and Reduced Deforestation: Landswap and Agrarian Reform in Indonesia. *For. Pol. Econ.* 199, 102291. doi:10.1016/j.forpol.2020.102291

Nurrochmat, D. R., Darusman, D. R., and Ekayani, M. (2016b). *Kebijakan Pembangunan Kehutanan & Lingkungan: Teori Dan Implementasi*. Bogor: IPB Press.

Nurrochmat, D. R., Dharmawan, A. H., Obidzinski, K., Dermawan, A., and Erbaugh, J. T. (2016a). Contesting National and International forest Regimes: Case of Timber Legality Certification for Community Forests in Central Java, Indonesia. *For. Pol. Econ.* 68, 54–64. doi:10.1016/j.forpol.2014.09.008

Nurrochmat, D. R., Nugroho, I. A., HardjantoPurwadianto, A., Purwadianto, A., Maryudi, A., and Erbaugh, J. T. (2017). Shifting Contestation into Cooperation: Strategy to Incorporate Different Interest of Actors in Medicinal Plants in Meru Betiri National Park, Indonesia. *For. Pol. Econ.* 83, 162–168. doi:10.1016/j.forpol.2017.08.005

Nurrochmat, D. R., Pribadi, R., Siregar, H., Justianto, A., and Park, M. S. (2021). Transformation of Agro-Forest Management Policy under the Dynamic Circumstances of a Two-Decade Regional Autonomy in Indonesia. *Forests* 12, 419. doi:10.3390/f12040419

Palm Oil Research (2017). *Untangling the Palm Oil Debate. Statistics*. www.palmoilresearch.org/statistics.html (Accessed 03 12 2017).

Peacock, M. (2021). "Amartya Sen and Rational Choice," in *The Concept of Commitment* (Oxfordshire (UK): Routledge).

Pribadi, R., Nurrochmat, D. R., Suhendang, E., and Siregar, H. (2020). Enhancing the Role of the District Government in Decentralized forest Management. *J. Manajemen Hutan Tropika* 26 (2), 114–122. doi:10.7226/jtfm.26.2.114

Rahmani, T. A., Nurrochmat, D. R., Hero, Y., Park, M. S., Boer, R., and Satria, A. (2021). Evaluating the Feasibility of Oil palm Agroforestry in Harapan Rainforest, Jambi, Indonesia. *Fs* 5 (2), 458–477. doi:10.24259/fs.v5i2.10375

Roslinda, E., Darusman, D., Suharjito, D., and Nurrochmat, D. R. (2012). Stakeholders Analysis on the Management of Danau Sentarum National Park Kapuas Hulu Regency West Kalimantan Province. *jtfm* 18 (2), 78–85. doi:10.7226/jtfm.18.2.78

Sahide, M. A. K., Burns, S., Burns, S., Wibowo, A., Nurrochmat, D. N., and Giessen, L. (2015a). Towards State Hegemony over Agricultural Certification: From Voluntary Private to Mandatory State Regimes on Palm Oil in Indonesia. *jtfm* 21 (3), 162–171. doi:10.7226/jtfm.21.3.162

Sahide, M. A. K., Nurrochmat, D. R., and Giessen, L. (2015). The Regime Complex for Tropical Rainforest Transformation: Analysing the Relevance of Multiple Global and Regional Land Use Regimes in Indonesia. *Land Use Policy* 47 (2015), 408–425. doi:10.1016/j.landusepol.2015.04.030

Sen, A. (1999). *Development as Freedom*. Oxford: Oxford University Press.

Sheriffdeen, M., Nurrochmat, D. R., Perdinan, P., and Di Gregorio, M. (20202020). Indicators to Evaluate the Institutional Effectiveness of National Climate Financing Mechanisms. *Fs* 4 (2), 358–378. doi:10.24259/fs.v4i2.10309

Silva, D. (2021). *Brazil's Atlantic Forests Are Naturally Regenerating Much Faster than Expected. A Cost-Effective Solution with Impressive Returns*. Bogor: CIFOR. Forests News, 1st September 2021 https://forestsnews.cifor.org/74340/brazils-atlantic-forests-are-naturally-regenerating-much-faster-than-expected?fnl=en&utm_campaign=CIFOR_ICRAF_Newsletter&utm_medium=email&utm_source=Mailchimp_Newsletter_September.

Sukwika, T., Darusman, D., Kusmana, C., and Nurrochmat, D. R. (20162016). Evaluating the Level of Sustainability of Privately Managed forest in Bogor, Indonesia. *Biodiversitas* 17 (1), 241–248.

UNFCCC (2015). *Key Aspects of the Paris Agreement*. United Nations Framework Convention on Climate Change. https://unfccc.int/process-and-meetings/the-paris-agreement/the-paris-agreement/key-aspects-of-the-paris-agreement (accessed 25 12 2021).

World Bank (2012). *Inclusive Green Growth, the Pathway to Sustainable Development*. Washington DC: World Bank.

World Growth (2011). *The Economic Benefits of Palm Oil to Indonesia. A Report by World Growth*. Arlington: World Growth.

Yovi, E. Y., and Nurrochmat, D. R. (20182018). An Occupational Ergonomics in the Indonesian State Mandatory Sustainable forest Management Instrument: A Review. *For. Pol. Econ.* 91, 27–35. doi:10.1016/j.forpol.2017.11.007

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BRIEF RESEARCH REPORT
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Rapid Water Quality Assessment as a Quick Response of Oil Spill Incident in Coastal Area of Karawang, Indonesia

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The purpose of this study was to assess the effect of oil spills on seawater quality along the coastal waters of Karawang Regency. Several laboratories were involved in measuring water quality to get representativeness of the location of the exposed waters both spatially and temporally. The measurement of seawater quality was carried out *in situ* and in the laboratory. Seawater quality data were compared with quality standards and discussed descriptively. All key water quality parameters (total petroleum hydrocarbon, polycyclic aromatic hydrocarbon, phenol, MBAS, and oil and grease) were below the detection limit of equipment, and a number of metals generally met quality standards. Only shortly after the oil spill in the vicinity of the spill source, the Ni metal exceeded the quality standard. However, after some time, spatially and temporally Ni has met the quality standard. Parameters not related to oil spills such as total phosphate generally did not meet quality standards. This might be related to the high activity on land, such as waste from domestic, industry, and agricultural activities entering coastal waters. Based on intertemporal data, the effect of an oil spill on water quality was temporary. This shows that the handling of the impact of the oil spill has shown good results and the quality of seawater remained quite good. Oil spills that float on the ocean surface were picked up, and those that washed ashore were cleaned up and collected.

Keywords: oil spill, PAH, TPH, MBAS, oil and grease

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INTRODUCTION

The amount of oil entering marine waters worldwide is estimated as around 2.4 million tons per year. Sources of oil pollution are natural seepage and anthropogenic sources such as discharges from storage facilities and refineries, discharge of ballast water from tankers, borehole leaks, and ruptured pipelines, carry-over discharges, industrial and urban discharges, and atmospheric deposits (Tong et al., 1999; Lan et al., 2015). The cause of oil spills is estimated as around 30–50% due to human error either intentionally or unintentionally. Meanwhile, as much as 20–40% are caused by equipment failure or malfunction (Michel and Fingas, 2016).

The oil spill event from well drilling activity off the coast of Karawang that occurred at the end of July 2019 has the potential to have an impact on the condition of seawater quality in coastal waters. The dominant current pattern that leads to the west brings the oil spill to the west coast of Karawang Regency. Pertamina Hulu Energi Offshore North West Java (PHE ONWJ) as the company in charge of the well operations has conducted a number of prevention activities since the oil spill incident through the following steps (Stevens and Auraed, 2008; Xu et al., 2013; Singkran, 2014; Marzooq et al., 2019; Karbela et al., 2020; Haule and Freda 2021):

1) Predict the distribution of oil spills with the oil spill modeling program (MoTum).
2) Deploy a combat team that directly monitors and manages oil spills.
3) Conduct observations with UAVs (unmanned aerial vehicles) and helicopters to ensure that coastal areas are exposed to oil spills.
4) Interpret satellite image data with daily temporal resolution. The satellite images used are Sentinel-1A, Radarsat-2, Cosmo, and Skymed-1.
5) Conduct direct surveys to locations of coastal ecosystems exposed to oil spills.
6) Collaborate with universities to conduct impact studies on various environmental aspects.
7) Coordinate with relevant government agencies in the context of mitigating the impact of the oil spill.

Besides being used as an offshore oil and gas mining area, the coastal waters of Karawang Regency are widely used for various activities such as capture fisheries, aquaculture, marine tourism, and sea transportation. In addition to being the object of receiving the impact of an oil spill incident, various activities located on the coast can also have an impact on the condition of water quality. Therefore, in the impact mitigation activities, sampling of seawater quality is conducted in coastal waters and around river mouths (Stevens and Auraed, 2008; ITOPF, 2014).

Oil spills due to oil and gas exploration and production activities have often occurred in places, on both small and large scales. However, the effect of an oil spill on seawater quality may differ depending on the characteristics of the oil, current patterns, and mitigation and handling (Da Silva et al., 2009; Rout and Sharma, 2013). The aim of this study was to provide an overview of the impact of oil spills on water quality conditions in the coastal waters of Karawang Regency.

MATERIALS AND METHODS

Water quality data were obtained from the results of sampling conducted by Environmental Research Center (ERC) IPB and an accredited independent laboratory appointed by PHE ONWJ. In this study, the selection of key water quality parameters considers the following criteria: 1) having a quality standard value based on KepMenLH 51/2004, 2) related to the general characteristics of an oil spill, and 3) related to impact handling. The key parameters for assessing the impact of an oil spill are set, namely, polycyclic aromatic hydrocarbon (PAH), total petroleum hydrocarbon (TPH), phenol, oil and grease, and MBAS (surfactant) (Ewida 2014; ITOPF, 2014; Eljaiek-Urzola et al., 2019; Han et al., 2021). All key parameters were analyzed by the Indonesian standard of analytical method (SNI).

Sampling of seawater quality was carried out by team 1 from Intertek Laboratory, Syslab, and Bogor Labs under the coordination of PHE ONWJ (17 July–6 November 2019) and team 2 from ERC IPB University (15–17 August 2021). The division of the two teams took into account the wide study area and time. The location and time of sampling of seawater quality by Intertek Laboratory, Syslab, and Bogor Labs and ERC IPB Laboratory are presented in **Figure 1** and **Table 1.**

Sampling of seawater quality by Intertek Laboratory was carried out on 23 July–10 October 2019 at 16 locations starting from around the well (the source of the oil spill), seawaters around the well, to Sedari Beach. At several locations, repeated sampling was carried out to see changes in seawater quality conditions over time so that the total samples analyzed amounted to 25 samples. ERC IPB University conducted seawater sampling on 14–17 August 2019 at 12 points, starting from the Estuary of Tengkolak Beach, Sukakerta, along the west to Tanjung Pakis Beach.

At the beginning of the oil spill (17 July 2019), sampling was carried out at two locations, namely, at a distance of ±250 m southeast of the well (exposed seawater) and at a distance of ±11 km northwest of the well (seawater not exposed).

Water samples were taken from below the surface using a Kemmerer water sampler. The collected water quality data were analyzed spatially between the sampling and control locations, compared with the Ministry of Environment Decree (MoE Decree), No. 51 of 2004 Sea Water Quality Standards, for Marine Biota (attachment 3), and analyzed descriptively.

RESULTS AND DISCUSSION

In locations of seawater exposed to oil spills, TSS was found to reach 845 mg/L, when compared to that in the control area (TSS 4 mg/L), and the TSS quality standard was 20 mg/L. The high TSS is thought to be due to the rise of seabed sediments into the water column and surface layers along with the release of oil from leaking drilling wells (Ifelebuegu et al., 2017).

Data collected from team 1 show that BOD at the location of water exposed to oil spills is greater (5.8 mg/L) than that in the control location (<2.0 mg/L) but still meets the quality standard (20 mg/L). Other parameters that do not meet the quality standards of seawater quality for marine biota (MOE Decree, 2004) at the location of exposed water are total phosphate, copper (Cu), and nickel (Ni). At the control location, Cu also did not match the quality standard so that the Cu concentration at both locations could be considered a general condition.

Ni concentration at the location of water exposed to oil spill 0.12 mg/L has passed the quality standard (0.05 mg/L). This condition describes the water quality at the beginning of the incident because the sampling was conducted on 17 July 2021, when the oil spill was still around the source, not yet dispersed. At locations 1 km, 5–10 km, and 30 km from the source of the spill, Ni levels show concentrations that meet quality standards. Likewise, when sampling was conducted in August, September, and October 2019, Ni met the quality standard and even the concentration was below the detection limit. The presence of Ni may be related to the presence of this element in the fingerprint of YYA-1 well crude oil (0.9 mg/dry kg). However, this element in

FIGURE 1 | Sketch of seawater quality sampling locations in Karawang Regency.

TABLE 1 | Water quality sampling site and time by Intertek Laboratory, Bogor Labs, and ERC IPB.

Cluster	Location code	Location	Sampling date
A. Intertek Laboratory, Syslab, and Bogor Labs (Team 1)			
1. Marine water at the source of oil spill inside YYA-1	Oil-exposed marine water	Distance ±250 m southeast of YYA-1	17 July 2019
	Non-exposed marine water	Distance ±11 km northwest of YYA-1	17 July 2019
	AL-1, AL-2, Al-3, Al-4	AL-1 (Coastal of Sedari), Al-2 (Coastal of Pantai Pelangi), Al-3 (Coastal of Pusaka Jaya), Al-4 (marine water, ±12 km northeast of YYA-1)	23 September 2019
	1 km of source	±1 km northwest of source	07 August 2019, 15 September 2019, 09 October 2019
	5–10 km of source	5–10 km west of source	07 August 2019, 16 September 2019, 09 October 2019
	30 km of source	Coastal of Pantai Sedari, 30 km west of source	07 August 2019, 16 September 2019, 11 October 2019
	Control (area outside oil spill distribution)	±10 km southeast of source	17 September 2019, 10 October 2019
2. Karawang Regency	Sedari (outside oil spill distribution)	3.7 km off Sedari Beach	06 August 2019, 28 September 2019, 20 October 2019
	Sedari 1 and Sedari 2 (within oil spill distribution)	250 m off Sedari Beach	06 August 2019, 28 September 2019, 20 October 2019
	Pantai Pelangi and Mouth Dobolan River	Pantai Pelangi and Mouth Dobolan River	05 November 2019
	Marine water of Sedari	Marine water of Sedari	06 November 2019
B. ERC IPB University (Team 2)			
3. Karawang Regency	4, 5	4 (Pantai Tanjung Pakis), 5 (Pantai Segar Jaya)	15 August 2019
	6, 7, 8, 9	6 (Estuary of Pantai Pelangi), 7 (Estuary of Sungai Buntu), 8 (Estuary of Pusaka Jaya), 9 (Pantai Ciparage)	16 August 2019
	10, 11, 12, 13	10 (Pantai Cemara), 11 (Pantai Sedari), 12 (Pantai Tambak Sari), 13 (Pantai Tambak Sumur)	15 August 2019
	14, 15	14 (Estuary of Pantai Tengkolak, Sukakerta), 15 (Pantai Pasir Putih, Sukajaya)	17 August 2019

seawater is immediately diluted and scattered by ocean currents spatially and temporally.

Based on seawater quality data around the oil spill source, it is known that the presence of an oil layer was observed by Intertek Laboratory, Syslab, and Bogor Labs at a location 1 km to the northwest from the well on 7 August 2019 and 15 September 2019. This is an indication that there is an oil spill that has escaped the obstruction of the oil boom installed around the source of the oil spill. However, based on the results of laboratory analysis from the sampling location, it can be assessed that the presence of an oil layer on the sea surface has no significant effect on the overall water quality condition because all parameters related to oil such as PAHs, TPHs, phenols, detergents (MBAS), and oil and grease meet quality standards, and even the concentration is below the detection limit.

The parameter that may be affected by the presence of an oil layer is the dissolved oxygen (DO) content. The results of *in situ* measurements showed a DO concentration of 4.4 mg/L on 7 August 2019 at a sampling location 1 km from the source, when at this location a layer of floating oil was also found on the surface. However, at the same location and time, the brightness (9 m) and turbidity (0.7 NTU) were still in the good category, meeting the quality standard. On the contrary, the results of measurements on 15 September 2019 at a sampling location of 1 km showed a relatively good DO content (5.75 mg/L), 3 m brightness, and low turbidity (<0.5 NTU). Similar phenomena were found by Chen et al (2017) who studied the impact of two oil spill events on the water quality along the coastal area of Kenting National Park, Taiwan.

Observing the phenomena that occur, the effect of an oil spill on DO parameters, turbidity, and brightness is highly dependent on the thickness and area of the oil layer on the water surface. The thicker the oil layer, the lower the penetration of sunlight into the water column, which can continue to disrupt photosynthesis by plankton so that it indirectly causes the dissolved oxygen level in the water to become low (González et al., 2009). The oil layer directly can also cause the dissolved oxygen level in the water to become low due to the obstruction of the diffusion process from the air (Ifelebuegu et al., 2017). In this incident, oil spills were intercepted by oil booms and skimmers and immediately picked up periodically, so that there were not many oil spills that escaped from the oil boom barrier, only in the form of lumps that were not continuous, and were washed away by currents to the west from the source of the spill.

Parameters of seawater quality around oil spills that do not meet quality standards generally indicate contamination from domestic waste, namely, relatively high levels of nutrients, especially total phosphate. In almost all sampling times and locations, total phosphate did not meet the quality standard (0.015 mg/L). The presence of phosphate in water is generally caused by anthropogenic pollution such as the use of detergents, run-off from agricultural fertilizers, industrial waste, and domestic waste (Daneshgar et al., 2018). Sampling locations are more dominant on the coast, so that the influence of river flows that carry untreated domestic wastewater is visible, supported by high turbidity. Seawater quality data from sampling in early October 2019 (after the well was successfully closed at the end of September 2019) also still show total phosphate concentrations that do not meet the quality standards, thus confirming that this condition is not due to an oil spill.

The characteristic of volatile oil is indicated by the presence of odor in the air around the well. However, at all times and locations of sampling carried out by the Intertek Team, no significant odor was recorded from the presence of an oil layer on the surface of water because the proportion of oil that was washed away was much less than the proportion of oil that was captured, collected, and brought to the mainland for further handling. This condition can also be related such that offshore waters are open and air circulation tends to be better than that on the coast, so that the smell of the oil layer does not feel significant.

In addition to water quality sampling, Intertek Laboratory also measures air quality around platforms and residential areas on the coast. The results of ambient air quality measurements at the well location on 15 September 2019 showed that the hydrocarbon content in the ambient air was 537 μg/Nm^3. This concentration exceeds the ambient air quality standard based on PP 41 of 1999, which is 160 μg/Nm^3. The hydrocarbon content is much greater than the results of sampling at the same location on 9 October 2019 or after the well leak was successfully handled, which is <50 μg/Nm^3. The hydrocarbon concentration in the ambient air can strengthen the explanation that the well oil tends to be volatile or undergoes evaporation. There are a number of volatile oil fractions which are short chains. The long carbon chain oil fraction was carried by ocean currents away from the spill source. The lower carbon numbers are gases, intermediate compounds are liquids, and higher members of the series are solids (Nolan, 2019). Evaporation is the process of weathering the oil with the largest portion. Evaporation reduces significantly the volume of oil remaining in the air or soil after a spill (Lindgren and Lindblom, 2004). Fate of oil contamination in the marine environment includes spreading and advection, evaporation, dissolution, dispersion, emulsification, photo-oxidation, sedimentation and shoreline stranding, and biodegradation (Stevens and Auraed, 2008; Gong et al., 2013; Passow and Overton, 2021).

The results of observations from Intertek Laboratory, Syslab, and Bogor Labs in the waters of Sedari Beach (**Table 2**) in August–October 2019 did not find the presence of an oil layer on the sea surface, both in the exposure area and outside the exposure. The content of PAH, MBAS, phenol, and oil and grease was all below the detection value. Likewise, the dissolved metal parameters all meet the quality standards. Seawater quality parameters that do not meet the quality standards are generally an indication of the influence of domestic, industrial, and agricultural wastewater contamination which is characterized by high concentrations of total phosphate and nitrate.

The parameters of seawater quality that are the main concern by team 2 (ERC IPB University) are focused on parameters directly related to oil spills such as the presence of an oil layer on the surface, odor, TPH (total petroleum hydrocarbon), PAH (polycyclic aromatic hydrocarbon), phenol, MBAS, oil and grease, and metals. In addition to turbidity, in general the

TABLE 2 | Coastal water quality of Karawang (ERC IPB University).

No.	Parameter	Unit	Standard[a]	Coastal of Tanjung Pakis (4)	Coastal of Segar Jaya (5)	Estuary of Pantai Pelangi (6)	Estuary of Sungai Buntu (7)	Estuary of Pusaka Jaya (8)	Coastal of Ciparage (9)
Physical Characteristic									
1	TSS	mg/l	80	40	27	25	36	29	36
2	Turbidity	NTU	<5	9.04	23.5	9.22	1.46	2.16	2.16
3	Transparency	cm	300	5	5	5	44	47	50
4	Oil film	—	Nil	Nil	Nil	Nil	Nil	Nil	Nil
5	Color	—	—	Brown	Grayish brown	Brown	Greenish brown	Brown	Light brown
6	Odor	—	Natural	No odor	No odor	No odor	No odor	No odor	No odor
Chemical Characteristic									
1	pH	—	7–8.5	7.87	7.98	7.8	7.83	7.88	8.02
2	Salinity	psu	Natural	30	31	34	31	31	34
3	Phenol	mg/l	0.002	Bdl	Bdl	Bdl	Bdl	Bdl	Bdl
4	Oil and grease	mg/l	1	Bdl	Bdl	Bdl	Bdl	Bdl	Bdl
5	MBAS (surfactant)	mg/l	1	0.09	0.11	0.18	0.08	0.17	0.05
6	PAH	mg/l	0.003	Bdl	Bdl	Bdl	Bdl	Bdl	Bdl
7	TPH	mg/l	-	Bdl	Bdl	Bdl	Bdl	Bdl	Bdl
8	Cu	mg/l	0.008	Bdl	Bdl	Bdl	Bdl	Bdl	Bdl
9	Cd	mg/l	0.001	Bdl	Bdl	Bdl	Bdl	Bdl	Bdl
10	Pb	mg/l	0.008	Bdl	Bdl	Bdl	Bdl	Bdl	Bdl
11	Hg	mg/l	0.100	Bdl	Bdl	Bdl	Bdl	Bdl	Bdl
12	As	mg/l	0.012	Bdl	Bdl	Bdl	Bdl	Bdl	Bdl
13	Zn	mg/l	0.050	0.14	0.1	0.03	0.04	0.05	0.1
14	Ni	mg/l	0.050	Bdl	Bdl	Bdl	Bdl	Bdl	Bdl
15	Cr(VI)	mg/l	0.005	Bdl	Bdl	Bdl	Bdl	Bdl	Bdl
No.	**Parameter**	**Unit**	**Standard[a]**	**Coastal of Cemara (10)**	**Coastal of Sedari (11)**	**Coastal of Tambak Sari (12)**	**Coastal of Tambak Sumur (13)**	**Coastal of Tengkolak Sukakerta (14)**	**Coastal of Pasir Putih Sukajaya (15)**
Physical Characteristic									
1	TSS	mg/l	80	27	23	35	37	31	23
2	Turbidity	NTU	<5	7.28	1.81	3.22	3.4	8.51	7.28
3	Transparency	cm	300	8	8	7	7	7	25
4	Oil film	—	Nil	**Present**	Nil	Nil	Nil	Nil	Nil
5	Color	—	—	Brown	Brown	Brown	Brown	Light brown	Light brown
6	Odor	—	Natural	Odor	No odor	No odor	No odor	No odor	No odor
Chemical Characteristic									
1	pH	-	7–8.5	8.05	8.09	8.07	8.08	7.94	7.91
2	Salinity	psu	Natural	33	33	33	33	33	31
3	Phenol	mg/l	0.002	Bdl	Bdl	Bdl	Bdl	Bdl	Bdl
4	Oil and grease	mg/l	1	Bdl	Bdl	Bdl	Bdl	Bdl	Bdl
5	MBAS (surfactant)	mg/l	1	0.07	0.2	0.08	0.07	0.1	0.2
6	PAH	mg/l	0.003	Bdl	Bdl	Bdl	Bdl	Bdl	Bdl
7	TPH	mg/l	—	Bdl	Bdl	Bdl	Bdl	Bdl	Bdl
8	Cu	mg/l	0.008	Bdl	Bdl	Bdl	Bdl	Bdl	Bdl
9	Cd	mg/l	0.001	Bdl	Bdl	Bdl	Bdl	Bdl	Bdl
10	Pb	mg/l	0.008	Bdl	Bdl	Bdl	Bdl	Bdl	Bdl
11	Hg	mg/l	0.100	Bdl	Bdl	Bdl	Bdl	Bdl	Bdl
12	As	mg/l	0.012	Bdl	Bdl	Bdl	Bdl	Bdl	Bdl
13	Zn	mg/l	0.050	0.04	0.02	0.01	0.01	0.02	0.01
14	Ni	mg/l	0.050	Bdl	Bdl	Bdl	Bdl	Bdl	Bdl
15	Cr(VI)	mg/l	0.005	Bdl	Bdl	Bdl	Bdl	Bdl	Bdl

[a]*Quality standard (MoE Decree No. 51/2004 Marine Biota). Bdl, below detection limit.*

observed seawater quality parameters meet the quality standards (MOE Decree, 2004). TSS in all observation locations ranged from 22 to 42 mg/L, was still classified as moderate, and met the quality standard (80 mg/L). Turbidity levels vary, in some coastal locations and river mouths. Turbidity >5 NTU, exceeding the quality standard, was seen at nine observation locations, namely, Muara Pantai Pelangi, Tanjung Pakis, Pantai Segar Jaya, Pantai Cemara, Pantai Sedari, Tambak Sedari, Pantai Pasir Putih, dan Muara, and Pantai Tengkolak. Turbidity is related to the condition of coastal sediments which are generally fine muddy and muddy sand that is easily stirred due to waves and currents.

Physical parameters, pH, and salinity are not affected by oil spills. A similar phenomenon was encountered by Ewida (2014) who conducted research on the Nile River and also found no effect on the physical parameters of waters due to oil spills.

Based on the concentration of phenols, oil and grease, surfactants (MBAS), PAHs, TPHs, and dissolved metals, overall they were low and even undetectable (**Table 2**). Indications of contamination were seen at Pantai Cemara, due to the presence of a layer of oil in the waters and the smell of oil at the time of observation (16 August 2019). However, seawater quality in general is not affected by oil spill events because all concentrations of seawater quality parameters related to oil meet the quality standards.

This condition can be attributed to the rapid response efforts in the form of prevention by installing oil booms and skimmers since the occurrence of the oil spill in the vicinity of the leaking oil well. This effort can significantly reduce the volume of oil that reaches the surrounding coastal waters. Given the nature of oil which has a relatively high paraffin wax content (29.32%Wt), it tends to be in the form of small lumps of oil that float on the surface and do not mix or dissolve in water. Biodegradation can remove up to 60% of spilled oil. The photochemical process can convert an amazing oil by up to 50%. Some of the oil on the surface that is undergoing weathering will be washed up along the coastline (Passow and Overton, 2021). High air temperatures and sea breeze speeds can increase the weathering of oil (Lindgren and Lindblom, 2004). This natural process can reduce the volume of oil spilled in seawaters (Wang et al., 2016).

CONCLUSION

Some of the key parameters for assessing the impact of an oil spill are PAH, TPH, phenol, oil and grease, and MBAS (surfactant). All key water quality parameters and a number of metals generally meet the quality standards. Just moments after the oil spill in the vicinity of the spill source, the Ni metal exceeds the quality standard. However, after some time, spatially and temporally Ni has met the quality standard.

Parameters that are not related to oil spills such as total phosphate generally do not meet the quality standards. This may be related to the high activity on land, such as waste from domestic, industrial, and agricultural activities entering coastal waters.

Based on intertemporal data, the effect of an oil spill on water quality is temporary. This shows that the handling of the impact of the oil spill has shown good results and the quality of seawater remained quite good. Oil spills that float on the ocean surface are picked up, and those that washed ashore are cleaned up and collected.

DATA AVAILABILITY STATEMENT

The original contributions presented in the study are included in the article/supplementary material, further inquiries can be directed to the corresponding author.

AUTHOR CONTRIBUTIONS

HE as the first author contributed to the whole process of manuscript writing. Mursalin and SH were responsible for the planning of sampling, GIS analysis, and data interpretation.

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REFERENCES

Chen, C.-C., Tew, K. S., Ho, P.-H., Ko, F.-C., Hsieh, H.-Y., and Meng, P.-J. (2017). The Impact of Two Oil Spill Events on the Water Quality along Coastal Area of Kenting National Park, Southern Taiwan. *Mar. Pollut. Bull.* 124 (2), 974–983. doi:10.1016/j.marpolbul.2017.02.034

Da Silva, E. M., Aguiar, M. C. P., Navarro, M. D. F. T., and Chastinet, C. D. B. E. A. (2009). Impact of Petroleum Pollution on Aquatic Coastal Ecosystems in Brazil. *Environ. Toxicol. Chem.* 16 (1), 112–118. doi:10.1002/etc.5620160112

Daneshgar, S., Callegari, A., Capodaglio, A., and Vaccari, D. (2018). The Potential Phosphorus Crisis: Resource Conservation and Possible Escape Technologies: A Review. *Resources* 7, 37. doi:10.3390/resources7020037

Eljaiek-Urzola, M., Romero-Sierra, N., Segrera-Cabarcas, L., Valdelamar-Martínez, D., and Quiñones-Bolaños, É. (2019). Oil and Grease as a Water Quality Index Parameter for the Conservation of Marine Biota. *Water* 11, 856. doi:10.3390/w11040856

Ewida, A. Y. I. (2014). Oil Spills: Impact on Water Quality and Microbial Community on the Nile River, Egypt. *Int. J. Environ.* 3 (4), 192–198.

Gong, Y., Zhao, X., Cai, Z., O'Reilly, S. E., Hao, X., and Zhao, D. (2013). A Review of Oil, Dispersed Oil and Sediment Interactions in the Aquatic Environment: Influence on the Fate, Transport and Remediation of Oil Spills. *Mar. Pollut. Bull.* 79 (1-2), 16–33. doi:10.1016/j.marpolbul.2013.12.024

González, J., Figueiras, F. G., Aranguren-Gassis, M., Crespo, B. G., Fernández, E., Morán, X. A. G., et al. (2009). Effect of a Simulated Oil Spill on Natural Assemblages of marine Phytoplankton Enclosed in Microcosms. *Estuar. Coast. Shelf Sci.* 83, 265–276. doi:10.1016/j.ecss.2009.04.001

Han, H., Huang, S., Liu, S., Sha, J., and Lv, X. (20212021). An Assessment of Marine Ecosystem Damage from the Penglai 19-3 Oil Spill Accident. *Jmse* 9, 732. doi:10.3390/jmse9070732

Haule, K., and Freda, W. (2021). Remote Sensing of Dispersed Oil Pollution in the Ocean-The Role of Chlorophyll Concentration. *Sensors* 21 (10), 3387. doi:10.3390/s21103387

Ifelebuegu, A. O., Ukpebor, J. E., Ahukannah, A. U., Nnadi, E. O., and Theophilus, S. C. (2017). Environmental Effects of Crude Oil Spill on the Physicochemical and Hydrobiological Characteristics of the Nun River, Niger Delta. *Environ. Monit. Assess.* 189, 173. doi:10.1007/s10661-017-5882-x

ITOPF (2014). *Sampling and Monitoring of marine Oil Spill*. Technical Information paper 14. 16 p. London.

Karbela, B., Mayestika Afgatiani, P. M., and Parwati, E. (2020). Interseasonal Variability in the Analysis of Total Suspended Solids (TSS) in Surabaya Coastal Waters Using Landsat-8 Satellite Data. *Int. J. Remote Sensing Earth Sci.* 17 (2). doi:10.30536/j.ijreses.2020.v17.a3418)

Lan, D., Liang, B., Bao, C., Ma, M., Xu, Y., and Yu, C. (2015). Marine Oil Spill Risk Mapping for Accidental Pollution and its Application in a Coastal City. *Mar. Pollut. Bull.* 96 (1-2), 220–225. doi:10.1016/j.marpolbul.2015.05.023

Lindgren, C., and Lindblom, E. (2004). *Short Term Effects of Accidental Oil Pollution in Waters of the Nordic Countries*. Stockholm, Sweden: IVL Swedish Environmental Research Institute, 48.

Marzooq, H., Naser, H. A., and Elkanzi, E. M. (2019). Quantifying Exposure Levels of Coastal Facilities to Oil Spills in Bahrain, Arabian Gulf. *Environ. Monit. Assess.* 191 (3), 160. doi:10.1007/s10661-019-7287-5

Michel, J., and Fingas, M. (2016). *Chapter 7: Oil Spills: Causes, Consequences, Prevention, and Countermeasures*. World Scientific Series in Current Energy Issues Fossil Fuels, 159–201. doi:10.1142/9789814699983_0007

MOE Decree (2004). *Sea Water Quality Standards, for marine Biota*. Jakarta, Indonesia.

Nolan, D. P. (2019). *Physical Properties of Hydrocarbons and Petrochemicals in Handbook of Fire and Explosion Protection Engineering Principles for Oil, Gas, Chemical, and Related Facilities*. Fourth Edition. Elsevier.

Passow, U., and Overton, E. B. (2021). The Complexity of Spills: The Fate of the Deepwater Horizon Oil. *Annu. Rev. Mar. Sci.* 13, 109–136. doi:10.1146/annurev-marine-032320-095153

Rout, C., and Sharma, A. (2013). *Oil Spill in Marine Environment: Fate and Effects. 2nd International Conference on Evolution in Science & Technology & Eyne on Educational Methodologies March 3-4, 2013, Esteem-2013, PPIMT, Hisar (India)*. Hisar, India.

Singkran, N. (2014). Contingency Plan Improvement for Managing Oil Spills in the Coastal Waters of Thailand. *Mar. Pollut. Bull.* 89 (1-2), 149–159. doi:10.1016/j.marpolbul.2014.10.013

Stevens, L., and Auraed, D. (2008). *Criteria for Evaluating Oil Spill Planning and Response OperationsA Report to IUCNThe World Conservation Union*. Lusby, MD: Ecosystem Management & Associates, Inc. Technical Report 07-02 (Revised June 2008), 55p.

Tong, S. L., Goh, S. H., Abdulla, A. R., Tahir, N. M., and Wang, C. W. (1999). Asean marine Water Quality Criteria for Oil and Grease. *ASEAN-Canada CPMS-II Coop. Programme Mar. Sci.*, 28.

Wang, J., Sandoval, K., Ding, Y., Stoeckel, D., Minard-Smith, A., Andersen, G., et al. (2016). Biodegradation of Dispersed Macondo Crude Oil by Indigenous Gulf of Mexico Microbial Communities. *Sci. Total Environ.* 557-558, 453–468. doi:10.1016/j.scitotenv.2016.03.015

Xu, Q., Li, X., Wei, Y., Tang, Z., Cheng, Y., and Pichel, W. G. (2013). Satellite Observations and Modeling of Oil Spill Trajectories in the Bohai Sea. *Mar. Pollut. Bull.* 71 (1-2), 107–116. doi:10.1016/j.marpolbul.2013.03.028

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Determining High Conservation Values in Production Landscapes: Biodiversity and Assessment Approaches

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Conversion of natural, heterogenous tropical forests to intensively managed, monoculture-production landscapes is a major threat to biodiversity. This phenomenon is driven by global demand for commodities such as wood, palm oil, sugar, and soybean. The economies of many countries in tropical areas depend on these commodities, and there is a need to ensure economic welfare while protecting biodiversity. Certification schemes such as those developed by the Forest Stewardship Council and Roundtable for Sustainable Palm Oil are intended to provide incentive to companies to employ environmentally and socially sustainable production practices. One element of these certification schemes is the concept of High Conservation Values (HCVs) which fall into six categories that encompass ecological indicators and human dimensions. The HCV process has expanded beyond production landscapes to include long-term conservation planning. Despite expansion, implementation of the HCV process as it pertains to biodiversity is challenged, in part, by a lack of specificity regarding target metrics. Another challenge is that, in practice, there is a short time period for assessment, resulting in limited collection of primary data and a reliance on secondary data sources for interpolation. HCV guidance advances a precautionary approach to assessment, but in some regions, there is not enough known about the biology, behavior, or interspecific associations of species to effectively assess what is not observed. In this paper, we assess environmental HCVs in a well-studied timber production system in Sarawak, East Malaysia. Using an original long-term multi-method dataset of avifaunal surveys as well as published datasets of other taxa, we 1) assess biodiversity metrics at the site including presence of Rare, Threatened, and Endemic species, 2) assess change over time at assessment locations, and 3) evaluate costs and benefits of the various methods and provide best practice recommendations for HCV assessment and long-term monitoring. Finally, we recommend transparent data-archiving and sharing for improved accuracy and efficiency in the HCV process. Managed landscapes are important areas for ecological research that are beneficial not only to the restoration and conservation of species and ecosystems but also to well-informed certification and long-term sustainability.

Keywords: high conservation value, rare threatened and endemic species, biodiversity, avifauna, production landscape, native forest, timber plantation

1 INTRODUCTION

Tropical forest ecosystems represent less than 10% of the earth's land area but harbor over 60% of documented biodiversity (Bradshaw et al., 2009). Conversion of natural, heterogenous forests to intensively managed, monoculture-production landscapes is a major cause of forest loss in tropical regions and a primary threat to biodiversity (Giam 2017). Monoculture production systems have lower levels of biodiversity, fewer rare and threatened species, and are more difficult to restore than natural forests managed for timber (Edwards et al., 2010). However, there is increasing pressure to convert both logged and old growth forests to plantation landscapes for commodities such as wood, palm oil, sugar, and soybean (Wilcove and Koh 2010; Carrasco et al., 2014; Griscom and Goodman 2015). The extent of forested land that is in some form of production far exceeds protected forests both in tropical regions and globally (Meijaard and Sheil 2012). These commodities account for a substantial proportion of GDP (Purnomo et al., 2020) and are essential to the economic growth of countries. While halting this process is not possible, there are approaches to land-use planning and change that can reduce the amount of native forest loss and sustain viable populations of many species within production landscapes.

One strategy to address this issue is certification and green labeling. This is a consumer-focused approach in which the buyer may select products that have been accredited to meet certain environmental and social standards. Certification schemes such as those developed by the Forest Stewardship Council (FSC) are intended to provide incentive to companies to use environmentally and socially sustainable production practices (Kollert and Lagan 2007, but see; Chen et al., 2010). To become FSC-certified, companies must demonstrate a commitment to ten criteria associated with legal compliance, worker, community, and indigenous rights, environmental impact reduction, and ongoing monitoring and planning. There is debate over the long-term benefits of certification to biodiversity protection (e.g., Edwards et al., 2010; Carlson et al., 2017; Trolliet and Vogt, 2019), but the demand for certified products has increased in recent years, suggesting that social pressure will continue to influence commodity and natural resource markets (Dinerstein et al., 2019; The Economist Intelligence Unit 2021).

One element of certification (FSC Criterion 9) is the concept of High Conservation Values (HCVs), which falls into six broad categories that encompass ecological indicators and human dimensions associated with ecosystem health (Brown et al., 2013; Forest Stewardship Council 2015). The HCV process is increasingly used as a tool for land use planning at large spatial scales (Senior et al., 2015; Areendan et al., 2020).

Despite increasing use of the HCV process in conservation, challenges to effective implementation as it pertains to biodiversity and ecosystem values remain, due, in part, to a lack of specificity regarding target metrics (Senior et al., 2015; Tayleur et al., 2016). Also, in practice, there is a short time period for environmental assessment, resulting in limited collection of primary data (rapid field surveys at a small number of sites) and a reliance on secondary data sources for interpolation (Sollmann et al., 2017). HCV guidance recommends a precautionary approach to assessment, but in some regions (e.g., certain tropical regions with high levels of biodiversity), not enough is known about the biology, behavior, or interspecific associations of species to assess what is not observed.

In this paper, we present an assessment of biodiversity HCVs from a well-studied timber production system in Sarawak, East Malaysia: Sarawak Planted Forests (SPF). Using an original long-term multi-method dataset of avifaunal surveys as well as published datasets of other taxa, we 1) evaluate the presence of Rare, Threatened, and Endemic (RTE) species at the site as well as changes over time in species presence, 2) compare detection rates of different field methods, taking into account differences in spatiotemporal coverage, and 3) compare costs and benefits of the various methods. We place the work within the broader context of the HCV process by comparing biodiversity information from SPF to that presented in summary statements of successful HCV applications on the island of Borneo.

2 MATERIALS AND METHODS

2.1 Study Site

Field research was conducted within Sarawak Planted Forests (SPF; **Figure 1**), located within Bintulu Division (2.83°, 113.22°). Established in 1997, SPF was the first large-scale exotic timber plantation in Sarawak (Ellis 2007; Hall et al., 2007; Stuebing 2007). It was established near Bintulu, in part, because much of the landscape was already disturbed (logged native forest) or altered (primarily shifting cultivation). It encompasses an area of approximately 500,000 ha and includes ca 200,000 ha of exotic fast-growing softwoods: primarily *Acacia mangium*. The remaining area consists of designated Native Customary Rights land, and several large tracts of native forest intended to serve as both primary habitat and protective corridor for sensitive species (Stuebing 2007). Within the native forest area there is one Totally Protected Area known as Binyo-Penyilam National Park (BPNP, on paper called Danau Mujan), one protected area called Bukit Sarang (BS), a large area of previously logged (ca. 30 years ago) forest called Bukit Mina Wildlife Corridor (BMWC), and an adjacent area of native kerangas forest known as Bukit Nyegoh (BN). These areas include high priority ecosystems for protection, including peat swamp forest (BPNP), freshwater swamp (BS), and kerangas (BN) (WWF, 2009). Several national parks are adjacent to or near SPF including Bukit Mersing NP (adjacent), Bukit Kana NP (adjacent), Sungai Meluang NP (16 km from SPF), Bukit Tiban NP (18 km), Similajau NP (20 km), Hose Laga NP (28 km), and Bakun Islands NP (38 km). There is also considerable native forest within the SPF and to the South and East (**Figure 1**).

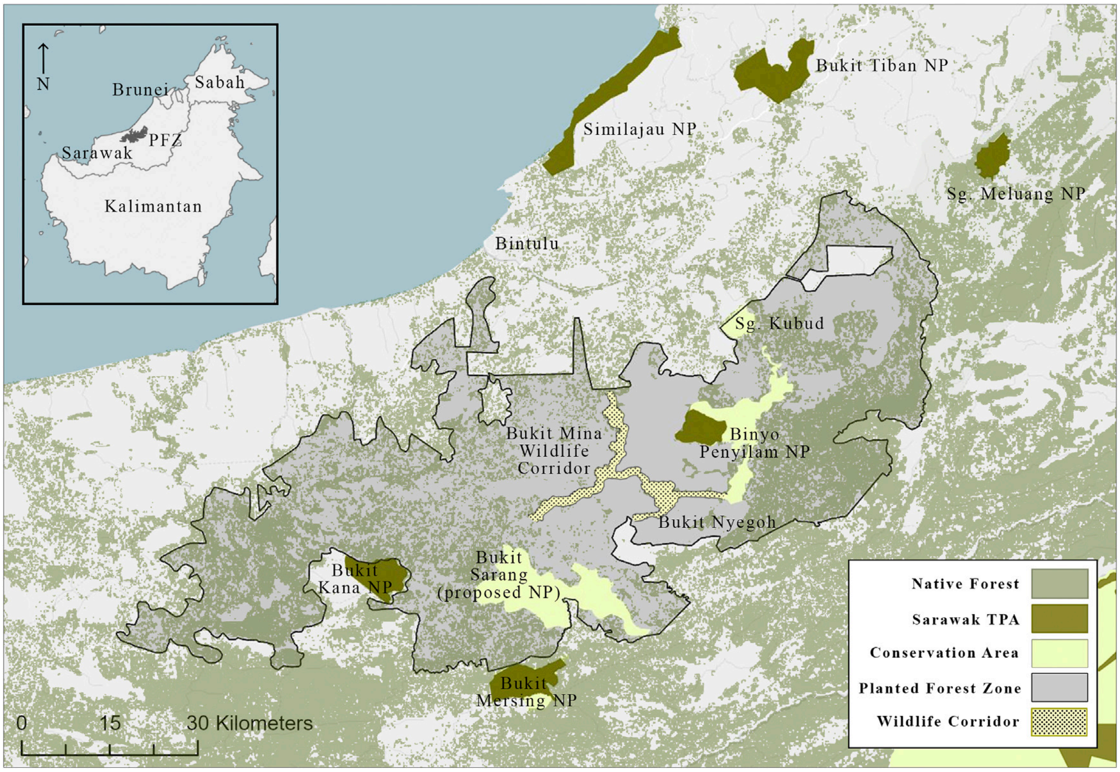

FIGURE 1 | Map of Sarawak Planted Forests and surrounding landscape. The area indicated as native forest is forest that was previously logged (30 + years ago). TPA, Totally Protected Area; SPF, Sarawak Planted Forests. Areas labelled as Conservation Area are areas within SPF that are designated for conservation but have no formal protection otherwise.

2.2 Field Methods

2.2.1 Point Count (PC) Surveys

Point count surveys were conducted in July–August 2006 (BMWC, n = 216BPNP, n = 80) and September 2019 (BPNP, n = 80). Trained observers conducted timed point counts every 50 m along transects that were 1,000 m in length from 0600–0930 (Styring et al., 2011). For a duration of 3 min, observers documented every bird detected (visually or aurally) and estimated the distance of the bird from the observer using laser rangefinders (Buckland et al., 2015). In 2019, each point count was also recorded with a digital recorder. Habitat data were also collected at each point including variables associated with forest type, habitat complexity, and stand structure (**Supplementary Table S1**).

2.2.2 Bird Banding (BB)

Bird banding was conducted using a standardized capture array at BMWC and BPNP in 2018 and 2019. We established a minimum of three ~1 ha net plots at each site with 36 net-meters distributed on each side and on a transect through the midpoint (see **Figure 1** for an example). Nets were deployed over three to 4 days between dawn and an hour before dusk to accumulate a minimum of 15 h of net operations including at least two dawn and one evening capture period. Once net arrays had accumulated the minimum passive-capture target hours as well as the requisite dawn and dusk sessions, we deployed a select or full array on a final morning to target under-sampled species using sound-baiting with song playback or alarm calls. Netting was conducted under safety standards of the North American Banding Council (NABC, 2001). All birds captured were processed by marking individuals with a numbered aluminum band to keep track of recaptures. Beyond the circumstances of capture, the data collected from each individual included species, physiological evidence relating to age (skull ossification and plumage characteristics), sex (plumage, measurements and breeding condition, including cloacal protuberance, brood patch and eggs in the oviduct), condition (fat score and mass as well as plumage characteristics such as feather wear and plumage aspect), and molt status (active molt of flight feather and body tracts). Data categorizations largely followed the Institute for Bird Populations' Monitoring Avian Productivity and Survivorship Protocol (DeSante et al., 2021). When possible, we took an in-hand series of photographs showing head, back, underparts, a spread-wing and tail, as well as a series of photographs of remiges and wing coverts using a Dino-Lite Digital USB microscope at ~×25 magnification. Image data were cross-linked with field data and organized and accessioned into Adobe Bridge.

2.2.3 Dawn Chorus Surveys (DC)

Observers conducted dawn chorus surveys starting 15 min before dawn and lasting approximately one hour after dawn at BPNP (14–18, 27 September; 3 October 2019), BMWC (9–10, 12–20, 22–25. 27–28 March; 4–6. 8–12 April 2018), and BSCA (13–16 June 2017) (Parker, 1991). Observers slowly walked established survey routes while recording sounds with a field recorder and directional microphone. The recordist slowly moved the microphone in a circle and at angles from directly overhead to downslope. An additional field observer accompanied the recordist and quietly pointed out new calls and songs.

2.2.4 Autonomous Recorders (ARs)

Weatherproof, programmable recording units were deployed at BSCA (13 June–12 July 2017; $n = 1$), Bukit Nyegoh (9 March to 4 April 2018; $n = 1$) and BPNP (14 September–4 October 2019, $n = 8$). The recorders were programmed to record at two, 2-h intervals at dawn and dusk (starting from 15 min pre-dawn and again from 105 min pre-dusk). Recordings were saved to SD card and downloaded to Arbimon https://arbimon.rfcx.org/) for pattern matching analysis.

2.3 Analysis

2.3.1 Biodiversity Inventories

2.3.1.1 Avifauna

A master checklist of birds observed in SPF was compiled from surveys from 2005 to 2019. Most of the survey work was conducted by the current authors and researchers from Louisiana State University Museum of Natural Science (2006, 2007, 2008, 2010, 2011, 2017, 2018, and 2019). Other surveys included: a 2005 rapid assessment by a research team from the Academy of Sciences of Philadelphia, a 2009 rapid assessment of the Bukit Sarang area by field guide author Quentin Phillips, and a 2018 survey by a research team from the Smithsonian Institution. The species list generated from these surveys represents both systematic surveys (described in field methods) as well as incidental observations.

2.3.1.2 Other Taxa

We conducted a systematic literature review using the following search terms in google scholar: “Sarawak Planted Forests”, Sarawak + “Planted Forests Zone”, Sarawak + “PFZ”, “SPFZ”, “SPF”, “Bukit Sarang”, “Bukit Mina”, “Bukit Nyegoh”, “Binyo-Penyilam”, “Binyo”, and “Penyilam”. Results were then filtered to include studies with titles that covered the topics of ecology, biodiversity, taxonomy, conservation, forestry, or the name of a specific plant or animal. The remaining articles where then read to determine if presence information for one or more animal species was included in the study. Information from articles that included species information was then compiled into tables by taxonomic class. Details on the methods used and duration of study were also compiled.

2.3.1.3 Determination of Rare, Threatened, and Endemic Species (RTEs)

For all documented species, we searched the IUCN, (2021) for assessment information and noted species considered Vulnerable or more threatened (Vu+). We also searched CITES, (2021) Appdendix I, II, and III as well as species lists of the Wildlife Protection Ordinance of Sarawak (Forestry Department Sarawak, 1998). Whether or not a species was endemic was determined by: 1) research papers, 2) bird and mammal field guides (Phillipps and Phillipps, 2014; Myers, 2016; Phillips and Phillipps, 2016), 4) and descriptions in the IUCN, (2021).

2.3.2 Species Richness, Detection Rates, and Change Over Time

For point count and bird banding datasets, we compared daily species detection rates among the four sites using a Kruskall-Wallis Chi-squared test in R Core Team, (2021). We used Dufrene and Legendre, 1997) Indicator Species approach in PC-Ord 7.0 (McCune and Mefford, 2018) to determine if certain species indicated three different habitat types: peat swamp forest (BPBNP), kerangas forest (BN) and lowland dipterocarp forest (BMWC). To assess change over time at BPNP, we conducted Non-Metric Multidimension Scaling analysis (NMS) with year of survey (2006, 2019) as the grouping variable. A Multi-response Permutation Procedure (MRPP) was used to test the H_0 = no difference in community structure between survey years. We also compared daily species richness detection rates by method

TABLE 1 | Species analyzed for presence in dawn chorus and autonomous recordings using pattern matching analysis.

English name	Latin name	IUCN category
Crested Fireback	*Lophura ignita*	Vu
Bornean Peacock-pheasant	*Polyplectron schleiermacheri*	En, BE
Bulwer’s Pheasant	*Lophura bulweri*	Vu, BE
Great Argus	*Argusianus argus*	Vu
Wallace’s Hawk-eagle	*Nisaetus nanus*	Vu
Large Green Pigeon	*Treron capellei*	Vu
Grey Imperial Pigeon	*Ducula pickeringii*	Vu
Short-toed Coucal	*Centropus rectunguis*	Vu
Bornean Ground-cuckoo	*Carpococcyx radiceus*	NT, BE
Bonaparte’s Nightjar	*Caprimulgus concretus*	Vu
White-crowned Hornbill	*Berenicornis comatus*	En
Rhinoceros Hornbill	*Buceros rhinoceros*	Vu
Helmeted Hornbill	*Rhinoplax vigil*	CE
Black Hornbill	*Anthracoceros malayanus*	Vu
Wreathed Hornbill	*Rhyticeros undulatus*	Vu
Wrinkled Hornbill	*Rhabdotorrhinus corrugatus*	En
Great Slaty Woodpecker	*Mulleripicus pulverulentus*	Vu
Long-tailed Parakeet	*Belocercus longicaudus*	Vu
Blue-headed Pitta	*Hydrornis baudii*	Vu, BE
Bornean Bristlehead	*Pityriasis gymnocephala*	NT, BE
Straw-headed Bulbul	*Pycnonotus zeylanicus*	CE
Hook-billed Bulbul	*Setornis criniger*	Vu
Bornean Wren-babbler	*Ptilocichla leucogrammica*	Vu, BE
Large-billed Blue-flycatcher	*Cyornis caerulatus*	Vu
Chestnut-capped Thrush	*Geokichla interpres*	En
Greater Green Leafbird	*Chloropsis sonnerati*	En

(point counts or bird banding) with Kruskall-Wallis Chi-squared test and Indicator Species Analysis.

2.3.3 Detection of Rare, Threatened, and Endemic Species

The term "rare" can refer to species with restricted ranges, low population density, patchy occupancy, or skittish and elusive behavior (Gaston, 1994; Cerqueira et al., 2013). For those reasons, we did not consider degree of rarity as a criterion for assigning a species as an RTE. However, we did look at relative abundance of species determined to be Vulnerable or above by the IUCN in the point count dataset. We generated rank-abundance charts in excel to look at patterns of relative abundance and detection per km of survey transect. This allowed us to see patterns of abundance in species with significant threats to their persistence.

We established a list of RTE's that met the following criteria: a species 1) was listed as Near-threatened, Vulnerable, or Endangered by the IUCN, (2021), 2) was likely to be detected using the surveys methods (e.g., most nocturnal birds were unlikely to be detected by our methods), and 3) provided high-quality sound files that could be used for pattern-matching analysis (see below). We ended up with a list of 26 species, of which six were Bornean Endemics and 24 were considered Vulnerable or Endangered by the IUCN (**Table 1**).

Using templates of bird vocalization recordings, we used a pattern-matching analysis in Arbimon RFCx (2021) (https://arbimon.rfcx.org/) to search for RTE's in the recordings from dawn chorus surveys and the autonomous recorders for each study site. Matches were validated by ARS, and the date of first detection for each sample unit was determined in matches validated as "present". Date of first detection was also documented for all RTE's detected in point count and bird banding surveys.

We compared daily detection rates of RTEs using Kruskall-Wallis Chi-squared analysis in R Core Team, (2021). We analyzed the relationships between spatial coverage and survey duration for the methods on daily RTE detections using ordinary least squares regression with per-sample survey area and duration as variables and interaction terms. We used a model selection approach with backward and forward stepwise selection and Akaike Information Criterion (AIC) to determine the model of best fit.

A table summarizing method type, data produced by each method, gear costs, people needed (and skills), and post-survey processing time was compiled to compare costs and benefits of each method. Gear costs were determined from gear lists that the survey team used each field season (and cost of new equivalent equipment in 2021, in USD). People required for each method was determined from field survey data and was determined to be the optimal number to complete the survey during the intended timeframe. Post-processing time was determined from timesheet accounting of each survey method. We then ranked each method as it compared to the others in terms of costs (equipment, person time, relative training time) and 'benefits' in the form of data resolution, spatial coverage, and long-term research potential of the datasets (with datasets that produce a large amount of acoustic or image data having higher potential compared to datasets with a smaller amount of digital image or sound data).

2.3.4 Comparison to successful HCV applications.

To compare the results from SPF to successful HCV applications, we searched for summary reports of successful HCV applications on the island of Borneo using the HCV Network's Report search function (https://hcvnetwork.org/find-a-report/?acs-action=advanced-search) and searching for summary reports from Malaysia, Indonesia, and Brunei. Reports were then filtered to include only those from Sabah and Sarawak (Malaysia) and Kalimantan (Indonesia). Each report was read and we compiled information on; 1) the duration of field work focused on HCV-1 Biodiversity, 2) the type and number of species documented as well as the IUCN threat status of documented species.

3 RESULTS

3.1 Biodiversity Inventories

Combined survey efforts resulted in the detection of 216 bird species across the four sites (**Supplementary Table S2**). Fifteen species were listed as Vulnerable, three as Endangered and one (*Rhinoplax vigil*) as Critically Endangered. This represents 66% of all IUCN-listed bird species (Vulnerable+) occurring in lowland forests in Borneo. Twenty-one species were CITES-listed. Forty-five species were listed as Protected in Sarawak and 15 as Totally Protected. New species were added to the area list in six of the eight field seasons of this research, with the most recent addition being Bonaparte's Nightjar, considered Vulnerable by the IUCN in September 2019. Point counts, dawn chorus recordings, bird banding, and autonomous recorders detected 13 of the 19 IUCN-listed (Vu +) species in the study, with the remaining six species either observed incidentally by the survey team (*Rhinoplax vigil, Lophura ignita, and Alophoixus tephrogenys*) or detected by the other survey teams (*Nisaetus nanus, Pitta nympha, and Ptilocichla leucogrammica*).

For non-avian taxa, a literature review using the described search terms resulted in a list of 83 publications that included ecological or biodiversity research focused on the study sites described in this paper. Of those publications 60 included species-level information with 18 publications documenting flora and 42 documenting fauna. When two or more publications provided the same species information, the earliest published paper was selected as the reference. Ultimately, we referenced 25 publications presenting specific data documenting presence of non-bird animal taxa at the sites of interest. The animal classes with presence information were: Actinopterigii, Amphibia, Reptilia, Mammalia, Insecta (Odonata), Malacostraca, and Gastropoda (**Supplementary Tables S3–S8**).

The resulting species lists documented 205 non-bird species, 36 of which were listed as Vu+. A number of new species were described in these papers including: one new species of fish (Hui and Lim, 2007), one new species of frog (Inger et al., 2006), eight new species of Odonate including one new Genus (Dow and Orr, 2012; Dow, 2013; Dow, 2020), and two new species of crab (Grinang and Ng, 2021; Ng, 2021). A new distributional record was also documented for a rare bat species (Rahman et al., 2010). Effort across these studies ranged from relatively constant and continuous for mammals (camera trapping, constant effort over a period of 2 years, and continuing currently) to shorter in duration (gastropods, fishes; see **Supplementary Table S3**). Mammals and Odonates were sampled most intensively out of these groups, and the numbers of IUCN Vu + species detected represented 66% (mammals) and 27% of the possible species.

3.2 Species Richness, Detection Rates, and Change Over Time

Survey efforts that focused on total bird species richness and abundance comprised 32, 1-km point count transects, totaling 156 survey hours, and 1724 individual detections of 139 species and 10, 1-ha mist net plots over 30 survey days, 295 survey hours, and 409 captures of 67 species. Among-site variation in daily species richness did not differ significantly, but indicator species analysis identified 17 species that were significant indicators of lowland dipterocarp forest (found in BMWC), two indicators of kerangas forest (BN), and three indicator species of peat swamp forest (BPNP; **Table 2**). Community analysis (NMS) and comparison (MRPP) of BPNP a from 2006 to 2019 revealed a significant difference in community structure between the two time periods and species accumulation curves show higher species accumulation in 2019 (**Figures 2**, **3**).

Analysis of detection rates of all species from point counts and bird banding resulted in a significant difference in daily detection rate between the two methods with point counts having significantly higher daily species detections than banding (Kruskal–Wallis chi-squared = 47.676, df = 1, p-value < 0.00001; **Figure 4**), with 82 indicators of point counts and three indicators of bird banding (**Table 3**).

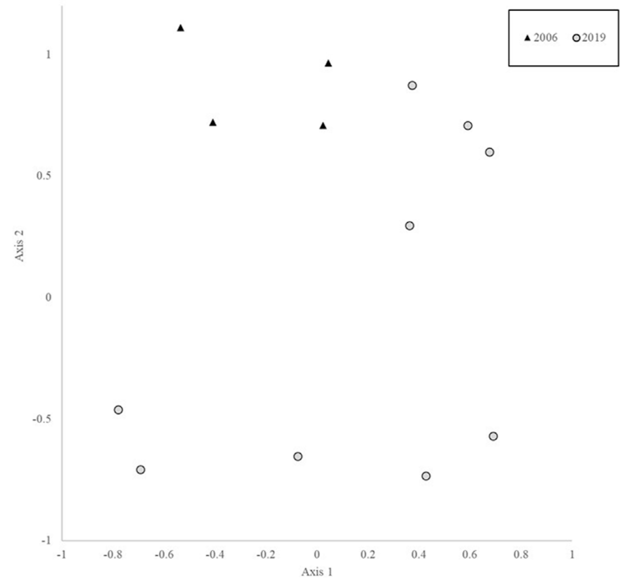

FIGURE 2 | NMS ordination of point count transect data at Binyo (2006 and 2019). MRPP value T = 2.9, A = 0.5, and p = 0.01.

TABLE 2 | Indicator species for forest type.

Habitat	Species	P
Lowland Dipterocarp Forest	*Anthracoceros malayanus*	<0.001
	Psilopogon duvaucelii	0.036
	Eurylaimus ochromalus	0.003
	Calyptomena viridus	0.016
	Terpsiphone affinis	0.002
	Pycnonotus erythrophthalmos	0.001
	Tricholestes criniger	0.018
	Stachyris maculata	0.040
	Stachyris rufifrons	0.029
	Macronous borneensis	0.013
	Malacocincla malacensis	0.003
	Malacopteron affine	0.049
	Malacopteron magnirostre	0.010
	Trichastoma bicolor	0.001
	Irena puella	0.035
	Copsychus malabaricus	0.006
	Diceaum trigonostigma	0.026
Kerangas (Heath) Forest	*Malacopteron albogulare*	0.048
	Cyornis umbratilis	<0.001
Peat Swamp Forest	*Chrysophlegma miniaceum*	0.048
	Aegithina viridissima	0.019
	Diceaum cruentatum	0.029

3.2.1 Detection of Rare, Threatened, and Endemic Species

Survey efforts focused on detection of RTEs included 40 days and 42 h of dawn chorus surveys and 10 independent autonomous recorder deployments accumulating 600 h of sampling time. We conducted 448 pattern matching analyses, resulting in 108,212 matches of templates and 46 analyses (with a total of 2,855 matches) resulting in at least one confirmed detection among the matches. We confirmed 431 matches as actual detections of 11 of the 26 focal species.

Daily RTE detection varied significantly among methods with point counts having higher daily detections of RTEs compared to bird banding (Kruskal Wallis chi-squared = 16.9, DF = 3, p = 0.0007), and both point counts and ARU's having higher daily accumulation of RTEs (Kruskal Wallis chi-squared = 17.9, DF = 3, p = 0.0005; **Figures 5A,B**).

Regression models of the response of RTE detection to sampling duration, sampling area, and interaction term resulted in a significant model (F = 3.81, DF = 3, 38, p = 0.02, r^2 adj = 0.2138; **Table 4**). Backward and forward stepwise model selection resulted in congruous results and two nearly equivocal models with the best explanatory power: 1) the full model that accounted for sampling area and sampling duration plus interaction between the two parameters, and 2) a model including just sample duration (**Table 4**). Daily RTE detection rates increased with sample area significantly (F = 54.75, DF = 1, 30, p < 0.0001, adj r^2 = 0.6342; **Figure 6**). Rank abundance plots from point count data plotted eight of the 13 RTE species detected at the study sites. The RTE species with the highest abundance

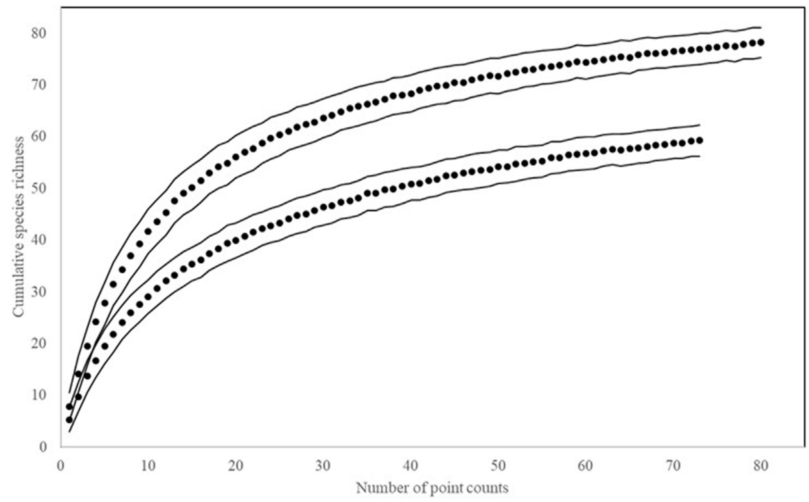

FIGURE 3 | Species accumulation plots for point count surveys at Binyo-Penyilam (2006 and 2019). Lines represent standard deviation. Per point richness (S) and diversity (H′) ± (s) 2006: S = 5.6 ± 2.6, H' = 1.5 ± 0.5; 2019: S = 7.7 ± 2.7, H' = 1.9 ± 0.4.

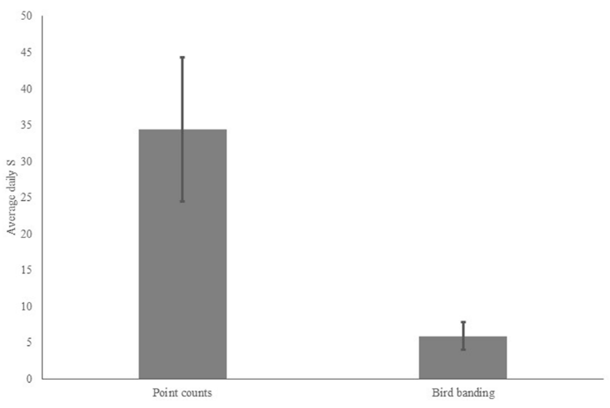

FIGURE 4 | Average daily species richness of point counts versus bird banding (Kruskal–Wallis chi-squared = 47.676, df = 1, *p*-value < 0.00001).

was Black Hornbill (*Anthracoceros malayanus;* 1.4% of total abundance) and that with the least was Wrinkled Hornbill (*Rhabdotorrhinus corrugatus*; 0.03% total abundance; **Figures 7A,B**).

3.2.2 Costs and Benefits of Methods

Dawn chorus surveys were the least costly method in terms of equipment, and equipment-life is 5–10 years if protected from moisture and shock (**Table 5**). The method requires two observers, at least one highly-skilled in bird vocalization. Preparation time is about 15 min per survey, and processing time after survey completion is about 3 hours. Point count surveys require similar gear as dawn chorus surveys, but the addition of a laser rangefinder is needed to make accurate distance estimates (they can also help with some habitat data collection). Point counts also require time investment from preparation to post-survey processing. Field time for point counts is longer due to collection of habitat data. Over a 10-days survey, 7–10 days of data can be collected for both point counts and dawn chorus surveys. Autonomous recording units are more expensive to deploy, and they tend to have a shorter lifespan due to their exposure to the elements. Person and time investment in the field is lower compared to other methods, and the skills required for field deployment are more technical than skill in sound identification. Sample time can be maximized by long deployments and continuous or nearly continuous recording. Post-survey processing time for ARUs is substantial, with an average of 4 days for every 10 days of deployment (recording 4 hours per day). Bird banding is the most expensive of the four methods in terms of initial gear costs, but much of the gear can have a long lifespan with proper care. The method also requires substantial time investment in preparation and more people are needed for field work over a longer (10-h) field day. Post-survey processing time is relatively low for bird banding, and the amount of data gathered per individual is higher than other methods.

3.3 Comparison to Successful HCV Applications

We reviewed the summary statements from 33 approved HCV applications from Sabah, Sarawak, and Kalimantan. All of these statements provided information regarding the timeline of the process including the number of field days focused on assessment presence of RTE species in the proposed area. Twenty-six of the statements also included information on species that were documented to be presented including species listed as Vulnerable or more threatened by the IUCN. Among those applications, the average number of days of field work was 8.9 (*sd* ± 5.7). The species documented averaged 10 (*sd* ± 6.2; **Supplementary Table S9**). In many cases, the focus was on highly endangered mammal species such as Bornean Orangutan or Sunda Pangolin (**Supplementary Table S9**). By comparison, the

TABLE 3 | Indicator species for field method.

Species	Method	P
Aegithina tiphia	Point count	0.019
Aegithina viridissima		<0.001
Aethopyga siparaja		0.031
Alcedi meninting		0.001
Anthracoceros malaynus		<0.001
Anthreptes malacensis		0.005
Anthreptes rhodolaema		0.051
Arachnothera crassirostris		0.003
Arachnothera flavigaster		0.053
Arachnothera longirostra		<0.001
Argusianus argus		0.015
Buceros rhinoceros		0.049
Cacomantis merulinus		0.001
Cacomantis sonneratii		<0.001
Calorhamphus fuliginosus		<0.001
Centropus sinensis		<0.001
Chalcoparia singalensis		<0.001
Chalcophaps indica		0.001
Chloropsis cyanopogon		0.048
Chrysophlegma miniaceum		0.001
Coracina fimbriata		0.016
Covus enca		0.003
Corydon sumatranus		0.019
Diccaeum cruentatum		<0.001
Dicaeum trigonostigma		<0.001
Dicruris paradiseus		0.006
Dryocopus javensis		<0.001
Ducula aena		0.002
Eurylaimus javanicus		0.017
Eurylaimus ochromalus		<0.001
Gracula religiosa		<0.001
Harpactes diardii		0.050
Harpactes kasumba		<0.001
Hemicircus concretus		0.016
Hemirocne longipennis		0.017
Hemipus hirundinaceus		0.005
Hypogramma hypogrammicum		<0.001
Hypothymis azurea		<0.001
Irena puella		<0.001
Leptocoma brasiliana		0.016
Loriculus galgulus		<0.001
Macronous ptilosus		0.001
Malacopteron affine		<0.001
Malacocincla malacensis		<0.001
Malacopteron magnum		0.010
Micropternus brachyurus		0.045
Macronous borneensis		<0.001
Nyctiornis amictus		0.002
Oriolus xanthonotus		<0.001
Orthotomus atrogularis		<0.001
Orthotomus ruficeps		0.001
Orthotomus sericeus		<0.001
Pellorneum captistratum		<0.001
Trichastoma rostratum		<0.001
Phaenicophaeus curvirostra		<0.001
Picus puniceus		0.052
Pityriasis gymnocephala		0.002
Platysmurus aterrimus		0.002
Prinia flaviventris		<0.001
Prionochilus xanthopygius		<0.001
Psilopogon duvaucelii		<0.001
Psilopogon chrysopogon		0.016
Psilopogon henricii		0.049
Psilopogon rafflesii		0.000
Psittacula longicauda		0.016
Pcynonotus atriceps		<0.001
Pycnonotus brunneus		<0.001
Pycnonotus erythrophthalmos		<0.001
Pycnonotus simplex		<0.001
Rhaphidura leucopygialis		0.001
Rhinortha chlorophaea		<0.001
Rhipidura javanica		<0.001
Spilornis cheela		<0.001
Stachyris erythroptera		<0.001
Stachyris maculata		<0.001
Stachyris nigricollis		0.002
Stachyris rufifrons		0.002
Terpsiphone affinis		0.012
Treron olax		0.017
Tricholestes criniger		0.039
Trichostoma bicolor		0.041
Trichostoma rostratum		0.001
Alophoixus phaeocephalus	Bird banding	0.026
Ceyx erithaca (rufidorsa)		0.014
Malacopteron albogulare		0.049

(Continued in next column)

TABLE 3 | (*Continued*) Indicator species for field method.

avian survey work (not including rapid assessments by other teams) totaled 71 calendar days by teams in the field and an additional 30 days of autonomous recording. Average field visit length at the sites was 14.2 (*sd* ± 10.4). On most research trips, teams also conducted surveys in *Acacia mangium* plantation which, if included, averaged 24 (*sd* ± 9.4).

4 DISCUSSION

4.1 Biodiversity Metrics for HCV

The Sarawak Planted Forest (SPF) sites surveyed in this study were embedded within a landscape that, although heavily disturbed and largely covered by plantation, contained substantial areas of native forest, represented high-priority ecosystems, and were in close proximity to other protected areas (**Figure 1**). Original bird data from this study combined with research focused on other taxa resulted in a compiled list of 420 animal species, of which 55 are listed as Vulnerable or more threatened by the IUCN. Additionally, 12 species new to science have been discovered within the area, with some possibly endemic to sites within SPF. This assessment did not evaluate information from floristic studies, which also have described new species (e.g., Kiew and Sang, 2009; Kiew and Sang, 2013; Lin C.-W. et al., 2014; Lin W. et al., 2014; Lin et al., 2017).

The purpose of biodiversity criteria in the HCV process is to document evidence of RTEs rather than generate a comprehensive biodiversity inventories. The field work element of HCV applications from other locations in Borneo was relatively brief and combined field survey methods with interviews of locals. In most cases the focus of these efforts was on conspicuous and charismatic species, such as orangutans and hornbills, although some applications provided more comprehensive datasets. Although there is no specific guidance about whether some RTEs are more indicative of HCV compared to others, the most frequently mentioned species

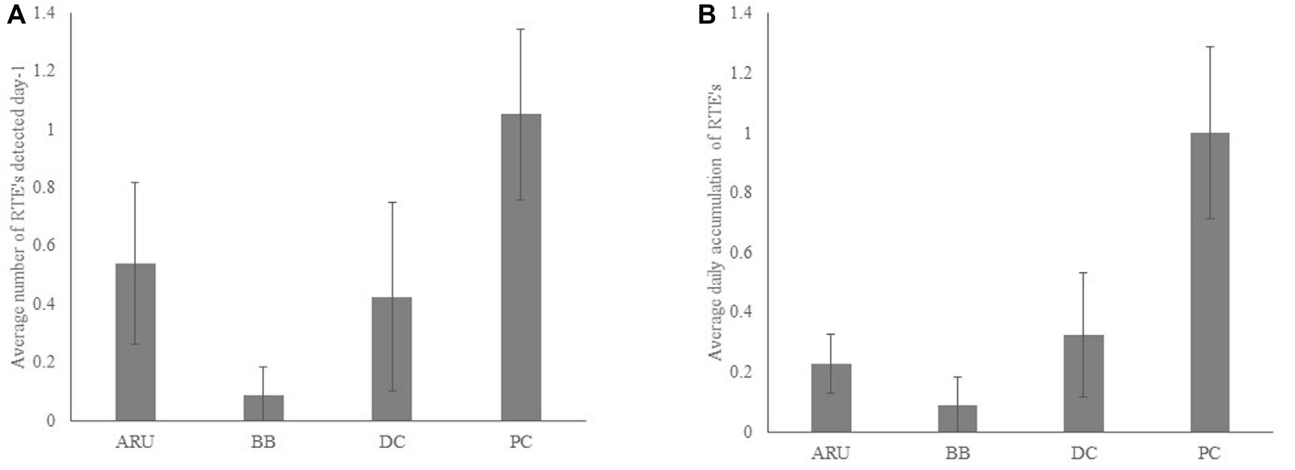

FIGURE 5 | Average (per day) detection of **(A)**. RTE's (Kruskal Wallis chi-squared = 16.9, DF = 3, p = 0.0007); and **(B)** number of total RTE's by method (Kruskal Wallis chi-squared = 17.9, DF = 3, p = 0.0005). Error bars represent 95% confidence intervals. Pairwise comparisons of methods indicated significant differences between bird banding and point counts **(A and B)** and between bird banding and autonomous recording units **(B)**.

TABLE 4 | Regression table a. and model selection and b. for RTE ~ Sampling days + Area (ha) + Sampling days x Area (ha). F = 3.81, DF = 3, 38, p = 0.02, r^2 adj = 0.2138.

A

	Estimate	Std.	t	*p*
(Intercept)	2.30291	0.69142	3.331	0.00244
Sampling days	–0.03031	0.05372	–0.564	0.57709
Area (ha)	–0.08208	0.05065	–1.621	0.11632
Sampling days x Area (ha)	0.0324	0.01331	2.434	0.02157

B

Model	AIC
Full Model	**20.82**
Sampling days + Area (ha)	23.49
Area (ha)	21.35
Sampling days	**20.34**

Values in bold represent equivalently low AIC values.

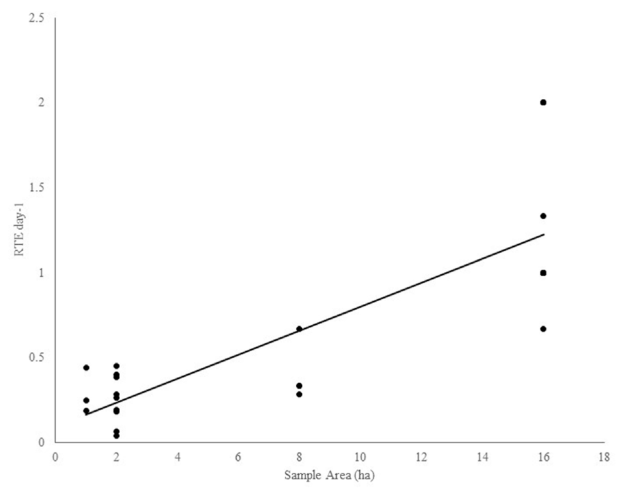

FIGURE 6 | RTE's per day across sample area.

in the assessments included: *Pongo pygmaeus*, *Helarctos malayanus*, *Sus barbatus*, and *Manis javanica*. Hornbills (*Rhinoplax vigil, Buceros rhinoceros,* and *Anthracaceros malayanus*) were also mentioned in a number of applications. Data from SPF documents presence of the most frequently-mentioned species with the exception of Bornean Orangutan as the area is outside of the species' range.

4.2 Best Practices for Field Assessment and Monitoring

We evaluated the efficacy and costs of four different survey methods commonly used for biodiversity inventory of birds. Our findings largely agree with other comparative research. Each method requires different skill sets and has different biases in detection. Studies that compare methods usually recommended that multiple methods be used when the goal is comprehensive biodiversity inventories (Martin et al., 2010; Cellis-Murello et al., 2012; Leach et al., 2016; Wheeldon et al., 2019; Darras et al., 2019). Many HCV applications document a pre-assessment phase in addition a full assessment, and there are long-term monitoring requirements (Furumo et al., 2019). We recommend specific types of survey methods for assessment and monitoring.

4.2.1 Autonomous Recording Units–An Effective Tool for Assessment and Monitoring

Programmable autonomous recorders are becoming increasingly affordable, durable, and lightweight, making the deployment of

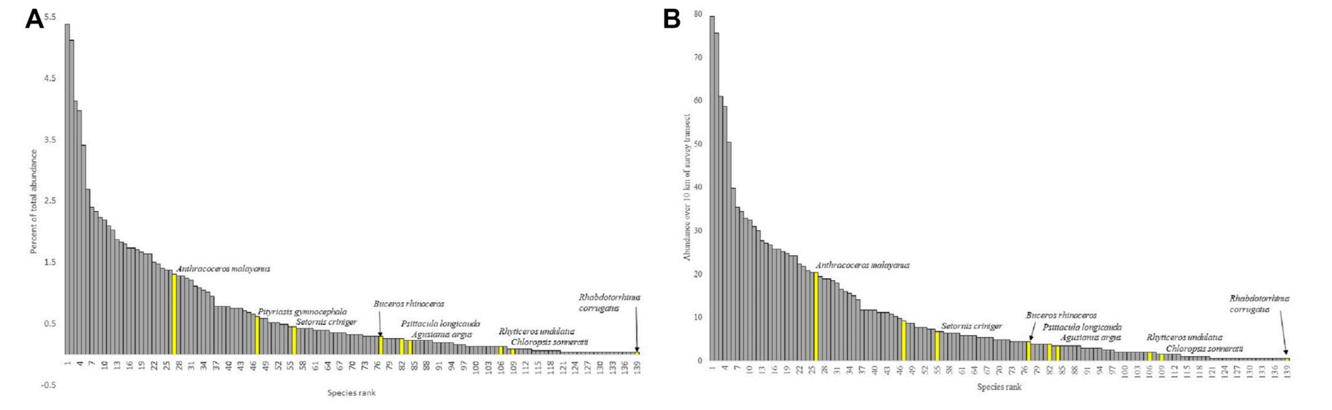

FIGURE 7 | Rank abundance **(A)** percentage of total abundance by species rank (from most abundant to least) and **(B)** as average number of individuals per 1 km of survey transect. Yellow highlighted bars represent the eight species listed as Vulnerable or more threatened by the IUCN that were detected on during point counts.

numerous ARUs across a large landscape feasible. Additionally, the archive of raw sound data produced by ARU's can serve as an important resource for subsequent research projects (Digby et al., 2013; Krause and Farina, 2016). Some drawbacks of this method are: 1) non-vocal species are not detected, 2) large amounts of data to search for focal species, 3) demands of large datasets in terms of computing power and media storage, 3) maintenance, repair, and replacement costs. However, many of these drawbacks will become less problematic as technology advances (Digby et al., 2013; Darras et al., 2018; Metcalf et al., 2021).

In our study, autonomous recording units had comparable detection rates of RTE's compared to dawn chorus surveys, but they surpassed DC surveys when geographic coverage was larger, and deployment time was longer (weeks/month). This finding is consistent with other studies that recommend ARUs be deployed across a large geographic area and sample strategically to increase probability of detection (Digby et al., 2013; Metcalf et al., 2021). ARUs are becoming more affordable with higher recording capacity and longer battery life (**Table 5**) but will likely require long-term costs for maintenance, data processing, and replacement. They lower in-person field-time requirements and the datasets generated by ARUs are large. The amount of post-sampling processing time can be substantial (**Table 5**) but the field of pattern matching analysis and song recognition is rapidly evolving, making speed of identification most likely quicker in the future. Ultimately, ARU's are likely to be the most useful method for detecting and documenting rare, nomadic, and elusive species, which constitute a substantial proportion of RTEs. We recommend that ARUs be deployed for full assessments and for long-term monitoring efforts. Ideally, sampling strategies would include high probability detection times (such as the dawn chorus) as well as strategic sampling during other time periods across a large area (Metcalf et al., 2021).

4.2.2 Point Count Surveys for Assessment and Periodic Monitoring

Point count surveys had the highest rates of overall species detection as well as the highest rates of detection for RTE's compared to the other survey methods examined in this study (**Figures 4**; **Figures 5A,B**). This is likely due in part to the relatively large area covered (**Figure 6**; **Table 5**). This finding is consistent with research comparing point counts and ARUs when the geographic scale covered by each method is similar (Klingbeil and Willig 2015; Leach et al., 2016). Point counts also produce data on the entire community of birds (species and abundances) and can be readily analyzed with habitat data collected at each survey point to determine key habitat features important in predicting community composition (Styring et al., 2011). They also have the advantage of being relatively low-cost in terms of equipment, and they are quick to process once the field survey is complete (Klingbeil and Willig, 2015). However, they require a relatively intensive survey effort and skilled observers with the ability to detect most of the bird species by sight and sound. If less-skilled observers conduct the surveys, they may make recordings of their point counts including verbal notes about the appearance or sound of unidentified birds. The notes and recorded surveys can be shared with a more experienced collaborator who can likely make determinations from many of the unidentified observations using the verbal notes and cues in the recordings.

We recommend that point counts and associated habitat surveys be part of both full assessments and long-term monitoring efforts (with point count resurveys occurring every 5–10 years). A robustly-sampled dataset can serve as a baseline of community-level information. Distance sampling is a particularly useful approach because circumstances that may influence detection (observer differences, habitat change, etc.) can be identified and accounted for among survey years, habitats, and observers (Buckland et al., 2015).

Although point counts had the highest detection rates for overall species and RTEs, they did not detect all of the RTEs present at the study sites. RTEs are generally uncommon to rare (**Figures 7A,B**) and tend to require larger survey areas over longer time periods. Nine of the 13 RTE species detected by the survey methods used in this study, were detected during point count surveys. The remaining four species were detected by other methods: *Berenicornis comatus* (ARU, DC), *Caprimulgus concretus* (ARU), *Centropus rectunguis* (ARU), and

TABLE 5 | Methods details. A. Equipment lists and costs do not include basic field supplies needed for ornithological work such as binoculars and field notebooks/datasheets, etc. Equipment lifespan is an estimate based on the lifespan of gear used in the surveys described in this study. Gear was subjected to hot and humid conditions during field deployment, then cleaned and stored in secure, dry storage. Post processing estimates are based on a hypothetical 10-days survey trip (not including travel time to/from survey location). This processing time would increase for autonomous recorders if more than 10 recorders are deployed. All prices are in USD. B. Rank of methods. Rankings are from * to **** with * representing low and **** representing high for the variables listed.

A

Method	Data Generated	Coverage (ha)	Equipment	Equipment cost	Equipment lifespan (y)	People needed	Skills	Sample days/ 10-days survey	Time sampled/ day (hrs)	Prep time prior to first sample	Post processing time (hrs)	Number post-survey days to completion	Recommended for
Autonomous recording	presence	1.8/unit	Autonomous recorder, waterproof case, batteries, SD cards	$2,000	2–7	2	programming and maintenance, species identification by sound	10	4–24	5	30	4	detecting RTE's over a time period >10 days
Dawn chorus surveys	presence	7.5/ sample day	Digital recorder, microphone, batteries, SD cards	$410	5–10	2	Expert bird identification by site and sound	7–10	1	0.25	3	0	pilot study/pre-assessment
Point counts	presence, abundance, habitat	15/ sample day	Digital recorder, microphone, batteries, SD cards, laser rangefinders	$720	5–10	2	Expert bird identification by site and sound	7–10	6 (point count + hab survey)	0.25	3	0	full assessment + monitoring
Bird banding	presence, abundance, age, sex, ondition, molt	1 ha/plot	Mist nets, banding supplies, bands	$3,000	5–10	3	At least one experienced bird bander	5–7	10	6	2	1	full assessment + monitoring

B

Method	Data resolution	Potential area coverage	Equipment affordability	Person effort	Post data collection potential	Potential to deploy with minimal training	Ability to develop research projects
Autonomous recorders	*	****	**	**	***	****	***
Bird Banding	****	*	*	*	***	*	****
Dawn chorus surveys	**	***	****	***	*	**	*
Point count surveys	***	***	***	***	*	***	**

Mulleripicus pulverulentus (ARU, DC). Three other species were detected by point counts at some locations, but with other methods at other locations: *Buceros rhinoceros* (BS, ARU; BN, DC), *Psittacula longicauda* (BN, DC), and *Setornis criniger* (BN, BB). Assessment approaches using more than one survey method have better chances of detecting more RTEs. These findings are consistent with other research suggesting that acoustic methods and point counts be used together (Digby et al., 2013; Darras et al., 2017; Wheeldon et al., 2019).

4.2.3 Bird Banding as the Cornerstone of Long-Term Monitoring

Mist-netting and bird banding provides the most detailed information on the population structure, demography, and seasonal patterns of avian cycles for understory forest birds (DeSante et al., 2001; Dunn and Ralph, 2004). However, it is not the ideal method for conducting comprehensive biodiversity inventories due to the limited spatial scale of capture compared to detections from other survey methods (Remsen and Good, 1996).

Bird banding provided the lowest daily richness rates in this study and required high sampling and pre-sampling effort. It is the most expensive of the four methods at the outset, but long-term equipment costs are relatively low with proper equipment care. Though banding has low detection rates for richness, it is the best way to document secretive and inconspicuous understory species that do not vocalize frequently including migratory species (Martin et al., 2010). During the study period, we banded migratory species not detected by other methods such as *Larvivora cyane*, a Least Concern species (IUCN Redlist) that is thought to be declining range wide. We also banded resident species that were rare, patchy, not-well known, and not detected with other methods including: *Harpactes orrophaeus,* an uncommon trogon with unknown breeding season and behavior, and *Malacopteron albogulare*–a rare species restricted to nutrient-poor forests such as kerangas and peatswamp. *M. albogulare* is a Near-threatened species with an unknown breeding biology. We netted, numerous individuals in breeding condition at BN over a 3-week period in March-April 2018.

Even among species that are readily detected during point counts, many are still relatively unknown in terms of breeding biology and population demography including species that are likely to be captured in nets such as *Setornis criniger,* and *Copsychus pyrropygus*. Because of the wealth of critical data that can be collected on birds in the hand, we recommend that bird banding an essential element of long-term monitoring efforts. Banding efforts could occur during wetter and drier months each year and would provide vital information on the seasonal cycles and population dynamics of many understudied species. Long-term monitoring efforts that included bird banding could result in valuable information on the vital rates such as productivity, survivorship, sex ratio, and dispersal. Such information could be used to more effectively predict population change and manage sensitive populations (Francis and Wells, 2003; Dunn and Ralph, 2004).

4.2.4 Dawn Chorus Recordings–When the Opportunity Arises

The use of recordings of the dawn chorus to rapidly inventory avifaunal diversity was an early and effective use of acoustic tools in the study of biodiversity (Parker, 1991). These surveys need to be conducted by people who are highly skilled at bird vocalization identification (Parker was reported to know the vocalizations of over 4,000 species). In this study, dawn chorus recordings had lower daily detection rates compared to point counts. This result contrasts with the findings of Parker, (1991). One likely cause for the difference is the amount of transect length covered each day which averaged 2 km for Parker and <1 km for our survey teams. To cover more transect during a 1-h dawn chorus survey, observers must be moving along trails or roads that are relatively clear and easy to navigate quickly. Observers must also be selective in recording species with the goals of documenting new species. The quality and power of the direction microphone used could also have an impact, but is likely not large in the case of this study. Dawn chorus recordings likely have the potential to detect as many or more species per day as point counts given those considerations.

There are challenges with the data from dawn chorus surveys in terms of comparability across sites and between time periods (Herzog et al., 2002). This is because the goal of the recordist is to document as many species as possible a dawn, which may result in changing effort across a survey on any given day. Additionally, advancement of passive methods has begun to replace this type of survey method due, in part, to the difficulty in finding the expertise needed (Krause and Farina, 2016). However, dawn chorus surveys could be undertaken opportunistically. They are useful in siting locations for future point count transects and bird banding plots. The recordings serve as a reference archive of the environment including presence of other vocal species (such as vocal mammals and frogs). They are also low cost in terms of equipment and post-survey processing time is relatively short. Although dawn chorus surveys conducted by less skilled observers do not document as many species (due to the less targeted nature of the recording), they are a very useful learning tool for observers and technicians who wish to improve their bird identification skills.

4.3 Implementation

Because organizations are often limited in time and resources available for assessment and monitoring, methods that are low cost, low effort, and that rapidly result in the needed data will be implemented more readily than methods that require higher investment in resources (Greenwood, 2007; Danielsen et al., 2008; Neate-Clegg et al., 2020). In practice, the issues of time, effort, expertise, and data quality are often at tension with one another with relatively low-cost approaches requiring more intensive quality control efforts (Wiggins et al., 2013).

4.3.1 Assembling Survey and Monitoring Teams: Partnerships and Knowledge Building

A challenging element of conducting assessments or implementing long-term monitoring is finding and assembling teams of people with the applicable knowledge and skills in the required timeframe. In much of North America and Europe, this challenge is met by hiring skilled applicants for short time periods, relying on volunteer participation, or hiring environmental consultants. There are strengths and weaknesses associated with each option in terms of cost, data integrity, and transparency. In countries that may not have a large volunteer base or many individuals with the relevant knowledge base, partnerships with NGOs or research institutions from other countries may be an effective approach (Danielsen et al., 2008). The expense of these collaborations varies in terms of cost-sharing and time, with collaborations that commit to local knowledge building having the greatest long-term success (Greenwood 2007; Danielsen et al., 2008).

For some methods, such as deployment of ARUs, collaborations could be relatively low cost with experienced partners providing training and troubleshooting to local researcher via remote workshops and tutorials. Data analysis could be collaborative with pattern-matching analysis and species determinations occurring via a shared analysis and data management platform. Other methods, such as point counts, may require in-person survey work and training. This could occur through a combination of workshops focused on species identification and survey methodology. More intensive knowledge-development, like that involved with bird banding, may take the form of intensive training workshops and development of skilled in-country banders who can then train others (Greenwood, 2007).

A large amount of what we know about bird populations in North America and parts of Europe comes from large community-science programs such as MAPS (Monitoring Avian Productivity and Survivorship, DeSante, 2001), Breeding Bird Survey (Ralph et al., 1995) and eBird (Aceves-Bueno et al., 2017). These efforts work when there are large numbers of people who watch birds recreationally. The popularity of birdwatching is growing around the world (e.g. Walther and White, 2018). Cultivating the advancement of a community-science culture among birdwatching communities will help broaden the knowledge base among local conservation-interested communities that can be used for long-term monitoring. Ensuring High Conservation Values over the long-term relies heavily on knowing the population patterns and processes of sensitive species in an area, which, in turn, relies on having a knowledgeable workforce and community base. Developing this knowledge base is a high priority to the long-term viability of the HCV process.

4.3.2 Collection, Sharing, and Archiving of Sound, Image, and Observational Data and Metadata

Each method described in this study can generate diagnostic evidence of species presence such as a sound recording or photograph, though not every species can be effectively documented in a given survey or with a given method. For example, birds not readily captured in mist nets are difficult to photograph in a way that is diagnostic, not all species vocalize regularly, and some rarely vocalize at all making recording difficult. But positive evidence in the form of image or sound is a useful resource in documenting species presence as well as variation over space and time. We recommend that, whenever possible, assessment and monitoring methods have accompanying sound or image data and that all data and metadata be shared broadly (Wiggins et al., 2013). Point count surveys can be recorded with either a good quality digital recorder and directional microphone, or a smart device set for high-resolution recording. Dawn chorus surveys and autonomous recorders generate large amounts of acoustic evidence, and birds captured and processed during banding activities can be photographed at high resolution and with macro- and microscope lenses to evaluate plumage and molt. Making these media available to the broader scientific and conservation communities improves overall understanding of species distributions, variations, and seasonal patterns (Michener 2015). Many databases are available to professional and community scientists such as iNaturalist (https://www.inaturalist.org/home), xeno-canto (https://www.xeno-canto.org/), eBird (https://ebird.org/home), and gbif (https://www.gbif.org/). Crowdsourced analytical tools such as R and Arbimon (https://arbimon.rfcx.org/) have made it increasingly possible for researchers and interested community members to undertake collaborative research by sharing templates and analysis results (Lahoz-Monfort and MacGrath, 2021). These tools provide a venue for more people to participate in documenting diversity, and we are learning more about species distributions and uses of disturbed habitats as a result. Although there are risks of sharing data, especially for species that are threatened by the illegal wildlife trade, protections are in place in many cases that help prevent this problem (Tullock et al., 2018).

4.4 The Value of Disturbed Landscapes

Five timber plantations and other production landscapes often have limited access, and we therefore, know much less about these areas compared to densely populated areas or public-access protected areas. Research from SPF area provides insight into an industrial production landscape. Over a 15-year period, research teams, produced a relatively comprehensive species list for birds and other taxa. Despite a large amount of effort, new species are being added to the list regularly. This is likely due, in part, to patterns of commonness and rarity in high-diversity ecosystems, and, in part, to transients passing through an area, and in part due to changes in habitat and vegetation relating to larger scale patterns of land use and changes in climate. One likely, but understudied cause is the nomadic nature of many Bornean species and how patterns of large-scale movement and population change are influenced by weather and climate cycles that are longer than 1 year in duration. Species such as *Sus barbatus* and *Helarctos malayanus* undergo long-term patterns of population growth and decline associated with El Nino events (Curran and Leighton, 2000; Wong et al., 2005). Other species such as *Carpococcyx radiceus* are known to follow *Sus barbatus* movements, and others such as *Pityriasis gymnocephala*, *Corydon sumatranus*, are nomadic, but the root causes are still unknown. As these species face multiple population threats, understanding their basic biology is increasingly important to their long-term survival. Sensitive species use disturbed and degraded natural forests (Edwards et al., 2011) as well as novel ecosystems such as exotic timber plantation if the conditions in the environment are suitable (Sheldon et al., 2010; Styring et al., 2011; Stying et al., 2018). It is important that we understand the role of disturbed landscapes including production landscapes to these animals.

DATA AVAILABILITY STATEMENT

The original contributions presented in the study are included in the article/**Supplementary Materials**, further inquiries can be directed to the corresponding author.

ETHICS STATEMENT

The animal study was reviewed and approved by The Evergreen State College Animal Care Committee.

AUTHOR CONTRIBUTIONS

AS was involved in fundraising, research design, field work, data analysis, and manuscript preparation. JU and RR were involved in fundraising, research implementation and priority-setting, and field work for all field seasons. KK was responsible for a large portion of data analysis and paper writing. DF was responsible for field implementation of avian banding and oversaw logistics of many field operations from 2017–19. NM, AJ, and LJ undertook extensive field work and logistics from 2006 to 2019. MT, KDS, KNS, and LD undertook extensive field work from 2017–19 as did MN who also organized datasets and generated much needed maps. LG undertook field work in 2019 and was responsible for all technical aspects of autonomous design and implementation. BN NS, PH, and BT, did extensive field work in 2017, 2018, and 2019 respectively and DJ helped bird logistical efforts in 2006–2007.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fenvs.2022.783794/full#supplementary-material

REFERENCES

Aceves-Bueno, E., Adeleye, A. S., Feraud, M., Huang, Y., Tao, M., Yang, Y., et al. (2017). The Accuracy of Citizen Science Data: a Quantitative Review. *Bull. Ecol. Soc. Am.* 98, 278–290. doi:10.1002/bes2.1336

Arbimon RFCx (2021). Rainforest Connection® Arbimon. Available at: https://arbimon.rfcx.org/(Accessed September 21, 2021).

Areendran, G., Sahana, M., Raj, K., Kumar, R., Sivadas, A., Kumar, A., et al. (2020). A Systematic Review on High Conservation Value Assessment (HCVs): Challenges and Framework for Future Research on Conservation Strategy. *Sci. Total Environ.* 709, 135425. doi:10.1016/j_scitotenv.2019.135425

Bradshaw, C. J., Sodhi, N. S., and Brook, B. W. (2009). Tropical Turmoil: a Biodiversity Tragedy in Progress. *Front. Ecol. Environ.* 7, 79–87. doi:10.1890/070193

Brown, E., Dudley, N., Lindhe, A., Muhtaman, D. R., Stewart, C., and Synnott, T. (2013). Common Guidance for the Identification of High Conservation Values. HCV Resource Network. Available at: https://hcvnetwork.org/wp-content/uploads/2018/03/HCVCommonGuide_English.pdf (Accessed September 21, 2021).

Buckland, S. T., Rexstad, E. A., Marques, T. A., and Oedekoven, C. S. (2015). *Distance Sampling: Methods and Applications*. Switzerland: Springer.

Carlson, K. M., Heilmayr, R., Gibbs, H. K., Noojipady, P., Burns, D. N., Morton, D. C., et al. (2017). Effect of Oil Palm Sustainability Certification on Deforestation and Fire in Indonesia. *Proc. Natl. Acad. Sci. U.S.A.* 115, 121–126. doi:10.1073/pnas.1704728114

Carrasco, L. R., Larrosa, C., Milner-Gulland, E. J., and Edwards, D. P. (2014). A Double-Edged Sword for Tropical Forests. *Science* 346, 38–40. doi:10.1126/science.1256685

Cellis-Murello, A., Deppe, J. L., and Ward, M. P. (2012). Effectiveness and Utility of Acoustic Recordings for Surveying Tropical Birds. *J. Field Ornithol.* 83, 166–179. doi:10.1111/j.1557-9263.2012.00366.x

Cerqueira, M. C., Cohn-Haft, M., Vargas, C. F., Nader, C. E., Andretti, C. B., Costa, T. V. V., et al. (2013). Rare or Elusive? A Test of Expert Knowledge about Rarity of Amazon Forest Birds. *Divers. Distrib.* 19, 710–721. doi:10.1111/ddi.12033

Chen, J., Innes, J. L., and Tikina, A. (2010). Private Cost-Benefits of Voluntary Forest Product Certification. *Int. For. Rev.* 12, 1–12. doi:10.1505/ifor.12.1.1

CITES (2021). Appendices I, II, & III. Available at: https://cites.org/eng/app/appendices.php (Accessed September 21, 2021).

Curran, L. M., and Leighton, M. (2000). Vertebrate Responses to Spatiotemporal Variation in Seed Production of Mast-Fruiting Dipterocarpaceae. *Ecol. Monogr.* 70, 101–128. doi:10.1890/0012-9615(2000)070[0101:vrtsvi]2.0.co;2

Danielsen, F., Burgess, N. D., Balmford, A., Donald, P. F., Funder, M., Jones, J. P., et al. (2008). Local Participation in Natural Resource Monitoring: a Characterization of Approaches. *Conserv. Biol.* 23, 31–42. doi:10.1111/j.1523-1739.2008.01063.x

Darras, K., Batáry, P., Furnas, B. J., Grass, I., Mulyani, Y. A., and Tscharntke, T. (2019). Autonomous Sound Recording Outperforms Human Observation for Sampling Birds: a Systematic Map and User Guide. *Ecol. Appl.* 29, e01954–1265. doi:10.1002/eap.1954

Darras, K., Batáry, P., Furnas, B. J., Celis-Murillo, A., Van Wilgenburg, S. L., Mulyani, Y. A., et al. (2017). Comparing the Sampling Performance of Sound Recorders versus Point Count Bird Surveys: a Meta Analysis. *J. Appl. Ecol.* 55, 2575–2586. doi:10.1111/1365-2664.13229

Darras, K., Furnas, B., Fitriawan, I., Mulyani, Y., and Tscharntke, T. (2018). Estimating Bird Detection Distances in Sound Recordings for Standardizing Detection Ranges and Distance Sampling. *Methods Ecol. Evol.* 9, 1928–1938. doi:10.1111/2041-210x.13031

DeSante, D. F., Burton, K. M., Kaschube, D. R., Velez, P., Froehlich, D., and Albert, S. (2021). *MAPS Manual: 2021 Protocol*. Point Reyes Station, CA: The Institute for Bird Populations.

DeSante, D. F. (2001). Monitoring Avian Productivity and Survivorship (MAPS): a Sharp rather Than Blunt Tool for Monitoring and Assessing Landbird Populations. *Widlife*, 511–521. doi:10.1007/978-94-011-2868-1_39

DeSante, D. F., Nott, M. P., and O'Grady, D. R. (2001). Identifying the Proximate Demographic Cause(s) of Population Change by Modelling Spatial Variation in Productivity, Survivorship, and Population Trends. *Ardea* 89, 185–207.

Digby, A., Towsey, M., Bell, B. D., and Teal, P. D. (2013). A Practical Comparison of Manual and Autonomous Methods for Acoustic Monitoring. *Methods Ecol. Evol.* 4, 675–683. doi:10.1111/2041-210x.12060

Dinerstein, E., Vynne, C., Sala, E., Joshi, A. R., Fernando, S., Lovejoy, T. E., et al. (2019). A Global Deal for Nature: Guiding Principles, Milestones, and Targets. *Sci. Adv.* 5 (4), eaaw2869. doi:10.1126/sciadv.aaw2869

Dow, R. A. (2013). *Drepanosticta burbachi spec. nov*, from Sarawak, Borneo, a new species allied to *D. dulitensis kimmins*, with notes on related species (Zygoptera: Platystictidae). *Odonatologica* 42, 203–210.

Dow, R. A., and Orr, A. G. (2012). *Telosticta*, a new damselfly genus from Borneo and Palawan (Odonata: Zygoptera: Platystictidae). *Raffles Bull. Zoology* 60, 361–397.

Dow, R. A. (2020). Revision of the genus *Coeliccia* Kirby in Borneo part III: two new species from *C. membranipes*-group from Sarawak and Brunei (Odonata: Zygoptera: Platycnemididae). *Zootaxa*, 4890. doi:10.11646/zootaxa.4890.4.2

Dufrene, M., and Legendre, P. (1997). Species Assemblages and Indicator Species: The Need for a Flexible Asymmetrical Approach. *Ecological* 67, 345–366.

Dunn, E., and Ralph, C. J. (2004). Use of mist nets as a tool for bird population monitoring. *Stud. Avian Biol.* 29, 1–6.

Edwards, D. P., Hodgson, J. A., Hamer, K. C., Mitchell, S. L., Ahmad, A. H., Cornell, S. J., et al. (2010). Wildlife-friendly oil palm plantations fail to protect biodiversity effectively. *Conserv. Lett.* 3, 236–242. doi:10.1111/j.1755-263x.2010.00107.x

Edwards, D. P., Larsen, T. H., Docherty, T. D. S., Ansell, F. A., Hsu, W. W., Derhé, M. A., et al. (2011). Degraded Lands Worth Protecting: The Biological Importance of Southeast Asia's Repeatedly Logged Forests. *Proc. R. Soc. B* 278, 82–90. doi:10.1098/rspb.2010.1062

Ellis, W. J. (2007). "The Planted Forests Project" in Proceedings of the Regional Conference of Biodiversity Conservation in Tropical Planted Forests in Southeast Asia,". Editors R. Stuebing, J. Unggang, J. Ferner, and B. Giman (Kuching: Forest Department, Sarawak Forestry Corporation, Grand Perfect Sdn Bhd), 13–15.

Forest Stewardship Council (2015). FSC International Standard: FSC principles and criteria for forest stewardship. FSC-STD-01-001 V 5-2 EN. 2015. Available at: https://fsc.org/en/document-centre/documents/resource/392 (Accessed September 21, 2021).

Forestry Department of Sarawak (1998). Wildlife Protection Ordinance: Schedule I Parts 1. Available at: https://forestry.sarawak.gov.my/page-0-205-790-Totally-Protected-Animals-of-Sarawak.html and 2 https://forestry.sarawak.gov.my/page-0-206-791-Protected-Animals-of-Sarawak.html (Accessed September 21, 2021).

Francis, C. M., and Wells, D. R. (2003). "The bird community at Pasoh: composition and population dynamics," in *Pasoh: Ecology of a Lowland Rain Forest in Southeast Asia*. Editors T. Okuda, N. Manokaran, Y. Matsumoto, S. C. Thomas, and P. S. Ashton (Tokyo, Japan: Springer-Verlag), 375–393. doi:10.1007/978-4-431-67008-7_27

Furumo, P. R., Barrera-Gonzalez, E. I., Espinosa, J. C., Gomez-Zuluaga, G. A., and Aide, T. M. (2019). Improving long-term biodiversity monitoring on certified oil palm plantations in Columbia by centralizing efforts at the sector level. *Front. For. Glob. Change* 2, 1–7. doi:10.3389/ffgc.2019.00046

Gaston, K. J. (1994). "What is rarity?," in *Rarity*. Editor K. J. Gaston (Dordrecht: Springer), 1–21. doi:10.1007/978-94-011-0701-3_1

Giam, X. (2017). Global Biodiversity Loss from Tropical Deforestation. *Proc. Natl. Acad. Sci. U.S.A.* 114, 5775–5777. doi:10.1073/pnas1706264114

Greenwood, J. D. (2007). Citizens, science, and bird conservation. *J. Ornithol.* 148, S77–S124. doi:10.1007/s10336-007-0239-9

Grinang, J., and Ng, P. K. L. (2021). A new species of the genus *Arachnothelphusa* Ng, 1991 (Crustacea: Decapoda: Gecarcinucidae) from a limestone cave in Sarawak (Malaysian Borneo). *Raffles Bull. Zoology* 69, 1–7. doi:10.26107/RBZ-2021-0001

Griscom, E. W., and Goodman, R. C. (2015). Reframing the sharing vs sparing debate for tropical forestry landscapes. *J. Trop. For. Sci.* 27, 145–147.

Hall, G., Webb, G., and Manolis, C. (2007). "Conservation management plan for the Planted Forest Zone (PFZ), Bintulu, Sarawak" in," in Proceedings of the Regional Conference of Biodiversity Conservation in Tropical Planted Forests in Southeast Asia. Editors R. Stuebing, J. Unggang, J. Ferner, and B. Giman (Kuching: Forest Department, Sarawak Forestry Corporation, Grand Perfect Sdn Bhd), 31–35.

Hui, T. H., and Lim, K. K. P. (2007). "The Freshwater Fishes of the Binyo-Penyilam and Bukit Sarang Conservation Areas, Bintulu Division, Sarawak," in Proceedings of the Regional Conference of Biodiversity Conservation in Tropical Planted Forests in Southeast Asia. Editors R. Stuebing, J. Unggang, J. Ferner, B. Giman, and K. K. Ping (Kuching: Forest Department, Sarawak Forestry Corporation, Grand Perfect Sdn Bhd), 79–82.

Inger, R. F., Stuebing, R. B., and Stuart, B. L. (2006). The Tadpole of Rana Glandulosa Boulenger (Anura: Randiae). *Raffles Bull. Zoology* 54, 465–467.

IUCN (2021). The IUCN Red List of Threatened Species. Version 2021-2. Available at: https://www.iucnredlist.org (Accessed September 20, 2021).

Kiew, R., and Julia, S. (2013). Two new Monophyllaea (Gesneriaceae) Species from Sarawak, Borneo. *Phytotaxa* 129, 59–64. doi:10.11646/phytotaxa.129.1.6

Kiew, R., and Sang, J. (2009). Seven new Species of *Begonia* (Begoniaceae) From the Ulu Merirai and Bukit Sarang Limestone Areas in Sarawak, Borneo. *Gard. Bull. Singap.* 60, 351–372.

Klingbeil, B. T., and Willig, M. R. (2015). Bird Biodiversity Assessments in Temperate Forest: the Value of Point Count Versus Acoustic Monitoring Protocols. *PeerJ* 3, e973. doi:10.7717/peerj.973

Kollert, W., and Lagan, P. (2007). Do Certified Tropical Logs Fetch a Market Premium? *For. Policy Econ.* 9, 862–868. doi:10.1016/j.forpol.2006.03.005

Krause, B., and Farina, A. (2016). Using Ecoacoustic Methods to Survey the Impactsof Climate Change on Biodiversity. *Biol. Conserv.* 195, 245–254. doi:10.1016/j.biocon.2016.01.013

Lahoz-Monfort, J. J., and MaGrath, M. J. L. (2021). A Comprehensive Overview of Technologies for Species and Habitat Monitoring and Conservation. *Bioscience* 71, 1038–1062. doi:10.1093/biosci/biab073

Leach, E. C., Burwell, C. J., Ashton, L. A., Jones, D. N., and Kitching, R. L. (2016). Comparison of Point Counts and Automated Acoustic Monitoring: Detecting Birds in a Rainforest Biodiversity Survey. *Emu - Austral Ornithol.* 116, 305–309. doi:10.1071/mu15097

Lin, C.-W., Chung, S.-W., and Peng, C.-I. (2014). Three new species of Begonia (sect. Petermannia, Begoniaceae) from Sarawak, Borneo. *Phytotaxa* 191, 129–140. doi:10.11646/phytotaxa.191.1.8

Lin, W., Chung, S-W., and Peng, C-I. (2014). Begonia Hosensis (sect. Reichenheimia, begoniaceae), a new Species from Sarawak, Malaysia. *Taiwania* 59, 326–330. doi:10.6165/tai.2014.59.4.326

Lin, W., Chung, S-W., and Peng, C-I. (2017). Eleven new Species of Begonia (Begoniaceae) from Sarawak, Borneo. *Taiwania* 62, 219–251. doi:10.6165/tai.2017.62.219

Martin, T. E., Blackburn, G. A., and Simcox, W. (2010). An Assessment of the Effectiveness of Two Methods in Describing A Neotropical Cloud Forest Bird Community. *Ornitol. Neotropical* 21, 131–147.

McCune, B., and Mefford, M. J. (2018). *PC-Ord for Windows. Multivariate Analysis of Ecological Data*. Version 7.08. Glenedon Beach: MjM Software.

Meijaard, E., and Sheil, D. (2012). The Dilemma of Green Business in Tropical Forests: How To Protect What It Cannot Identify. *Conserv. Lett.* 5, 342–348. doi:10.1111/j.1755-263x.2012.00252.x

Metcalf, O. C., Barlow, J., Marsden, S., Gomes de Moura, N., Berenguer, E., Ferreira, J., et al. (2021). Optimizing Tropical Forest Bird Surveys Using Passive Acoustic Monitoring and High Temporal Resolution Sampling. *Remote Sens. Ecol. Conserv.* 8, 45–56. doi:10.1002/rse2.227

Michener, W. K. (2015). Ecological Data Sharing. *Ecol. Inf.* 29, 33–44. doi:10.1016/j.ecoinf.2015.06.010

Myers, S. (2016). *Birds of Borneo: Sabah, Sarawak, Brunei and Kalimantan*. 2nd ed. London: Helms.

Neate-Clegg, M. H. C., Horns, J. J., Adler, F. R., Kemahlı Aytekin, M. Ç., and Şekercioğlu, Ç. H. (2020). Monitoring The World's Bird Populations With

Community Science Data. *Biol. Conserv.* 248, 108653. doi:10.1016/j.biocon.2020.108653
Ng, P. K. L. (2021). *Geosarma sodalis*, a new Species of Vampire Crab (Crustacea, Brachyura, Sesarmidae) from a Limestone Cave in Central Sarawak, Malaysia. *Zookeys* 1031, 133–141.
North American Banding Council (2001). *The North American Banders' Study Guide*. NABC Publications Committee. Available at: http://www.nabanding.net/wp-content/uploads/2012/04/STUDYGUIDE1.pdf.
Parker, T. A., III. (1991). On the use of Tape Recorders In Avifaunal Surveys. *Auk* 108, 443–444.
Phillipps, Q., and Phillipps, K. (2014). *Phillips' Guide to the Birds of Borneo: Sabah, Sarawak, Brunei and Kalimantan*. 3rd ed. Princeton, New Jersey: PrincetonJohn Beaufoy.
Phillipps, Q., and Phillipps, K. (2016). *Phillips' Guide to the Mammals of Borneo and Their Ecology: Sabah, Sarawak, Brunei, and Kalimantan*. 3rd ed. Princeton, New Jersey: PrincetonJohn Beaufoy.
Purnomo, H., Okarda, B., Dermawan, A., Ilham, Q. P., Pacheco, P., Nurfatriani, F., et al. (2020). Reconciling oil Palm Economic Development and Environmental Conservation in Indonesia: A Value Chain Dynamic Approach. *For. Policy Econ.* 111, 102089. doi:10.1016/j.forpol.2020.102089
R Core Team. (2021). *R4.1.0: A Language and Environment for Statistical Computing*. Vienna, Austria: R Foundation for Statistical Computing. https://www.R-project.org/. (Accessed May 18, 2021).
Rahman, M. R. A., Achmadi, A. S., Tingga, R. C. T., and Hasan, N. H. (2010). A New Distributional Record of The Rare Bat *Coelops Robinsoni* from Sarawak, Malaysian Borneo. *J. Trop. Biol. Conservation* 7, 87–92.
Ralph, C. J., Droege, S., and Sauer, J. R. (1995). Managing and Monitoring Birds Using Point Counts: Standards And Applications. *USDA For. Serv. Gen. Tech. Rep.* 149, 161–168.
Remsen, J. V., and Good, D. A. (1996). Misuse Of Data From Mist-Net Captures To Assess Relative Abundance In Bird Populations. *Auk* 113, 381–398. doi:10.2307/4088905
Senior, M. J. M., Brown, E., Villalpando, P., and Hill, J. K. (2015). Increasing the Scientific Evidence Base in the "High Conservation Value" (HCV) Approach for Biodiversity Conservation in Managed Tropical Landscapes. *Conserv. Lett.* 8, 361–367. doi:10.1111/conl.12148
Sheldon, F. H., Styring, A., and Hosner, P. A. (2010). Bird Species Richness in a Bornean Exotic Tree Plantation: A Long-Term Perspective. *Biol. Conserv.* 143, 399–407. doi:10.1016/j.biocon.2009.11.004
Sollmann, R., Mohamed, A., Niedballa, J., Bender, J., Ambu, L., Lagan, P., et al. (2017). Quantifying Mammal Biodiversity Co-Benefits in Certified Tropical Forests. *Divers. Distrib.* 23, 317–328. doi:10.1111/ddi.12530
Stuebing, R. B. (2007). "Biodiversity Conservation in Theory And Practice: Lessons From The Planted Forest Zone, Sarawak," in Proceedings of the Regional Conference of Biodiversity Conservation in Tropical Planted Forests in Southeast Asia. Editors R. Stuebing, J. Unggang, J. Ferner, and B. Giman (Kuching: Forest Department, Sarawak Forestry Corporation, Grand Perfect Sdn Bhd), 16–30.
Styring, A. R., Ragai, R., Unggang, J., Stuebing, R., Hosner, P. A., and Sheldon, F. H. (2011). Bird Community Assembly in Bornean Industrial Tree Plantations: Effects of Forest Age And Structure. *For. Ecol. Manag.* 261, 531–544. doi:10.1016/j.foreco.2010.11.003
Styring, A. R., Unggang, J., Jukie, A., Tateh, O., Megom, N., and Sheldon, F. H. (2018). Bird Community Structure in Native Forest Fragments And *Acacia Mangium* Plantations In Borneo. *Wilson J. Ornithol.* 130, 112–130. doi:10.1676/16-125.1
Tayleur, C., Balmford, A., Buchanan, G. M., Butchart, S. H. M., Ducharme, H., Green, R. E., et al. (2016). Global Coverage of Agricultural Sustainability Standards, And Their Role In Conserving Biodiversity. *Conserv. Lett.* 10, 610–618. doi:10.1111/conl.12314
The Economist Intelligence Unit (2021). An Eco-Awakening: Measuring Global Awareness, Engagement, And Action For Nature. Available at: https://files.worldwildlife.org/wwfcmsprod/files/Publication/file/93ts5bhvyq_An_EcoWakening_Measuring_awareness__engagement_and_action_for_nature_ FINAL_MAY_2021.pdf?_ga2.162883323.630328229.1631817695- 1444868052.1631817695 (Accessed September 21, 2021).
Trolliet, F., Vogt, M., and Trolliet, F. (2019). "How Does Fsc Certification Oof Forest Management Benefit Conservation Of Biodiversity?" in *Sustainability and Certification Schemes in the Agricultural and Natural Resource Sectors*. Editor M. Vogt (London: Routledge), 93–110. doi:10.4324/9780203701737-4
Tulloch, A. I. T., Auerbach, N., Avery-Gomm, S., Bayraktarov, E., Butt, N., Dickman, C. R., et al. (2018). A Decision Tree for Assessing the Risks and Benefits of Publishing Biodiversity Data. *Nat. Ecol. Evol.* 2, 1209–1217. doi:10.1038/s41559-018-0608-1
Wong, S., Servheen, C., Ambu, L., and Norhayati, A. (2005). Impacts of Fruit Production Cycles on Malayan sun Bears and Bearded Pigs in Lowland Tropical Forest of Sabah, Malaysian Borneo. *J. Trop. Ecol.* 21, 627–639. doi:10.1017/s0266467405002622
Walther, B. A., and White, A. (2018). The Emergence of Birdwatching in China: History, Demographics, Activities, Motivations, and Environmental Concerns of Chinese Birdwatchers. *Bird. Conserv. Int.* 28, 337–349. doi:10.1017/s0959270917000557
Wheeldon, A., Mossman, H. L., Sullivan, M. J. P., Mathenge, J., and de Kort, S. R. (2019). Comparison of Acoustic and Traditional Point Count Methods to Assess Bird Diversity and Composition In the Aberdare National Park, Kenya. *Afr. J. Ecol.* 57, 168–176. doi:10.1111/aje.12596
Wiggins, A., Bonney, R., Graham, E., Henderson, S., Kelling, S., Littauer, R., et al. (2013). *Data Management Guide for Public Participation in Scientific Research*. Albuquerque, NM: DataONE.
Wilcove, D. S., and Koh, L. P. (2010). Addressing the Threats to Biodiversity from Oil-Palm Agriculture. *Biodivers. Conserv.* 19, 999–1007. doi:10.1007/s10531-009-9760-x
WWF (2009). High Conservation Value Forest (Hcvf) Toolkit for Malaysia: A National Guide fFor Identifying, Managing and Monitoring High Conservation Value Forests. Available at: https://hcvnetwork.org/wp-content/uploads/2018/05/2009HCVF-Toolkit-Malaysia_english.pdf (Accessed September 21, 2021).

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A Comparison of Biophysical Conditions Between Sundanese Migrant and Non-Migrant *Pekarangans* in Indonesia

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Pekarangan is a typical Indonesian home garden. This article aimed to look at biophysical conditions of *pekarangan* between Sundanese migrants and non-migrants. A total of 40 *pekarangans* in Selajambe and Ciomas Rahayu villages, West Java, were chosen as representative locations for the Sundanese non-migrant population (native Sundanese), and 40 *pekarangans* in Tegal Yoso and Tanjung Kesuma villages, Lampung, were chosen as representatives of the Sundanese migrant population. Research has been carried out in the period 2019–2021. To measure the biophysical conditions of *pekarangans*, we analyzed the *pekarangan* area, *pekarangan* size, number of species and individual of *pekarangan* plants, vertical diversity and horizontal diversity of plants, and the relationship between the *pekarangan* area and number of species and individual plants. The results showed that the difference in conditions of the *pekarangan* was indicated by the difference in the area and size but not by the diversity of the plants. Both types of *pekarangans* have the same level of diversity, as indicated by the number of individual plants that are almost the same in number per 100 m^2. In addition, a strong and positive correlation (0.69–0.88) between the area of *pekarangan* and the number of individual plants indicated that the small to medium size or large *pekarangan* sizes had almost the same diversity of plants. The difference lied in the type of plant that is cultivated. Migrant *pekarangans* are dominant in cultivating food crops, while non-migrant *pekarangans* are dominant in cultivating ornamental plants. The selection of plants that have important and valuable functions can be a solution in maintaining the area of the *pekarangan*. Choosing plants with a variety of functions can be an option for a small to medium *pekarangan* size. To improve the biophysical conditions of the *pekarangan* was also inseparable from the involvement of economic, social, and cultural aspects in the *pekarangan*.

Keywords: horizontal diversity, preferences, Sundanese ethnic, transmigration program, vertical diversity

INTRODUCTION

Pekarangan is a typical Indonesian home garden associated with the house (Arifin et al., 1998; Arifin et al., 2012; Hakim, 2014). As was the case with landscapes typical of other countries such as *satoyama* in Japan (Indrawan et al., 2014), *kihamba* in Tanzania (Santoro et al., 2020), and permaculture in Australia (Mollison, 1979), the *pekarangan* had its own characteristics, that is, the vegetation structure was characterized by a combination of multilayer plants, ranging from trees to shrubs, as well as its integration with livestock and fish. (Arifin et al., 1997). The condition of the garden can describe the relationship between the owner and the ecological characteristic of their environment (Kiesling and Manning, 2010). Therefore, *pekarangan* as a landscape unit must have various functions, ecologically, economically, and socio-culturally (Irwan and Sarwadi, 2017).

The research that has succeeded in identifying the function of the home garden ecologically, such as the home garden as a reservoir of plant diversity (Caballero-Serrano et al., 2016; Chatterjee et al., 2017; Gbedomon et al., 2017), especially traditional food crops (Galluzzi et al., 2010), non-timber forest products (Mohri et al., 2013), shade plants, and ornamental plants (Abebe et al., 2010), increased food diversity and family nutrition (van der Stege et al., 2010; Caballero-Serrano et al., 2019; Thamilini et al., 2019), such as fruit crops, vegetables (Mohri et al., 2013; Ali, et al., 2021), medicinal plants (Abebe et al., 2010), spice plants, and starch-producing plants (Arifin et al., 2012).

The role of the *pekarangan* was very important for biodiversity conservation and urban planning. Diverse garden plants have directly helped plant conservation activities (Webb and Kabir, 2009; Idohou et al., 2014). The types of plants that were planted came from the components of trees, bushes, and ground cover plants. (Arifin et al., 1998; Webb and Kabir, 2009). It has indirectly provided a habitat for wildlife, such as a variety of birds and other animals (Muwav and Bekessy, 2017). For urban planning, the home garden (*pekarangan*) is a green open space that is close to the family (Coolen and Meesters, 2012). In addition, the yard also has the function of increasing food for the family (Drescher et al., 2006). Therefore, *pekarangan* plants can also ameliorate the microclimate in urban areas (Budiastuti et al., 2018).

The transmigration program has been one of the flagship programs of the Indonesian government since the new order era until the reformation era (Titus, 1992; Fearnside, 1997; Ricklefs, 2008; Prihatin, 2013). The transmigration program aimed to improve the welfare of transmigrants and their surrounding communities, increase and equalize regional development, and strengthen national unity and integrity (UURI No.15 Year 1997). Each family who got this program received 0.25 ha of land for the house and garden (*pekarangan*) and also got 2 ha of farm land. Among the ethnicities on the island of Java who received this transmigration program was Sundanese ethnic (West Java) (Nyhus and Sumianto, 1999). One of the transmigration areas that had been developed since 1952 was the Purbolinggo sub-district (previously it was part of the Central Lampung Regency, but in 1999, a new regency was formed: East Lampung Regency so that the Purbolinggo sub-district became part of it). Lampung became the first and largest transmigration destination province by the government (Titus, 1992; Nyhus and Sumianto, 1999) not only for reasons of its strategic geographical position but also in terms of demographic aspects (Khoiriyah et al., 2019).

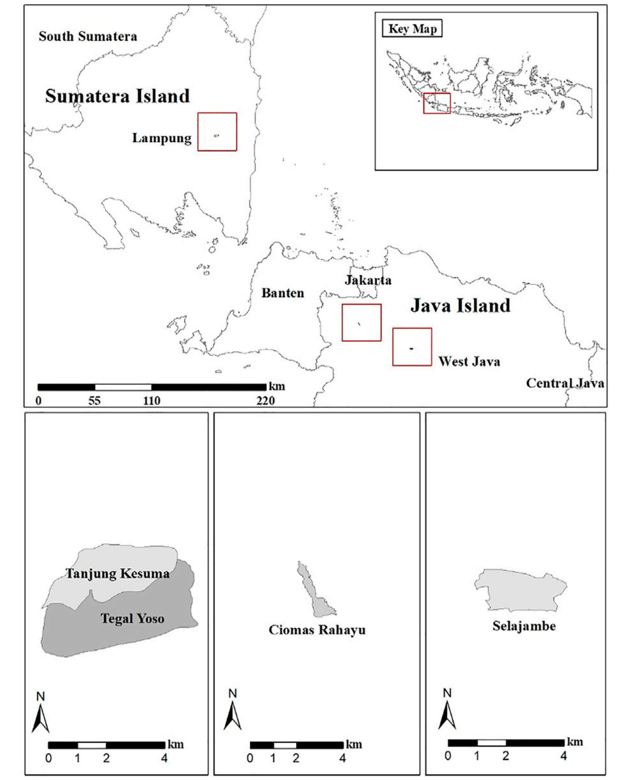


FIGURE 1 | Study site of Sundanese migrant and non-migrant *pekarangans* in Purbolinggo, Lampung, and Bogor—Cianjur, West Java Province.

Biophysical conditions of the *pekarangan* in a transmigration area are reported from various studies. The condition of *pekarangan* sustainability in the transmigration area at Central Sulawesi Province has been studied by Kehlenbeck and Maass (2006). The comparison of *pekarangan* species diversity in transmigration and non-transmigration areas has been reported by Kehlenbeck et al. (2007). Research about the *pekarangan* also reported the policy of intensification of the *pekarangan* in the transmigration settlement unit IV SP-6 Alue Peunyareng (Rananggono, 2012) and the importance of optimizing *pekarangan* as a model for developing transmigrant areas, Waplau District, Buru Regency, and Maluku Province (Nugraha et al., 2015). The use of a *pekarangan* with the agroforestry system in Sidomulyo Village, Katingan, Central Kalimantan, has been reported by Yustha (2017).

The results of research in the Sundanese *pekarangan* reported increasing the function of fruit and vegetable plants in Bogor and Cianjur (Ali et al., 2021). The ecological minimum size of the *pekarangan* was found to be 100 m^2 (Arifin et al., 1997; Arifin et al., 1998). The *pekarangan* was used as a place to increase food

TABLE 1 | Characterization of study sites.

Characterization variable	Migrant	Non-migrant
No. of sample (n)	40	40
Villages	Tegal Yoso—Tanjung Kesuma	Selajambe—Ciomas Rahayu
Sub-districts	Purbolinggo	Sukaluyu—Ciomas
Regencies/districts	East Lampung	Cianjur—Bogor
Provinces	Lampung	West Java
Kind of areas	Rural	Rural—Suburban
Year of transmigration	1952–1953	-
Distance from the nearest city (Km)	14	9–12
Elevation (m)	25–55	200–316
Rainfall/years (mm)	2,000–2,500	1,000–4,000
Average of temperature (°C)	27.8	26.5–27
People/household	4	4
Average of income/month (IDR)	1,000,000–1,505,000	1,500,000–2,700,000
Major employment	Farmer	Self-employed

diversity and nutrition for family (Azra et al., 2014) as household income (Antoh et al., 2019) and the commercialization of the *pekarangan* as a place to plant commercial crops (Prihatini et al., 2018; Abdoellah et al., 2020). The size of the non-migrant *pekarangan* was reduced due to urbanization factors (population, economic, and technological growth) (Seto, 2011; Ali et al., 2021) and the *pekarangan* land was fragmented by inheritance system, sale, and construction of new buildings (Arifin et al., 1998; Azra et al., 2014; Ali et al., 2021). Are there any differences of biophysical conditions between Sundanese migrants' and non-migrants' *pekarangan*? Therefore, the purpose of this article was to compare biophysical conditions of *pekarangan*s between Sundanese migrants' and non-migrants' *pekarangan*.

METHODS

Study Sites

The study areas of this research were located in Selajambe—Ciomas Rahayu Village, West Java, Indonesia; and in Tegal Yoso—Tanjung Kesuma Village, the transmigration area of East Lampung (**Figure 1**; **Table 1**). Research in Selajambe—Ciomas Rahayu was conducted in October–December 2019, and research in Tegal Yoso—Tanjung Kesuma was conducted in June–July 2021. Selajambe—Ciomas Rahayu village was chosen to be the representative of the Sundanese living on the island of Java, while Tegal Yoso—Tanjung Kesuma Village was chosen due to majority of the population being Sundanese who transmigrated to the East Lampung area. Selajambe—Ciomas Rahayu village was chosen due to being a rural area, and also, the average proximity of these villages to the city center (economic activities) ranges from 9 to 14 km. There were 40 *pekarangans taken* from each study area, so the total number of samples was 80 *pekarangan*s. The samples were determined by the purposive sampling technique (Sundanese). The number of samples was determined according to the sample determination by Arifin et al. (1998) and Ali et al. (2021). In total, 10 samples in Ciomas Rahayu Village and 30 samples in Selajambe Village were based on the representation of the number of *pekarangan* in the research location, respectively. Therefore, the villages of Tegal Yoso and Tanjung Kesuma followed these provisions to make comparisons easier. By collecting the data during the COVID-19 pandemic, we conducted research by implementing strict health protocols. We also collected some respondent data by an online survey.

Biophysical Conditions of a *Pekarangan*

To analyze the biophysical conditions of a *pekarangan*, there were four minimal variables of a *pekarangan*, that is, measuring the area and size, *pekarangan* zoning, number of species and individual plants per *pekarangan*, and the vertical diversity and horizontal diversity of plants. In this article, some of the *pekarangan* conditions of the *pekarangan* are measured such as the *pekarangan* area (m^2), size of the *pekarangan* (small to extra larges), the zone of the *pekarangan,* the number of species and individual plants per *pekarangan*, and the vertical and horizontal diversity of plants. In addition, this study also calculated the effect of the *pekarangan* area on the number of individual plants per *pekarangan*. These measurements were carried out to show the differences of biophysical conditions of the *pekarangan* between Sundanese migrants and non-migrants.

The area of the *pekarangan* (m^2) and the size of *pekarangan* are the important things in ecological value because the owner can use it to plant various plants, especially tree species. This can provide natural shade, provide fresh air, and also benefit from the fruit. In addition, the area of the *pekarangan* can also be a water catchment area when it rains, so that it becomes a source of water reserves that can be used for plants. Loss of yard area due to conversion to other uses will cause impacts such as the increase in temperature around the house because of the unavailability of land to plant trees, making the air feel hotter, causing large run-off when it rains, and causing puddles. Although currently small *pekarangans* are widely used with vertical garden patterns and potted plants, shade functions, fresh air, and water absorption cannot be provided well by a small *pekarangan*.

The *pekarangan* sizes (Arifin et al., 1998) were determined by data-centered descriptive statistics (Kaur et al., 2018), such as the mean, median, minimum value, maximum value, and standard deviation.

TABLE 2 | Criteria of vertical and horizontal diversities of plants.

Diversity	Information
Vertical	Plant height
V	Tree > 10 m
IV	5 m < large shrub, small tree ≤ 10 m
III	2 m < bush height, small shrub ≤ 5 m
II	1 m < herb, bush ≤ 2 m
I	Grass and shrubs ≤ 1 m
Horizontal	Plant function
1	Ornamental plants
2	Fruit plant
3	Vegetable plant
4	Spice plant
5	Medicinal plants
6	Starch-producing plants
7	Industrial raw material plant
8	Other crops (producing feed, firewood, shade, etc.)

Sources: Arifin et al. (1998); Arifin et al. (2010); Arifin et al. (2012).

$$\mathrm{u} = \text{Land Area}\ (\mathrm{m}^2) - \text{Building Area}\ (\mathrm{m}^2), \tag{1}$$

$$\widehat{\mathrm{u}} = \frac{1}{n}\sum_{i=1}^{n} ui. \tag{2}$$

u = *pekarangan* area. û = average *pekarangan* area. ui = area of the i-th *pekarangan*. n = number of sample *pekarangan*s

The *pekarangan* sizes were then grouped into Small ≤ 120 m^2, 120 m^2 < Medium ≤ 400 m^2, 400 m^2 < Large ≤ 1,000 m^2, and Extra Large > 1,000 m^2 (Arifin et al., 2012). The zoning of the *pekarangan* divided the *pekarangan* into four zones, that is, the front *yard*, left yard, right yard, and backyard (Arifin et al., 1998; Arifin et al., 2010). The number of species and individual plants per *pekarangan* was determined per 100 m^2. The same unit area was needed to compare (Peng et al., 2018) the number of plant species and individuals in migrants' and non-migrants' *pekarangans*. Vertical diversity was the grouping of plants based on the plant height, and horizontal diversity was the grouping of plants based on the plant function (**Table 2**) (Arifin et al., 1998; Arifin et al., 2010; Arifin et al., 2012).

Simple linear regression analysis was largely used to analyze between two biophysical conditions (Nelson, 2009) and also was conducted to see the effect of the *pekarangan* area on the number of individual plants in each location. Calculations and data processing were carried out in Microsoft Excel 2016. Shorting data was carried out to discard data with a high error value.

Knowing the biophysical condition of the *pekarangan* based on the type of migrants and non-migrants from the same ethnicity (Sundanese) provides evidence that whether different *pekarangan* areas and sizes of the *pekarangan* will have an impact on different plant diversities, both in species and number of individuals.

RESULTS AND DISCUSSION

Characterization of Study Sites

As a comparison area for the native Sundanese ethnic, Selajambe and Ciomas Rahayu villages were chosen. They are located in the West Java Province. The population of two villages consisted of 80% of the original residents (Sundanese Bogor and Cianjur), and 20% were immigrants (Sundanese from another area). These two villages were characterized by rural areas that have been heavily affected by urbanization. There were fewer people working as farmers. Many residents switched to work as entrepreneurs or Indonesian migrant workers (TKI) in foreign countries (Kertawibawa and Harun, 2012). The average income of the population was 1.5–2.7 million rupiahs/person/month. The increasingly expensive prices of basic necessities have forced residents to switch jobs to more promising sectors. In the Selajambe Village, there were still irrigated rice fields, but their existence continued to be eroded by the construction of garment factories. This development occurred because this village was traversed by the highway that connected Cianjur and Bandung. This definitely affected the biophysical conditions of the area including the *pekarangan*.

Lampung province has been known as a transmigration area since the Dutch colonial era. The transmigration program in Lampung province, specifically in East Lampung Regency, occurred in the period 1952–1953. Most of the people who transmigrated came from the island of Java (Sundanese and Javanese ethnics). Purbolinggo, as one of the sub-districts in the East Lampung Regency, consisted of Sundanese transmigrants. Tegal Yoso and Tanjung Kesuma villages were villages where the majority of them were Sundanese ethnic. The population of the two villages was 45% from Bandung, 40% from Sumedang, 13% from Tasikmalaya, and 3% from Majalengka, West Java Province. The annual rainfall and temperature in the Purbolinggo sub-district were suitable for people to carry out agricultural activities. Both villages were characterized by rural areas with the main commodities, such as rice and corn. The agricultural land system was based on the irrigation system. In December, usually farmers planted rice, and in June, farmers planted corn. The distance between two villages and Sukadana (the capital city of East Lampung Regency) was about 14 km. The two villages were also close to the Sumatran East Coast National Road. The average monthly income of residents who work as farmers was 1–1.5 million rupiahs/person/month. This number was higher than the Lampung poverty line, which was IDR 457,495/person/month.

Pekarangan: Performance, Area, Size, and Zone

The dynamics of changes that occurred especially in non-migrant *pekarangan*s were strongly influenced by urbanization factors. In Arifin et al. (1998), when the first research of a *pekarangan* had been conducted in those locations, the average area of the *pekarangan* in Selajambe—Ciomas Rahayu was 364.7 m^2. In 2019, the average area of the *pekarangan* was 150.7 m^2 (Ali et al., 2021), it was from the medium size in 1998 to be the small size in 2019. The reason of decrease was due to urbanization during 2 decades (1998–2019). The four urbanization factors that mostly influenced changes in the *pekarangan* were the increasing level of education, the use of technology, the increase in the average income of the community, and the increase in the built-up area (Ali et al., 2021). It happened because the economic development of the

TABLE 3 | Area of Sundanese migrant and non-migrant *pekarangans*.

Variable	Migrant	Non-migrant
Mean (m^2)	733.1[a]	150.7[a]
Median (m^2)	689.0[a]	85.8[a]
Minimum (m^2)	215.0	5.0
Maximum (m^2)	1994.0	748.0
Standard deviation	371.8	184.1

[a]*Means and medians in a row are significantly different at* p *0.05.*

TABLE 4 | Average number of species and individual plants of migrant and non-migrant *pekarangans* per 100 m^2.

Pekarangan	Number of plants	
	Species	Individual
Migrant	4	50
Non-migrant	19	53

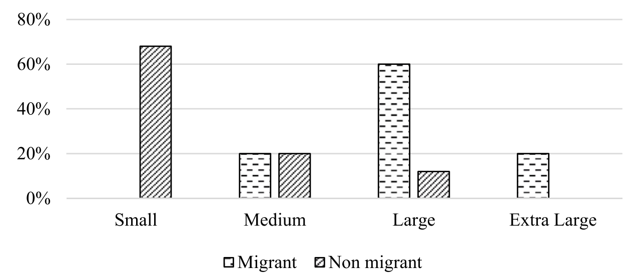


FIGURE 2 | Size of Sundanese migrant and non-migrant *pekarangans*.

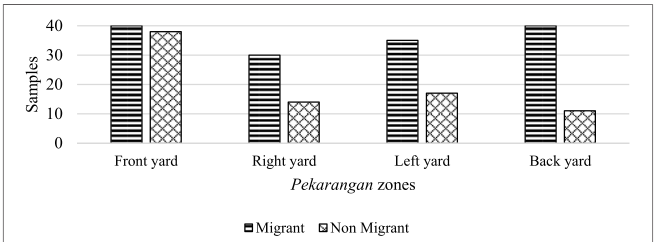


FIGURE 3 | Zones of Sundanese migrant and non-migrant *pekarangans*.

island of Java was much higher than the island of Sumatra. The level of education and ease of access to technology made the development of rural areas into suburban and even urban areas faster. The need for a place to live or a place to sell was also an important reason so that the *pekarangan* land became the first land to be converted into the new building. For the Sundanese themselves, the *pekarangan* became one of the inheritances that were divided among the children, thus making the *pekarangan* land fragmented and its size from medium to small. It was also happening in the migrant *pekarangan*. The *pekarangan* land that used to be large was handed down to children as a land to build a place to live (house). Although the decrease in the *pekarangan* area was still small, it was proven that the current *pekarangan* size was 80% large and extra-large. Does the difference in the size of the *pekarangan* between a migrant and non-migrant one affect the diversity of species and individual plants? This will be explained in the species and individual plants of *pekarangan* sections.

The performance of the migrant *pekarangan* is not much different from the non-migrant one. They still brought Sundanese habits and culture to the transmigration area. The difference was showed by the different sizes of the *pekarangan*. Migrant *pekarangans* were wider because the transmigration program provided land to build houses and *pekarangan*s on average of 0.25 ha. Currently, the average land area has been only 1,200 m^2. Since 1952 until now, the average *pekarangan* area of his/her house is 733.1 m^2 (large size), and 80% of the migrant *pekarangan* size is still in large and extra-large sizes (**Table 3**; **Figure 2**). The median value indicated that the size of the migrant *pekarangan* was ecologically well. Its size was above the minimum size of 100 m^2, while the non-migrant *pekarangan* was already below 100 m^2 (Arifin et al., 1998).

The front yard zone was almost found in migrant and nonmigrant *pekarangans* (**Figure 3**). The existence of a right and left yard in migrant *pekarangans* was higher than that of the non-migrant one and also the existence of a backyard. The front yard is still predominant because it is an important part of the house and can be used as a welcoming area. By that reason, the front yard was planted with many ornamental plants and other decoration stuff. Furthermore, the front yard was a characteristic sign of a house. It was shown by the Sundanese *pekarangans*. The other three zones began to decline due to the widening of houses or the construction of new buildings, for example, store, garage, etc. (Azra et al., 2014; Ali et al., 2020). Another reason for their decline in existence was due to being sold (Ali et al., 2021). The front yard zone was expected to be the most durable zone in a *pekarangan* because it had an important role and function for householders, especially the small *pekarangan* in urban areas. Nowadays, the front yard is not only being planted with the ornamental plants but also planted with vegetables, fruits, medicine, and spice plants. In particular, during this COVID 19 pandemic, many householders used their *pekarangan* while at home (work from home) for gardening, exercising, and other activities (Arifin et al., 2021; Montefrio, 2020; Sofo and Sofo 2020). However, in Arifin et al. (1998), the potential zone to be the most durable zone was the backyard. It was considered to be a potential space for biodiversity conservation such as food and medicinal plants, livestock, and fish ponds. The front yard was prone to change because it had a huge potential of being used for the construction of new buildings such as stall, workshop, and garage. (Arifin et al., 1998). Those activities were believed to have a positive influence in maintaining and increasing the immunity of the human body both physically and psychologically (Clatworthy et al., 2013; Buck, 2016; Soga et al., 2017; Corley et al., 2021). Because of that, the *pekarangan*, as the closest landscape unit in the house, was considered to be the best choice for doing those activities. This

FIGURE 4 | Conditions of the Sundanese migrant **(A)** and non-migrant **(B)** *pekarangans*.

phenomenon was occurring in urban, suburban, and rural areas (Sofo and Sofo, 2020).

Sundanese migrants and non-migrants lived close to their families. It has been proven by the position of their house. There was a place to dry agricultural products (rice or corn) in the front yard of Sundanese migrants' *pekarangan* (**Figure 4A**). The front yard, called *buruan* in Sundanese, was not only planted with ornamental plants such as in non-migrant *pekarangans* but also planted with food crops, for example, fruits, vegetables, spices, medicine, etc. (**Figure 4A**). The characteristics of the front yard of the Sundanese non-migrants were dominated by ornamental plants (**Figure 4B**). The majority of the front yards of non-migrants have been paved with concrete, asphalt, or cone blocks, while for non-migrants, the front yard was still left with soil. In migrant *pekarangans,* almost all the vegetations were directly planted on the ground, without the use of pots or planter boxes, while in non-migrant *pekarangans*, they planted them in pots or planter boxes.

Plant Species and Plant Individuals of the *Pekarangan*

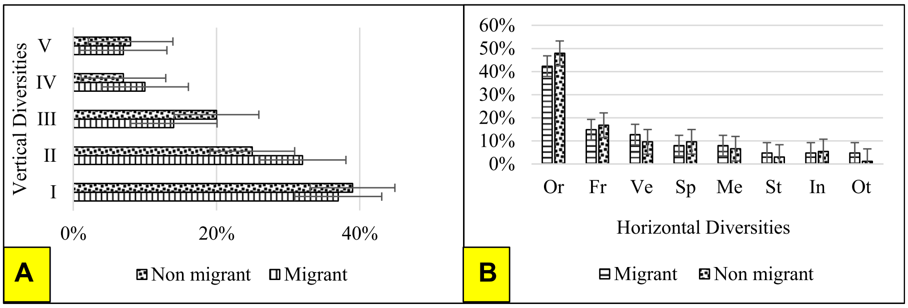
The difference in the average number of species planted in *pekarangan* migrants and non-migrants was different, but the number of individual plants was almost the same. This indicated that per 100 m^2 of the *pekarangan* area had almost the same number of individual plants, although the number of species was different (**Table 4**). The difference in the number of species occurs because the average area of the *pekarangan* was different. The average area of 150 m^2 in non-migrant *pekarangan* was used to plant various types of plant species as much as possible, both for ornamental and food functions. This also indicated that both migrant and non-migrant *pekarangans* were being used well by their owners. This can be seen in the number of individual plants which are almost the same in every 100 m^2 of the *pekarangan* area. The differences in the number of species depend on the type of plant (ground cover plants to high trees). In the migrant *pekarangan*, it was proven by three plant species with a large number of individuals, such as kale, leek (vegetable), and cassava (starch). In non-migrant *pekarangans*, it was also proven by 19 plant species with 53 individual plants. The plant species consisted of *Pleomele*, wild tea, euphorbia, *Sansevieria*, aloe vera, cordyline, asplenium (ornamental), banana, rambutan, mango, papaya, jackfruit (fruit), tree spinach, *Polyscias* (vegetable), turmeric, cayenne pepper, galangal (spice), sweet potato, and cassava (starch).

Vertical and Horizontal Diversity of Plants in a *Pekarangan*

The vertical diversity of plants in the *pekarangan* showed plant strata ranging from ground cover to high trees because the *pekarangan* looked like a forest which had layers of plants. In addition, there was also horizontal diversity which groups plants according to the functions mentioned by the owner of the *pekarangan*. Therefore, the use of plants in each ethnic was different. It was influenced by culture, mainly culinary and belief systems. The differences in the vertical diversity and horizontal diversity of plants between migrant and non-migrant *pekarangans* were not much different.

The total species of plants in migrant and non-migrant *pekarangans* were 189 and 167 species, respectively. From that number, it was clearly divided into the vertical diversity and horizontal diversity. In vertical diversity of plants in the *pekarangan*, there were 14 species in stratum V in both migrant and non-migrant *pekarangans*. There were 19 and 12 species of stratum IV in migrant and non-migrant *pekarangans.* There were 26 and 34 species of stratum III in migrant and non-migrant *pekarangans.* There were 61 and 42 species of stratum II in migrant and non-migrant *pekarangans.* Therefore, there were 69 and 65 species of stratum I in migrant and non-migrant *pekarangans.* The total ornamental (Or) and fruit (Fr) plants in migrant and non-migrant *pekarangans* were 80 and 28 species, respectively. The total vegetable (Ve), spice (Sp), medicine (Me), and starch (St) plants in migrant and non-migrant *pekarangans* were 24 and 16, 15 and 16, 15 and 11, and 9 and 5 species, respectively. The total industrial (In) plants in both migrant and non-migrant *pekarangan* were 9 species, respectively, and the last one, other (Ot) uses of plants were 9 and 2 species, respectively.

In **Figure 5A**, it can be seen that the condition of the vertical diversity of *pekarangan*s in migrants showed the presence of stratum I (height of plant under 1 m), which was higher than the other four strata. The interesting one was that the percentage of strata IV (height of trees 5–10 m) and V (height of tree > 10 m) was almost the same between the two research sites. It indicated



Ornamental (or), Fruit (Fr), Vegetable (Ve), Spices (Sp), Medicine (Me), Starch (St), Industrial (In), and Other (Ot) use of plant.

FIGURE 5 | Vertical diversity **(A)** and horizontal diversity **(B)** of plants in migrant and non-migrant *pekarangans*. Ornamental (or), fruit (Fr), vegetable (Ve), spices (Sp), medicine (Me), starch (St), industrial (In), and other (Ot) uses of plants.

that plants with a height of more than 5 m were still present in non-migrant *pekarangan*s, even though the size of the *pekarangan*s has declined. The tree plants that were maintained a lot must have had more functions for the owner. These trees usually had ecological functions such as climate amelioration (shelters) and food functions (fruit trees). The existence of shady trees, especially in sub-urban *pekarangan* areas such as Ciomas Rahayu Village, was still widely maintained.

In **Figure 5B**, the horizontal diversity of *pekarangan*s in the two research locations was also not too different. Ornamental plants still dominated among the eight functions of *pekarangan* plants. It was in accordance with the results of research Ortiz-Sanchez et al. (2015) and Irwan and Sarwadi, (2017) which stated that the home garden was dominated by ornamental plants. Ornamental plants in non-migrant *pekarangan*s were higher than migrants due to urbanization. The urbanized *pekarangan*s were dominated by ornamental plant species (Ali et al., 2021). *Acalypha siamensis* (wild tea) as an ornamental plant and *Musa paradisiaca* (banana) as a fruit plant were most commonly found in migrant and non-migrant *pekarangans*. This was in accordance with the conditions of rural *pekarangans* which were widely planted with ornamental and fruit plants (Mathewos et al., 2018; ElfridaMubarak and Suwardi, 2020) *Curcuma longa* Linn. (curcuma) as a spice plant was most commonly found in migrant and non-migrant *pekarangans* as well. In rural areas, there are still many people who grow plants for spices and seasoning in the *pekarangan* (Zuberi et al., 2014; Villa and García, 2017). Plants with other functions were more commonly found in migrant *pekarangans*. Plants with other functions are plants that function other than for food (Arifin et al., 1998), such as land boundary marker plants (*Cordyline fruticose* L. and *Dracaena fragrans* L.) (Werdiningsih, 2007) and fodder plants (*Pennisetum purpureum*), which were often found in rural *pekarangans* Schumach (Ivanova et al., 2021).

Correlation Between *Pekarangan* Size and the Number of Individual Plants

The effect of the *pekarangan* area was analyzed by simple linear regression and resulted in the effect of *pekarangan* area on the number of individual plants. In the migrant *pekarangan* (**Figure 6**), it can be seen that the regression graph showed a positive linear line, where upon the addition of 0.3905 m of the *pekarangan* area, the number of individual plants will be added. However, the concerned variable in this analysis is the x-value (increase in the area of the *pekarangan*). The r value (0.69) of the migrant *pekarangan* was in the category of strong correlation (Ridwan Aldila Melania Care et al., 2018), with a significant value = 0.000 < 0.05. The 48.02% of the number of plants can be explained by the influence of the *pekarangan* area and the other influences came from outside variables of the analysis.

In the non-migrant *pekarangan* (**Figure 6**), it can be seen that the regression graph also showed a positive linear line, where upon the addition of every 1 m^2 of the *pekarangan* area, the individual plants will increase by 0.4205. The r value (0.88) of non-migrant *pekarangan*s was in the category of strong correlation (Ridwan Aldila Melania Care et al., 2018) with a significant value = 0.00 < 0.05. There are 77.77% of the number of plants that can be explained by the influence of the *pekarangan* area and the other influences also came from outside variables of the analysis. From the regression graph, it was found that the area of the non-migrant *pekarangan* was very strongly correlated with the number of plants in the *pekarangan*.

It can be seen that the area of the *pekarangan* had strong to very strong correlations (0.69–0.88) with the number of individual plants. Although migrant and non-migrant *pekarangans* have different areas, they have a strong relationship with plant diversity. The larger the *pekarangan*, the greater is the diversity of the plants. It was different from the findings of Antoh et al. (2019) on their *pekarangan* research in Arguni Bawah, West Papua Province, where they found a large *pekarangan* with low diversity of plants (the correlation was positive, but weak). It was proven by the comparison of plant diversity (number of individual plants) per 100 m^2 of the *pekarangan* area. Adjustments were made to the area of the *pekarangan* owned. The large *pekarangan* was planted with a large number of individuals, although the variety of species was little. The small *pekarangan* was being planted with a large variety of species but the number of individuals/species was the same as the large *pekarangan* as well per 100 m^2. Wherever the Sundanese

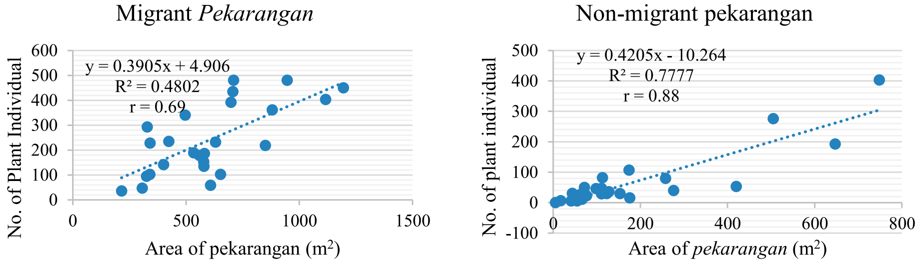


FIGURE 6 | Influence of the *pekarangan* size and the number of individual plants of migrant and non-migrant *pekarangans*.

lived, whether the size of *pekarangan* was large or small, the plant diversity of the *pekarangan* was high. It was due to their habit and culture. Mazumdar and Mazumdar (2012) stated that there was a functional value between the garden (plant diversity) and the family and culture.

Differences in conditions of migrant and Sundanese non-migrant *pekarangans* can be seen from the different types of plants planted. In migrants' *pekarangan*, plants from food types dominated, while in non-migrants' *pekarangan*, plants with ornamental functions dominated. This could be due to urbanization factors that are more influential on the Island of Java so that the dominant types of plants planted were also different. It must be improved so that the use of food plants also dominated in addition to the ornamental plants in small-medium sizes of the *pekarangan*.

Improving *Pekarangan* Conditions

All types of *pekarangan* sizes can display plant diversity. However, in non-migrants' *pekarangan*, the use of plants for food was still less when compared to migrants' *pekarangan*, so that the selection of multifunctional vegetation types can be suggested. In migrants' *pekarangan*, the potential for the loss of *pekarangan* area in the future was quite high due to development and urbanizations. Therefore, steps are needed in the utilization *pekarangan* so that the area can be maintained. So we made some considerations that can be applied to improve the condition of the *pekarangan*, both for migrant and non-migrant and other types of *pekarangans*.

First: a large *pekarangan* can be planted with various types of plant species in various functions. These can fulfill the criteria for the existence of vertical and horizontal diversities of plants. The valuable and important plants can be planted in the *pekarangan*, such as commercial crops (Abdoellah et al., 2020). Based on experience, the area of the *pekarangan* can exist if there was something valuable in it.

Second: for non-migrant *pekarangans*, a small *pekarangan* area was not a problem to display a shady and green *pekarangan*. Currently, there are many farming systems that do not require a lot of land for gardening, such as vertical gardens (do Valle Santos et al., 2019), hydroponic systems (Lal et al., 2020; Solis-Topanta et al., 2020), fish farming in buckets, hanging gardens, rooftop gardens, and planter boxes (Lal et al., 2020). These systems can be applied in the *pekarangan* to grow mainly vegetables, medicine, herbs, fruits, starch-producing shrubs, or herbs (Azra et al., 2014; Jesica et al., 2019), and wherever possible annual plants are chosen to be more sustainable. Although the area of the *pekarangan* was small, the diversity of species and individual plants remains high. Sundanese, who were attached to the culture of eating *lalap* (raw or boiled vegetables) should maintain this habit (way) (Septiani et al., 2020). They can grow various types of vegetables, apart from shrubs or herbs, but also from tree species, for example, *petai* (*Parkia speciosa* Hassk.), *jengkol* (*Pithecellobium jiringa* (Jack) Prain), *melinjo* leaf (*Gnetum gnemon* L.), *moringa* leaf (*Moringa oleifera* lamk.), and cashew leaf (*Anacardium occidental* L.). Therefore, the vertical diversity function existed too. In addition, high trees can also be a shelter, climate amelioration, windbreak (Turner-Skoff and Cavender, 2019), carbon sequestration (Mattsson et al., 2015), and as a place to live for wild animals (Turner-Skoff and Cavender, 2019). This would very well be applied on a regional scale where the owners of the *pekarangan* can form a community. It was widely evident in urban and sub-urban areas where the *pekarangan* size was small, and it gave a positive perception for urban communities (Grebitus et al., 2020; Wood et al., 2020). There were many related government programs from the Ministry of Agriculture *via* National Food Agency (BPN) that could be a way out in funding for *pekarangan* revitalization.

Third: to achieve a good condition, sustainability of the *pekarangan* did not only depend on improving the biophysical condition (*pekarangan* area, plant diversity) but also the positive role of economic, social, and cultural aspects (Mazumdar and Mazumdar, 2012; Antoh et al., 2019). As long as the *pekarangan* had these roles for its owner, its existence and biophysical condition will be good and sustainable. Therefore, the owner's preference in managing and utilizing *pekarangan* was an important factor to be considered because these three aspects were highly dependent on it. Therefore, campaigns on the need to maintain and improve biophysical conditions must continue to be carried out by the government and non-governmental organizations (NGOs) to create higher public awareness. Environmental awareness can be effectively carried out by social media, and it made a positive impact for the environment (Ragusa and Crampton, 2017; Kuppuswamy, 2018). Several activities and programs by both the government and NGOs continued to grow, especially those related to the use of *pekarangans*. The existing government programs are "Sustainable Food from *Pekarangan*" (P2L), Family Farming (PK), Creative Village Development, and Local Food Diversification (Asmoro et al., 2020; Food Security Agency, 2020). The examples from NGOs are Community of Indonesian *Pekarangan* and Productive Garden (KPKPID), Bogor Gardening, and other relevant communities. The main principle of these programs was how the *pekarangan* can be used as productively as possible. Group members can exchange ideas and experiences in utilizing the *pekarangan*. In addition, community members can also share their seeds, seedlings, and crops. In the future, the *pekarangan* can be one of the potential tourism destinations that will be diverse and interesting.

CONCLUSION

Biophysical conditions of migrant and non-migrant *pekarangans* differed in area and size but not so much in terms of plant diversity. Both have good plant diversity but differ in the dominant types of vegetation planted in each type of the *pekarangan*. In migrants' *pekarangan*, the dominant individual plants came from food plants, while in non-migrants' *pekarangan*, ornamental plants were dominated. The criteria for correlation between the *pekarangan* area and its plant diversity in migrant and non-migrant *pekarangans* are strong and positive. It indicated that even though the area of *pekarangan* was different, it still had the same high diversity of plants. Planting important and valuable crops was an option to maintain the *pekarangan* area to still exist, especially in migrants' *pekarangan*. The selection of plant species that had a variety of functions can be an option for small and medium size in non-migrants' *pekarangan*. The improvement of biophysical conditions of the *pekarangan* must also be accompanied by the improvement of the economic, social, and cultural aspects with awareness actions on the importance of using the *pekarangan*.

DATA AVAILABILITY STATEMENT

The original contributions presented in the study are included in the article/supplementary material; further inquiries can be directed to the corresponding author.

AUTHOR CONTRIBUTIONS

MSA developed ideas and wrote down the contents of the manuscript. HSA contributed as the supervisor, corrected the content along with corresponding authors. NA contributed as the supervisor and corrector. MA contributed as the supervisor and corrector.

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REFERENCES

Abdoellah, O. S., Schneider, M., Nugraha, L. M., Suparman, Y., Voletta, C. T., Withaningsih, S., et al. (2020). Homegarden Commercialization: Extent, Household Characteristics, and Effect on Food Security and Food Sovereignty in Rural Indonesia. *Sustain. Sci.* 15 (3), 797–815. doi:10.1007/s11625-020-00788-9

Abebe, T., Wiersum, K. F., and Bongers, F. (2010). Spatial and Temporal Variation in Crop Diversity in Agroforestry Homegardens of Southern Ethiopia. *Agroforest Syst.* 78, 309–322. doi:10.1007/s10457-009-9246-6

Ali, M. S., Arifin, H. S., and Arifin, N. H. S. (2020). The Dynamic of Pekarangan Selahuni 2 Homlet, Ciomas Rahayu Village, Bogor. *J. Nat. Resour. Environ. Manag.* 10 (3), 364–373. doi:10.29244/jpsl.10.3.364-373

Ali, M. S., Arifin, H. S., and Nurhayati, N. (2021). Urbanization Level and its Effect on the Structure and Function of Homegarden (*Pekarangan*) Vegetation in West Java, Indonesia. *Biodiversitas* 22 (1), 173–183. doi:10.13057/biodiv/d220123

Antoh, A. A., Arifin, N. H., Chozin, M. A., and Arifin, H. S. (2019). Short Communication: Agricultural Biodiversity and Economic Productivity of the Yards in Arguni Bawah, Kaimana District, West Papua Province, Indonesia. *Biodiversitas* 20 (4), 1020–1026. doi:10.13057/biodiv/d200412

Arifin, H. S., Arifin, N. H. S., and Munandar, A. (2010). *Pemanfaatan* Pekarangan *di Perdesaan*. Jakarta: Kementerian Pertanian RI.

Arifin, H. S., Munandar, A., SchultinK, G., and Kaswanto, R. L. (2012). The Role and Impacts of Small-Scale, Homestead Agroforestry Systems ("pekarangan") on Household Prosperity: an Analysis of Agro-Ecological Zones of Java, Indonesia. *Int. J. AgriScience.* 2 (10), 896–914. Available at: http://www.inacj.com/attachments/section/17/Temp October 2012-602 Schultink 3 C P F (896-914).pdf.

Arifin, H. S., Nurhayati, Kaswanto, Budiadi, Irwan, S. N. R., Faisal, B., et al. (2021). Landscape Management Strategy of Pekarangan to Increase Community Immunity During The Covid-19 Pandemic in Java Indonesia – Inductive Research. *IOP Conf. Ser.: Earth Environ. Sci.* 918, 012029. doi:10.1088/1755-1315/918/1/012029

Arifin, H. S., Sakamoto, K., and Chiba, K. (1998). Effects of Urbanization on the Vegetation Structure of Home Gardens in West Java, Indonesia. Japanese. *J. Trop. Agric.* 42 (2), 94–102. doi:10.11248/jsta1957.42.94

Arifin, H. S., Sakamoto, K., and Chiba, K. (1997). Vegetation in the Home Garden "*pekarangan*" in West Java, Indonesia. *Bull. Int. Assoc. Landsc. Ecol.* 3 (3), 38–40.

Asmoro, B. T., Utomo, A. K., Chasanah, M., and Al Mahbubi, N. M. (2020). Pendampingan Program Kawasan Rumah Pangan Lestari (KRPL) di Perumahan Warakawuri Seroja RT 24 Desa Sengguruh Kecamatan Kepanjen Kabupaten Malang. At-tamkin. *J. Pengabdi. Kpd. Masy.* 3 (2), 64–70.

Azra, A. L. Z., Arifin, H. S., Astawan, M., and Arifin, N. H. S. (2014). Analisis karakteristik *pekarangan* dalam mendukung penganekaragaman pangan keluarga di Kabupaten Bogor. *J. Lanskap. Indones.* 6 (2), 1–12. doi:10.29244/jli.2014.6.2.1-12

Buck, D. (2016). *Gardens and Health: Implications for Policy and Practice*. England: The Kings Fund.

Budiastuti, S., Purnomo, D., Supriyono, M. B., Yunindanova, M. B., Mahardini, P. C. A., and Utami, R. R. (2018). Land Management Strategy for Cocoa Cultivation at Home Gardens. *IOP Conf. Ser. Earth Environ. Sci.* 200, 012005. doi:10.1088/1755-1315/200/1/012005

Caballero-Serrano, V., McLaren, B., Carrasco, J. C., Alday, J. G., Fiallos, L., Amigo, J., et al. (2019). Traditional Ecological Knowledge and Medicinal Plant Diversity in Ecuadorian Amazon Home Gardens. *Glob. Ecol. Conservation* 17, e00524. doi:10.1016/j.gecco.2019.e00524

Caballero-Serrano, V., Onaindia, M., Alday, J. G., Caballero, D., Carrasco, J. C., McLaren, B., et al. (2016). Plant Diversity and Ecosystem Services in Amazonian Homegardens of Ecuador. *Agric. Ecosyst. Environ.* 225, 116–125. doi:10.1016/j.agee.2016.04.005

Chatterjee, R., Choudhuri, P., Chowdhury, R. S., and Thirumdasu, R. K. (2017). Diversity of Vegetable Crops in Home Gardens of Sub Himalayan Districts of West Bengal, India. *NDP. J. Horti. Plant. Sci.* 1 (1), 9–16.

Clatworthy, J., Hinds, J., and M. Camic, P. (2013). Gardening as a Mental Health Intervention: a Review. *Ment. Health Rev. J.* 18 (4), 214–225. doi:10.1108/MHRJ-02-2013-0007

Coolen, H., and Meesters, J. (2012). Private and Public Green Spaces: Meaningful but Different Settings. *J Hous Built Environ* 27, 49–67. doi:10.1007/s10901-011-9246-5

Corley, J., Okely, J. A., Taylor, A. M., Page, D., Welstead, M., Skarabela, B., et al. (2021). Home Garden Use during COVID-19: Associations with Physical and Mental Wellbeing in Older Adults. *J. Environ. Psychol.* 73, 101545. doi:10.1016/j.jenvp.2020.101545

do Valle Santos, W. C., Singh, D., Delgado Leandro da Cruz, L., de Carvalho Piassi, L. P., and Reis, G. (2019). Vertical Gardens: Sustainability, Youth Participation, and the Promotion of Change in a Socio-Economically Vulnerable Community in Brazil. *Educ. Sci.* 9 (3), 161. doi:10.3390/educsci9030161

Drescher, A. W., Holmer, R. J., and Iaquinta, D. L. (2006). *Urban Homegardens and Allotment Gardens for Sustainable Livelihoods: Management Strategies and Institutional Environments*. Editors B. Kumar, and P. Nair (Dordrecht, DE: Springer Dordrecht), 317–338. doi:10.1007/978-1-4020-4948-4_18

ElfridaMubarak, E., Mubarak, A., and Suwardi, A. B. (2020). Short Communication: The Fruit Plant Species Diversity in the Home Gardens and Their Contribution to the Livelihood of Communities in Rural Area. *Biodiversitas* 21 (8), 3670–3675. doi:10.13057/biodiv/d210833

Feanside, P. M. (1997). Transmigration in Indonesia: Lessons from its Environmental and Social Impacts. *Environ. Manag.* 21, 553–570. doi:10.1007/s002679900049

Food Security Agency (2020). *Technical Guidance of 2020 Family Farming*. Jakarta: Food Security Agency. Ministry of Agriculture of Republic of Indonesia.

Galluzzi, G., Eyzaguirre, P., and Negri, V. (2010). Home Gardens: Neglected Hotspots of Agro-Biodiversity and Cultural Diversity. *Biodivers. Conserv.* 19 (13), 3635–3654. doi:10.1007/s10531-010-9919-5

Gbedomon, R. C., Salako, V. K., Fandohan, A. B., Idohou, A. F. R., Glèlè Kakaï, R., and Assogbadjo, A. E. (2017). Functional Diversity of Home Gardens and Their Agrobiodiversity Conservation Benefits in Benin, West Africa. *J. Ethnobiol. Ethnomedicine* 13 (1), 66. doi:10.1186/s13002-017-0192-5

Grebitus, C., Chenarides, L., Muenich, R., and Mahalov, A. (2020). Consumers' Perception of Urban Farming-An Exploratory Study. *Front. Sustain. Food Syst.* 4, 79. doi:10.3389/fsufs.2020.00079

Hakim, L. (2014). *Etnobotani dan Manajemen Kebun-Pekarangan Rumah: Ketahanan Pangan, kesehatan, dan Agrowisata*. Malang: Penerbit Selaras. Available at: https://biologi.ub.ac.id/wp-content/uploads/2015/11/ETNOBOTANI-dan-MANAJEMEN-KEBUN-PEKARANGAN-RUMAH.pdf.

Idohou, R., Fandohan, B., Salako, V. K., Kassa, B., Gbèdomon, R. C., Yédomonhan, H., et al. (2014). Biodiversity Conservation in Home Gardens: Traditional Knowledge, Use Patterns and Implications for Management. *Int. J. Biodivers. Sci. Ecosyst. Serv. Manag.* 10 (2), 89–100. doi:10.1080/21513732.2014.910554

Indrawan, M., Yabe, M., Nomura, H., and Harrison, R. (2014). Deconstructing Satoyama - the Socio-Ecological Landscape in Japan. *Ecol. Eng.* 64, 77–84. doi:10.1016/j.ecoleng.2013.12.038

Irwan, S. N. R., and Sarwadi, A. (2017). Productive Urban Landscape in Developing Home Garden in Yogyakarta City. *IOP Conf. Ser. Earth Environ. Sci.* 91, 012006–012007. doi:10.1088/1755-1315/91/1/012006

Ivanova, T., Bosseva, Y., Chervenkov, M., and Dimitrova, D. (2021). Enough to Feed Ourselves!-Food Plants in Bulgarian Rural Home Gardens. *Plants* 10, 2520. doi:10.3390/plants10112520

Jesica, J., Kaswanto, R. L., and Arifin, H. S. (2019). Management of "*Pekarangan*" in Informal Settlement of Ciliwung River Riparian Landscape. *KnE Soc. Sci.*, 408–420. doi:10.18502/kss.v3i21.4984

Kaur, P., Stoltzfus, J., and Yellapu, V. (2018). Descriptive Statistics. *Int. J. Acad. Med.* 4 (1), 60–63. doi:10.4103/IJAM.IJAM_7_18

Kehlenbeck, K., Arifin, H. S., and Maass, B. L. (2007). *Plant Diversity in Homegardens in a Socio-Economic and Agro-Ecological Context.* Editors T. Tscharntke, C. Leuschner, M. Zeller, E. Guhardja, and A. Bidin (Berlin, DE: Springer Berlin, Heidelberg), 295–317.

Kehlenbeck, K., and Maass, B. L. (2006). *Are Tropical Homegardens Sustainable? Some Evidence from Central Sulawesi, Indonesia*. Editors B. Kumar, and P. Nair (Dordrecht, DE: Springer Dordrecht), 339–354. doi:10.1007/978-1-4020-4948-4_19

Kertawibawa, B. B., and Harun, I. B. (2012). Migration and Changing Desa-Kota Interaction. *Procedia - Soc. Behav. Sci.* 50 (July), 55–68. doi:10.1016/j.sbspro.2012.08.015

Khoiriyah, F., Fahri, A., Bramantio, B., and Sumargono, S. (2019). Sejarah Toponimi Daerah Transmigrasi Provinsi Lampung Melalui Tuturan Tradisi Lisan. *Agastya J. Sej. Dan. Pembelajarannya* 9 (2), 221–240. doi:10.25273/ajsp.v9i2.4419

Kiesling, F. M., and Manning, C. M. (2010). How Green Is Your Thumb? Environmental Gardening Identity and Ecological Gardening Practices. *J. Environ. Psychol.* 30 (3), 315–327. doi:10.1016/j.jenvp.2010.02.004

Kuppuswamy, S. (2018). A Study on the Environmental Campaigns in Traditional and Social Media. *Int. J. E-Politics.* 9 (1), 29–47. doi:10.4018/IJEP.2018010103

Lal, R. (2020). Home Gardening and Urban Agriculture for Advancing Food and Nutritional Security in Response to the COVID-19 Pandemic. *Food Sec.* 12, 871–876. doi:10.1007/s12571-020-01058-3Mattsson

Mathewos, M., Hundera, K., and Biber-Freudenberger, L. (2018). Planting Fruits and Vegetables in Homegarden as a Way to Improve Livelihoods and Conserve Plant Biodiversity. *Agriculture* 8 (12), 190. doi:10.3390/agriculture8120190

Mattsson, E., Ostwald, M., Nissanka, S. P., and Pushpakumara, D. K. N. G. (2015). Quantification of Carbon Stock and Tree Diversity of Homegardens in a Dry Zone Area of Monaragala District, Sri Lanka. *Agroforest. Syst.* 89, 435–445. doi:10.1007/s10457-014-9780-8

Mazumdar, S., and Mazumdar, S. (2012). Immigrant Home Gardens: Places of Religion, Culture, Ecology, and Family. *Landsc. Urban Plan.* 105 (3), 258–265. doi:10.1016/j.landurbplan.2011.12.020

Mohri, H., Lahoti, S., Saito, O., Mahalingam, A., Gunatilleke, N., Irham, et al. (2013). Assessment of Ecosystem Services in Homegarden Systems in Indonesia, Sri Lanka, and Vietnam. *Ecosyst. Serv.* 5, 124–136. doi:10.1016/j.ecoser.2013.07.006

Montefrio, M. J. F. (2020). Interrogating The "Productive" Home Gardener In A Time Of Pandemic Lockdown In The Philippines. *Food and Foodways.* 28 (3), 216–225. doi:10.1080/07409710.2020.1790142

Mollison, B. (1979). *Permaculture Two: Practical Design for Town and Country in Permanent Agriculture*. Australia: TAGARI.

Mumaw, L., and Bekessy, S. (2017). Wildlife Gardening for Collaborative Public-Private Biodiversity Conservation. *Australas. J. Environ. Manag.* 24, 242–260. doi:10.1080/14486563.2017.1309695

Nelson, D. (2009). Using Simple Linear Regression to Assess the Success of the Montreal Protocol in Reducing Atmospheric Chlorofluorocarbons. *J. Statistics Educ.* 17, 2. doi:10.1080/10691898.2009.11889520

Nugraha, A. T., Hasan, S., and Samantha, Y. (2015). Model Pengembangan Kawasan Transmigran, Kecamatan Waplau, Kabupaten Buru, Provinsi Maluku. *J. Agribisnis* 9 (1), 23–36. doi:10.15408/aj.v9i1.5071

Nyhus, P., and Sumianto, T. R. (1999). *Riding the Tiger: Tiger Conservation in Human-Dominated Landscapes*. Cambridge: Cambridge University Press.

Ortíz-Sánchez, A., Monroy-ortiz, C., Romero-Manzanarez, A., Luna-cavazos, M., and Castillo-España, P. (2015). Multipurpose Function of Home Gardens in the Family Subsistence. *Bot. Sci.* 93 (4), 791–806. doi:10.17129/botsci.224

Peng, Y., Fan, M., Song, J., Cui, T., and Li, R. (2018). Assessment of Plant Species Diversity Based on Hyperspectral Indices at a Fine Scale. *Sci. Rep.* 8, 1–11. doi:10.1038/s41598-018-23136-5

Prihatin, R. B. (2013). Revitalisasi Program Transmigrasi. *Aspirasi* 4 (1), 57–64.

Prihatini, J., Iskandar, J., Partasasmita, R., and Nurjaman, D. (2018). The Impacts of Traditional Homegarden Conversion into the Commercial One: A Case Study in Sukapura Village of the Upstream Citarum Watershed, West Java, Indonesia. *Biodiversitas* 19 (5), 1926–1940. doi:10.13057/biodiv/d190545

Ragusa, A., and Crampton, A. (2017). Environmental Campaign Awareness, Participation, and Media Visibility. *Int. J. Sustain. Econ. Soc. Cult. Context* 13 (2), 51–66. doi:10.18848/2325-1115/CGP/v13i02/51-66

Ranangonggono, P. (2012). Pelaksanaan intensifikasi lahan *pekarangan* transmigran di unit pemukiman transmigrasi IV SP-6 Alue Peunyareng; analisis kebijakan berdasarkan Keputusan Menteri Transmigrasi dan Pemukiman Perambah Hutan RI Nomor: KEP 147. master's thesis. Indonesia: Universitas Terbuka. MEN/1994. [Indonesia (ID)].

Ricklefs, C. M. (2008). *A History of Modern Indonesia*. New York: Palgrave Macmillan.

Ridwan Aldila Melania Care, F., Sugeng Subagio, B., and Rahman, H. (2018). Porous Concrete Basic Property Criteria as Rigid Pavement Base Layer in

Indonesia. *MATEC Web Conf.* 147 (8), 02008. doi:10.1051/matecconf/201814702008

Santoro, A., Venturi, M., Bertani, R., and Agnoletti, M. (2020). A Review of the Role of Forests and Agroforestry Systems in the Fao Globally Important Agricultural Heritage Systems (GIAHS) Programme. *Forests* 11 (8), 860. doi:10.3390/F11080860

Septiani, N., Hernawati, D., and Putra, R. R. (2020). Biodiversity of Potentially "Lalapan" Vegetables in Kampung Adat Naga, Tasikmalaya, Indonesia. *Biosfer* 13 (2), 201–2015. doi:10.21009/biosferjpb.v13n2.201-215

Seto, K. C. (2011). Exploring the Dynamics of Migration to Mega-Delta Cities in Asia and Africa: Contemporary Drivers and Future Scenarios. *Glob. Environ. Change* 21 (1), S94–S107. doi:10.1016/j.gloenvcha.2011.08.005

Sofo, A., and Sofo, A. (2020). Converting Home Spaces into Food Gardens at the Time of Covid-19 Quarantine: All the Benefits of Plants in This Difficult and Unprecedented Period. *Hum. Ecol.* 48 (2020), 131–139. doi:10.1007/s10745-020-00147-3

Soga, M., Gaston, K. J., and Yamaura, Y. (2017). Gardening Is Beneficial for Health: A Meta-Analysis. *Prev. Med. Rep.* 5, 92–99. doi:10.1016/j.pmedr.2016.11.007

Solis-Toapanta, E., Kirilenko, A., and Gómez, C. (2020). Indoor Gardening with Hydroponics: A Reddit Community Analysis to Identify Knowledge Gaps. *HortTechnology* 30 (3), 346–355. doi:10.21273/HORTTECH04574-20

Thamilini, J., Wekumbura, C., Mohotti, A. J., Kumara, A. P., Kudagammana, S. T., Silva, K. D. R. R., et al. (2019). Organized Homegardens Contribute to Micronutrient Intakes and Dietary Diversity of Rural Households in Sri Lanka. *Front. Sustain. Food Syst.* 3, 94. doi:10.3389/fsufs.2019.00094

Titus, M. J. (1992). Transmigration and Regional Development in Indonesia: Policy Options between Myth and Reality. *Populasi* 1 (3), 1992.

Turner-Skoff, J. B., and Cavender, N. (2019). The Benefits of Trees for Livable and Sustainable Communities. *Plants People Planet* 1, 323–335. doi:10.1002/ppp3.39

van der Stege, C., Vogl-Lukasser, B., and Vogl, C. R. (2010). *The Role of Homegardens in Strengthening Social-Ecological Resilience: Case Studies from Cuba and Austria*. Editors T. Plieninger, and C. Bieling (London, UK: Cambridge University Press), 261–282.

Villa, D., and García, N. (2017). Food Plants in Home Gardens of the Middle Magdalena Basin of Colombia. *Caldasia* 39 (2), 292–309. doi:10.15446/caldasia.v39n2.63661

Webb, E. L., and Kabir, M. E. (2009). Home Gardening for Tropical Biodiversity Conservation. *Conserv. Biol.* 23 (6), 1641–1644. doi:10.1111/j.1523-1739.2009.01267.x

Werdiningsih, H. (2007). Kajian Penggunaan Tanaman Sebagai Alternatif Pagar Rumah. *Enclosure*. 6 (1), 32–39.

Wood, J., Wong, C., and Paturi, S. (2020). Vertical Farming: An Assessment of Singapore City. *eTropic* 19 (2), 228–248. doi:10.25120/etropic.19.2.2020.3745

Yustha, Y. (2017). Pemanfaatan Lahan Pekarangan dengan Sistem Agroforestri oleh Masyarakat di Desa Sidomulyo, Kapuas Kuala, Kapuas. *Agrisilvika* 1 (1), 1–5. doi:10.31227/osf.io/h79e2

Zuberi, M. I., Kebede, B., Gosaye, T., and Belachew, O. (2014). Species of Herbal Spices Grown in the Poor Farmers' Home Gardens of West Shoa, Highlands of Ethiopia: an Ethnobotanical Account. *J. Bio. & Env. Sci.* 4 (4), 164–185.

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Carbon stocks and footprints of smallholder cacao systems in Polewali Mandar, West Sulawesi

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Cacao (*Theobroma cacao*) is a commodity that plays an important role in supporting economic and social development. However, cacao production can also be a major contributor to carbon emissions, which has stimulated various efforts toward sustainable cacao farm management. There remains a gap in knowledge regarding the links between carbon stocks and carbon footprints, which can serve as indicators of environment "friendliness." In this study, we investigated carbon stocks and carbon footprints in two cacao cultivation systems, agroforests and monocultural systems, and the biophysical aspects (biotic and abiotic factors) that might contribute to the variability of carbon levels. System inventories, soil samples, and farmer interviews identified the characteristics and management practices of two cacao production systems. Results show that cacao agroforests accumulated more carbon stocks than cacao monocultures, 134.4 Mg C ha^{-1} and 104.7 Mg C ha^{-1}, respectively, while cacao monocultural systems had higher carbon footprints than cacao agroforests, 1914.4 kg CO_2e ha^{-1} and 932.1 ± 251.6 kg CO_2e ha^{-1}, respectively. Canopy cover, tree density, and soil organic carbon were the biophysical aspects that showed a significantly positive correlation with carbon stock levels, while canopy cover had a significantly negative correlation with carbon footprint levels. These results suggest that cacao agroforests are more climate-friendly management systems due to their ability to maintain high carbon stock levels while producing low carbon footprints.

KEYWORDS

cacao agroforest, climate friendliness, green intensification, on-farm biodiversity, sustainable livelihood

1 Introduction

Cacao (*Theobroma cacao*) is a very popular cash crop and is a main agricultural commodity in many tropical countries (Hartemink, 2005). Cacao cultivation has become widespread in humid and sub-humid tropics and plays an important role in social and economic development (Pohlan and Pérez, 2010). In Indonesia, cacao is a priority crop, contributing significantly to the national economy as the third most important export commodity after oil palm (*Elaeis guineensis*) and rubber (*Hevea brasiliensis*). In all cacao-producing regions of Indonesia, agroforest systems are cultivated and managed by smallholders. Cacao agroforest (CAF) systems are characterized by their tree component. In the understory, the agroforest systems are dominated by cacao with the upper canopy comprised of a variety of tree species including forest remnants, secondary growth, planted timber, fruit, and nitrogen-fixing species. Food-producing tree species are commonly planted in smallholder cacao systems, with two-thirds of these trees being native forest species. Cacao trees can also be intercropped with other cash or food crops. Food crops, like maize (*Zea mays*), sweet potato (*Ipomoea batatas*), malanga (*Xanthosoma spp.*), and cucumber (*Cucumis sativus*), are often associated with cacao during the early years of its growth (Atangana et al., 2014).

The high diversity of tree species implies that CAF systems can be more environmentally friendly in terms of carbon (C) storage (Schroth et al., 2015). Environment friendliness is one indicator of sustainability related to the impact of commodity production on the climate. There are two measures of environmental friendliness that are discussed among scientists, C footprints and the standing C stocks. A C footprint is the amount of greenhouse gases (GHGs) emitted from the production of a commodity. Agricultural inputs, such as fertilizer, pesticides, and fossil fuels, as well as on-farm processing, are common variables that influence the level of the C footprints (Hillier et al., 2011; van Rikxoort et al., 2014). Another factor, which has the capacity to restrict and even decrease the size of the C footprint, is the C stock (Schroth et al., 2015). Carbon stocks are the total C sequestered in the C pools (in $Mg\,C\,ha^{-1}$) of a production system. The carbon stock of a land use system is considered an indicator of climate friendliness, mitigating global heating (van Rikxoort et al., 2014).

Cacao agroforests, with a diversity of shade trees and shade cover, often have high C stocks and low C footprints. A number of tree species of the remnant forest are maintained as shade cover on cacao farms, allowing higher biodiversity and providing continuous ecosystem services. A recent study found that various levels of shade cover and biodiversity can enhance the C stock by increasing the biomass (Dawoe et al., 2016). Shade cover and biodiversity provide many environmental benefits, such as natural pest control (Wielgoss et al., 2012) and minimization of soil erosion and nutrient leaching (Rice and Greenberg, 2000); these benefits can lead to a reduced need for agricultural inputs. Compared to these mixed systems, cacao monocultural or "full-sun" systems (cacao systems with limited or no shade trees) have high early production due to the greater density of cacao trees. However, the negative impact of monocultural systems on climate is higher due to the intense use of fertilizers, pesticides, and other chemical inputs (Schneider et al., 2017). High agrochemical inputs produce a greater C footprint and higher GHG emissions (IPCC and penman, 2003).

In this study, we conducted field research on cacao agroforests and monocultural systems with the objectives of 1) measuring the rate of the C stock and C footprint in the two cacao cultivation systems; 2) determining the contribution of biophysical aspects to the C stocks and C footprints; and 3) describing the impact of both systems on environmentally friendly management.

2 Materials and methods

2.1 Research site

This research was conducted in Polewali Mandar district, West Sulawesi Province, Indonesia, which is the main cacao production area in the province. Polewali Mandar is characterized by hills (49.65%) and flat-to-undulating landscapes (44.41%), with some mountainous topography (5.94%). More than half of Polewali Mandar (59.38% of the total area) has an elevation of 12–100 m above sea level, and the remaining area (40.62%) has an elevation of 100–480 m above sea level. A map of the study area is presented in Figure 1.

The climate in Polewali Mandar is classified as tropical humid with two rainy seasons, December–January and April–May. The driest period is August. The mean annual precipitation and temperature between 1982 and 2017 were 2,200 mm and 26.8°C, respectively. The average monthly variation in precipitation and temperature is illustrated in Figure 2.

2.2 Site selection and data collection

Four villages in Polewali Mandar were purposively selected based on cacao production being the main land use and a main source of household income, with both CAF and monocultural systems being practiced and community members being committed to the study. Here, 10 CAF and 10 cacao monocultural farms were randomly selected in each of the four villages. The CAF systems selected in this study were cacao plantations integrated with fruit and timber trees with cacao trees less than 50% of the total tree population. On average, the system featured 10 multi-stratum tree species (range 6–13 species) per farm, such as *Paraserianthes falcataria*, *Gmelina arborea*, *Mangifera indica*, and *Durio zibethinus*, and other commodity species. Meanwhile, the cacao monocultural systems selected in this study were plantations dominated by cacao trees, with only 1–4 shade tree species comprising a small portion of the total system (see Janudianto et al., 2014). In these monocultural systems, shade tree species were dominated by *Gliricidia sepium*, *Leucaena leucocephala*, and *Musa* spp. On the selected farms, interviews were conducted to collect information regarding farm operations: farm history, management practices—including on-farm supply chains (fertilizer, pesticide, fuel use, and all inputs)—yields, and management problems. The interview typically took 2 hours with the actual farm owners. After finishing the interview, field observation was conducted to identify more information related to the farm. A field inventory was implemented on each farm using a sampling plot to collect biomass data. Sampling plots were 20 m by 20 m squares, consisting of core and subplots as recommended by Abou Rajab et al. (2016). The plot design used in this study is presented in Figure 3.

In the core plots, data were collected to estimate above- and below-ground C stocks. The local and botanical names of all trees in the core plots were recorded. Diameter at breast height (DBH) was measured at 130 cm above the ground of all trees with a DBH of more than 5 cm. In the core plot, the necromass of all deadwood and fallen branches was calculated by measuring the length and diameter of each deadwood and branch. Inside the core plot, five subplots were developed. Each subplot was 1 m by 1 m and randomly distributed in the plot. All of the litter and herbaceous plants inside each subplot were collected. "Herbaceous plants" refer to all non-woody and grass plants growing in the understory. Litter and herbaceous plants were weighed and put into plastic (100 g) bags to be analyzed for C stocks. To estimate belowground C stocks, C from the soil and roots was considered. We directly calculated the soil

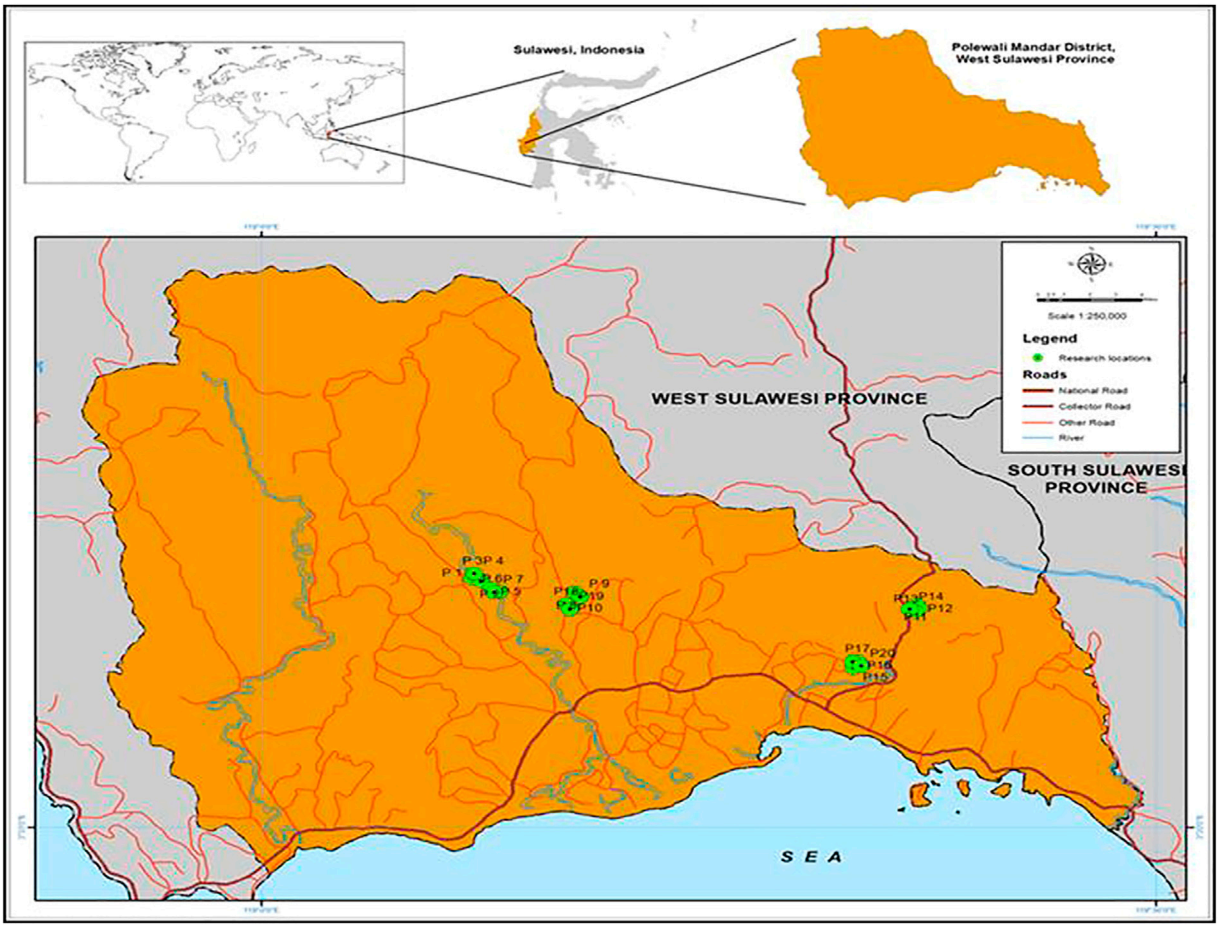

FIGURE 1
Map of the study site.

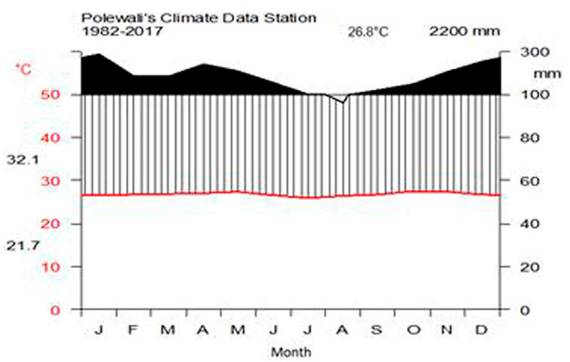

FIGURE 2
Average temperatures and annual precipitation in the study area.

organic content, while the soil inorganic content was excluded. Soil samples were collected from three subplots at depths of 0–10 cm, 10–20 cm, and 20–30 cm. To calculate the soil organic C stock, soil bulk density was considered. The soil and bulk density samples were collected by pressing sample rings into the soil after all herbaceous plants and litter had been removed. Soil samples were analyzed in the laboratory. The root C stock was estimated using a shoot and root ratio. Some biophysical aspects—such as light intensity, shade, soil type and fertility, annual temperatures, and precipitation—were also documented.

2.3 Carbon stock calculations

The C estimations were categorized as C stock and C footprint. The C stock included calculations for each C pool, while the C footprint calculation was based on total emissions from the system. In this study, we considered all C stocks at the farm level. Based on the IPCC and Penman's (2003) study, the C stock is pooled separately into aboveground (tree stand, deadwood, litter, and herbaceous) and belowground biomass (root and soil). Aboveground biomass from the tree stands and necromass were estimated using a non-destructive method and allometric equations (Hairiah et al., 2001). To estimate the tree biomass, we used Yuliasmara et al.'s (2009) method for cacao trees (Eq. 1), Arifin's (2001) method for *Coffea* sp. (Eq. 2) and *Musa* sp. (Eq. 3). For necromass, we used the formula (Eq. 4) developed by Hairiah et al. (2001). In cases where an allometric equation was not available for a

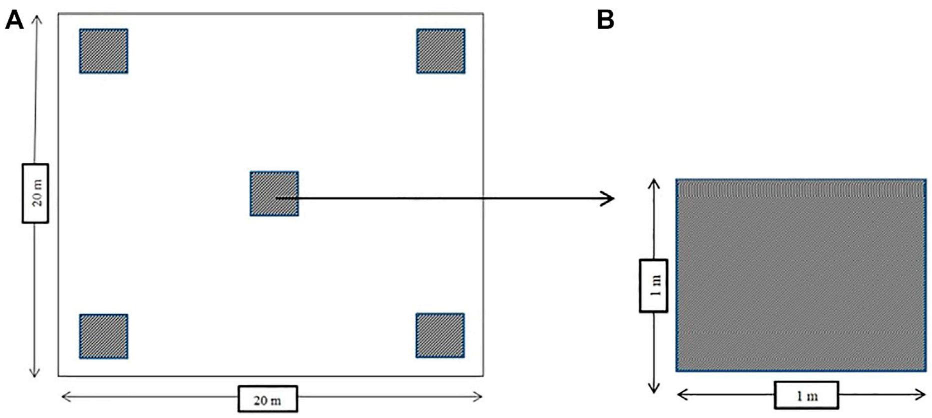

FIGURE 3
Core plot **(A)** and subplot **(B)** design.

specific species, we used the general allometric equation (Eq. 5) developed by Brown (1997).

$$Y = 0.1208.\,D^{1.98}, \quad (1)$$

$$Y = 0.281.D^{2.06}, \quad (2)$$

$$Y = 0.03.\,D^{2.15}, \quad (3)$$

$$Y = \pi.\,D^2.h.s/40, \quad (4)$$

$$ABG = \exp\{-2.134 + 2.530{*}\mathrm{In}\,(DBH)\}. \quad (5)$$

In the equations, Y is the biomass, D is the diameter at breast height (DBH), h is the length of the deadwood, s is the bulk density, and 40 is a constant. ABG is the aboveground biomass. The litter and herbaceous biomass were destructively harvested at the ground level and calculated using the formula developed by Hairiah et al. (2001), i.e.,

$$\text{Biomass } (g) = \frac{\text{Dryweight material } (g) \text{x Freshweight Total } (g)}{\text{Freshweight material } (g)}. \quad (6)$$

Fresh weight for each component of the sample litter and herbaceous plants was measured separately on-site. Then, we randomly selected representative subsamples of stems, branches, leaves, and roots to measure their fresh weight. The subsamples were taken to the laboratory and oven-dried at 75°C to a constant weight. The dry weight (biomass) for each component was calculated according to Eq. 6.

The belowground C stock (soil organic carbon, SOC) was calculated using the Walkley–Black method (Walkley and Black, 1934). The soil sample and bulk density values were used to extrapolate the SOC to a per hectare (Mg C ha^{-1}) value. The formula used was developed by Nair et al. (2009).

$$SOC\ (Mg\,C\,ha^{-1}) = CC * 10000 * SD\ (m) * BD\ (g/cm^3), \quad (7)$$

where CC is the organic carbon content, SD is the soil depth, BD is the bulk density, and 10,000 m^2 is the number of square meters per ha. The belowground C stock in the form of root biomass was calculated using the 25% shoot-to-root ratio as reported by Cairns et al. (1997). Biomass was converted to C stock by assuming a C content of 50% (IPCC and Penman, 2003).

2.4 Carbon footprint calculations

A questionnaire was developed to collect information regarding farm management practices, including the on-farm supply chain (fertilizer, pesticide, and fuel use). Based on the field data collection, the agricultural inputs considered in this study were mineral and organic fertilizer, pesticides, and fossil fuels for farm operations and transportation. All of the data were analyzed using the Cool Farm Tool software version 2.0 beta and converted into C emissions (CO_2e). The software combines several established empirical models for GHG emissions to give a single overall estimate based on current and previous farming practices (Hillier et al., 2011). The model includes several sub-models that break down the overall GHG emissions by farm management practices. GHG emissions from the production and distribution of a range of fertilizer types were taken from the Ecoinvent database (Frischknecht et al., 2007). For nitrous oxide (N_2O) and nitric oxide (NO) emissions from fertilizer application, the study used the multivariate empirical model of Bouwman et al. (2002), which is based on a global dataset of over 800 sites. The formula used was

$$N_2O = e^{\mathrm{const}} + \sum\nolimits_1^{n=1} \text{Factor class}\,(i), \quad (8)$$

where factor classes are fertilizer type x fertilizer application rate, crop type, soil texture, soil organic C, soil drainage, soil pH, soil cation exchange capacity, climate type, and application method. The model for ammonia (NH_3) emissions was slightly different from that given in FAO/IFA (2001),

$$NH_3 = FA.e\sum\nolimits_1^{n=1} \text{Factor class}\,(i), \quad (9)$$

where FA is the amount of fertilizer applied. Factors were determined by statistical analysis. NO and NH_3 emissions were converted to N_2O by the factor 0.01 as given in IPCC (2006). Leaching was assumed to occur at a rate of 0.3 N applied for a moist climate zone only; the conversion factor to N_2O of 0.01 was also employed. Emissions of CO_2 from urea application or liming were also accounted for using IPCC emission factors of 0.20 and 0.12 (IPCC, 2006), respectively. For emissions from pesticide application, the system used sources from the work of Audsley (1997), which produced averages of around 14.7, 18.4, 20.9, and 28.1 kg CO_2 equivalent per hectare for fungicide, growth regulator, herbicide, and insecticide, respectively. We excluded fuel use by machinery because the typical farming system in the study area does not use machinery. Fossil fuel use for transportation and distribution of fertilizer, herbicide, and fungicide was considered by the system and calculated according to the Ecoinvent database.

2.5 Biophysical aspects

During the field inventory, biophysical aspects were analyzed. The biophysical aspects considered in this study were biotic and abiotic factors that might contribute to the level of C. Biophysical data regarding canopy cover, tree diversity, tree density, SOC, applied fertilizer, and light intensity were collected. Canopy cover data were collected using the hemispheric photography method and were collected from 16 positions within each core plot using a digital camera, and then, the data were analyzed using ImageJ software (Ishida, 2004). Tree diversity and density data were collected from the inventory in each plot. Tree diversity data were then analyzed using the Shannon–Wiener (H') index (Smith, 1990). Applied fertilizer data were collected during the interview process with each farmer. Soil organic carbon data were collected using the same process used with the C stock calculation. Light intensity was analyzed using a lux meter (LX-113S) (Fairbairn, 1958).

2.6 Statistical analysis

The data obtained from the C stocks, C footprints, and some of the biophysical variables were analyzed using R statistical software (R studio version 3.4.3). Data exploration was conducted using descriptive statistics, such as minimum and maximum values, mean, range, and even boxplot to observe and view the distribution of the data. The significance of each measured parameter was tested by Student's t-test to perform a pairwise comparison of means. Correlation analysis was also performed to establish trends and relationships between the biophysical aspects, C stocks, and C footprints, as well as the productivity and sustainability of the cacao farm.

3 Results and discussion

3.1 Farm characteristics and biophysical aspects

The cacao plantations in this study were owned by smallholders, with the majority of farm sizes ranging from 0.25–2 ha. There were differences in biophysical aspects and farm characteristics, such as canopy cover, light intensity, tree density, the area under cacao, tree density, mineral fertilizer use, and annual cacao yield, between the monocultural and agroforest systems. The results indicate that variation between farms could have been reduced by including soil parameters, plantation age, and other biophysical and management characteristics as farm selection criteria. Reducing the variation could have revealed significant statistical differences in C storage and C footprints between cacao agroforests and cacao monoculture (CM) farms.

The agroforest systems had a greater proportion of canopy cover (50% and $p < 0.05$) and tree density (1,283 ha^{-1}) compared to the monocultural systems. However, agroforests had a lower level of light intensity (4,833.6), as shown in Table 1. Light intensity was closely related to the canopy cover. The greater the canopy cover, the lower the light intensity. The amount of natural light penetrating the understory is affected by the presence of shade trees (Manaker, 1996). The amount of light reaching the leaves decreases as sunlight passes downward through the canopy; thus, leaves on the upper part of the canopy tend to shade and reflect light away from the lower canopies (Chapman and Carter, 1976). According to Nair (2010), cacao is a species that requires shade, especially in the early phase of growth. The proportion of canopy cover and light intensity may affect the growth and development of cacao trees and, thus, affect yields. The use of shade trees may have a negative impact due to competition for light, water, and nutrients (Asare et al., 2019). However, their advantage in conserving and restoring land can contribute to stable long-term production (Beer et al., 1998).

In this study, cacao tree density and fertilizer application rates had a curious influence on the level of cacao production. Cacao monocultural systems had 9.5% higher yields than agroforest systems, 476 kg ha^{-1} compared to 435 kg ha^{-1}, respectively; however, the difference is not statistically significant. On a per tree basis, cacao yields were similar between the two production systems, 0.61 kg/tree (476 kg/783 trees) in monoculture systems and 0.69 kg/tree in agroforests (435 kg/628 trees)—see Table 1. So, while monocultural systems had more cacao trees per hectare and received higher fertilizer application rates, they did not have significantly higher yields. This implies that nutrient availability did not limit cacao productivity at the study site. Differences in farm characteristics and biophysical aspects between cacao monocultures and agroforests may be more relevant indicators of the sustainability of cacao farming. Long-term stable yields of cacao are not guaranteed by monocultural systems and high rates of fertilizer application. Other studies show that such systems can be more susceptible to pest attack due to the absence of insect and bird predators of those pests and the greater abundance of cacao trees (Bentley et al., 2004; Tscharntke et al., 2011; Wielgoss et al., 2012). Farmers' experience at the study site confirms this condition.

To ensure sustainable yields, the sustainability of ecosystem functions and biophysical aspects need to be considered. Effendy (2015) argued that there are two main elements that determine the productivity of a cacao farm, namely, the provision of nutrients and the management of the cacao trees. The control of cacao tree growth and density, as well as an increase in light penetration, is necessary to promote incremental increases in fruit production (Vernon and Sunderam, 1972). Studies have shown that appropriate shading levels can lead to adequate photosynthetic rates, growth, and profitable fruit yields. Shading can also reduce the effects of unfavorable ecological factors, such as low soil fertility, wind velocity, and excessive evapotranspiration (Miyaji et al., 1997). Multi-stratum plantations can maintain and enhance soil fertility with a subsequent increase in nutrient availability for the cacao crop (Isaac et al., 2007).

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TABLE 1 Farm characteristics and biophysical aspects of cacao farms.

Farm characteristic	CM	CAF
	Mean ± SE	Mean ± SE
Area under cacao (ha)	0.79 ± 0.15[a]	1.10 ± 0.17[a]
Density of tree stands (ha^{-1})	1,200 ± 123[a]	1,283 ± 112[a]
Density of cacao trees (ha^{-1})	783 ± 82[a]	628 ± 84[a]
Density of shade trees (ha^{-1})	418 ± 96	655 ± 64[a]
Density of large trees (ha^{-1})	225 ± 55[a]	335 ± 49[a]
Trees diversity (spc/farm)	3.4 ± 0.5[a]	6.7 ± 0.7[b]
Canopy cover (%)	25 ± 3.1[a]	50 ± 1.7[b]
Light intensity (lux)	4,833 ± 1,554[a]	4,450 ± 793[a]
Applied fertilizer (kg ha^{-1})	328 ± 37[a]	112 ± 31[b]
Annual cocoa yield (kg ha^{-1})	476 ± 58[a]	435 ± 56[a]

*Mean values with different superscript letters within each column denote significant ($p < 0.05$) differences between groups.

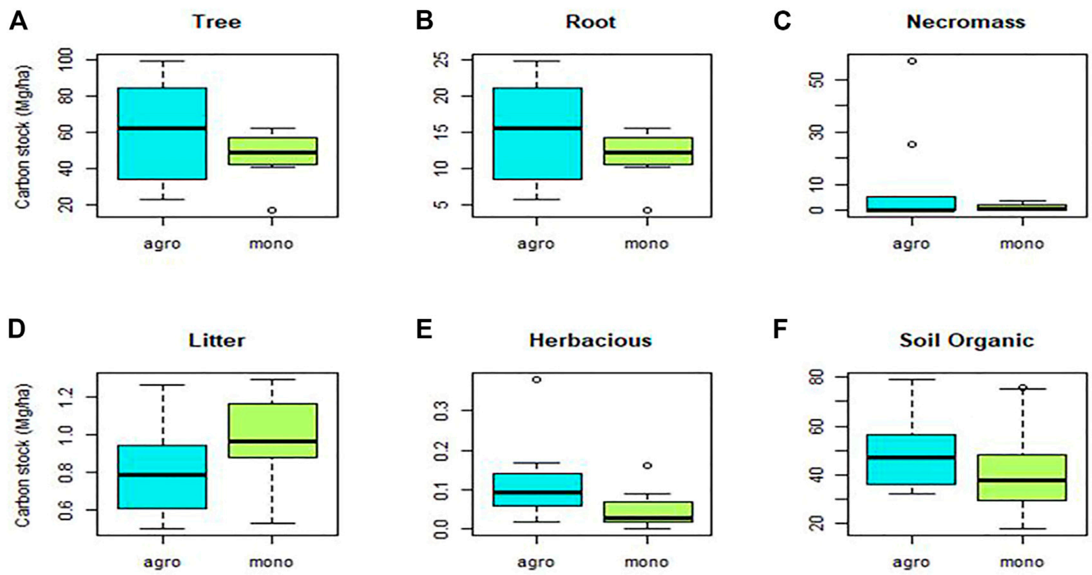

FIGURE 4
Carbon stocks of CM and CAF systems by carbon pool.

Additionally, cacao shade systems provide greater levels of biodiversity and other ecosystem services (food provision, water supply and regulation, plant and animal habitat, genetic diversity, pollination, pest control, and climate regulation), benefiting both farm families and broader society (Schroth and Harvey, 2007; De Beenhouwer et al., 2013; Vaast and Somarriba, 2014). Research in coffee (*Coffea* sp) and cacao systems have shown that shade trees play a key role in regulating humidity and temperature fluctuations (Beer et al., 1998) and in reducing the overall vulnerability of these systems (Lin et al., 2008).

3.2 Carbon stocks

Total C stocks per farm in this study ranged from 53 to 195 Mg C ha–1, with an average of 134.4 Mg C ha^{-1} in agroforest systems and 104.7 Mg C ha^{-1} in monocultural systems (Figure 4). Most of the C stock in agroforest systems accumulated in aboveground biomass (52.48% of the total stock), including 45.16% and 6.62% stored in the trees and necromass, respectively, with the remaining <1.5% accumulated in the litter and herbaceous plants. Most of the belowground C stock (47.54% of the total stock) in the agroforest

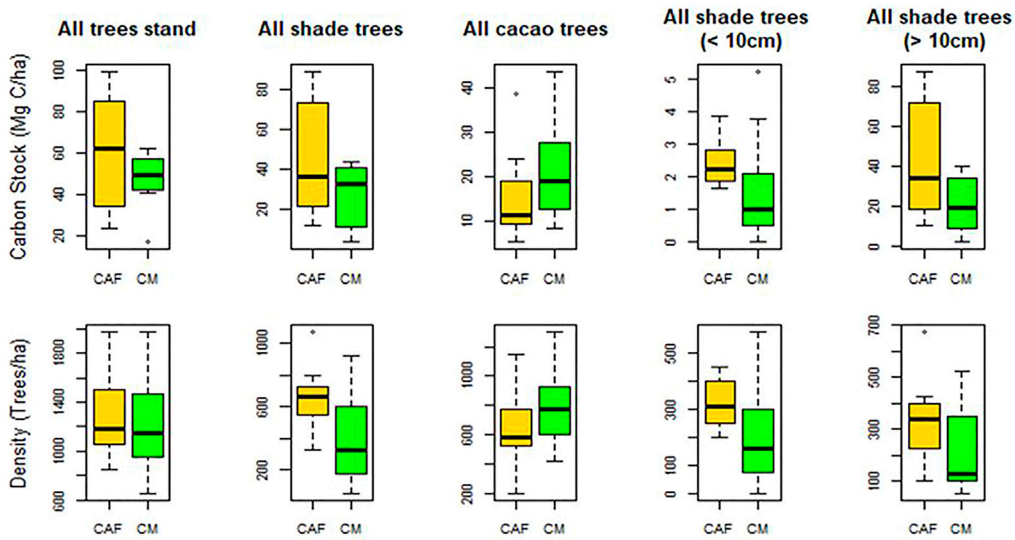

FIGURE 5
Tree density and carbon stocks of tree-type categories in CAF and CM systems.

systems accumulated in the soil (36.24%) with the remaining (11.31%) in the root biomass. By contrast, in monocultural systems, most of the C stocks accumulated below ground (52.24% of the total stock), specifically in the soil (40.88%) and roots (11.37%). The aboveground biomass of monocultural systems (47.68% of the total stock) accumulated in the tree stands (45.56%), necromass (1.15%), and litter and herbaceous plants (<1%).

Average C stocks for trees, necromass, litter, herbaceous plants, soil, and roots in agroforest systems were 60.7, 8.9, 0.82, 0.11, 48.7, and 15.2 Mg C ha^{-1}, respectively, while in monocultural systems, the respective C stocks were 47.7, 1.2, 0.97, 0.05, 42.8, and 11.9 Mg C ha^{-1}. The data are shown in Figure 5. Statistically, C stocks were not found to be significantly different between monocultural and agroforest systems in our study. This could be a result of the high variability in the C stocks of the systems attributed to the differences in age, structure, and management practices of cacao systems included in the site. Albrecht and Kandji (2003) also document the impact of the variation in smallholder practices and systems on C stocks and statistical analysis. The C stocks in our study were in the range of those reported by Silatsa et al. (2017) in Central Cameroon, which was 92.1 for mature and 144.5 Mg C ha^{-1} for old cacao trees. In our study, aboveground C stocks in agroforest systems were higher than the 49 Mg C ha^{-1} and 46 Mg C ha^{-1} reported for shaded cacao systems in Central America (Somarriba et al., 2013) and intensified cacao agroforests in Southern Bahia, Brazil (Schroth et al., 2015), but lower than the 100 Mg C ha^{-1} for cacao multishade systems as reported by Abou Rajab et al. (2016). These levels of C stocks in the monocultural and agroforest systems were similar to those reported by Roshetko et al. (2007) in smallholders' coffee and rubber (*H. brasiliensis*) systems. At the maximum of age 25+ years, rubber and coffee systems can store 190 and 100 Mg C ha^{-1}, respectively. These smallholder tree commodity systems are relevant to this study since Indonesia is a center for smallholder rubber, coffee, and cacao production.

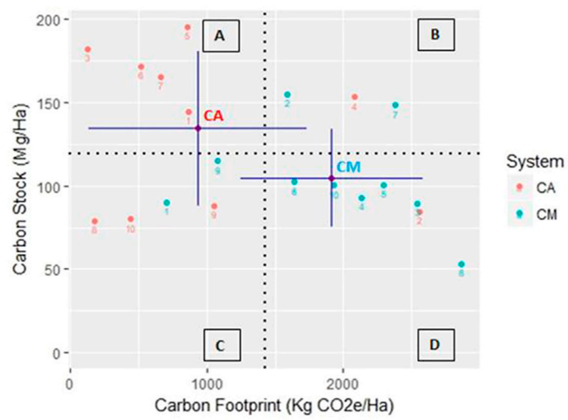

FIGURE 6
Relationship between the carbon stocks and footprints in the two systems. The dashed lines show the median of carbon stocks and footprints, respectively, dividing the fields into four quadrants of the most desirable **(A)**, least desirable **(D)**, and intermediate **(B, C)** climate impact for CAF and CM systems.

Tree density is an essential factor for C accumulation since most of the C stock in the CAF and CM systems is stored in the tree and root components, 56.47% and 56.93% of the total C stocks, respectively. The greatest proportion of C stocks is accounted for by the tree and their diameter size (Dawoe et al., 2016). The higher C stocks in the CAF system were contributed by the diversity of the shade trees (Figure 6). Somarriba et al. (2013) also revealed that the larger proportion of C stocks in most cacao agroforests is stored in the shade trees. The proportion of shade trees in the system and the density of the trees influence the C stock. The trend shows a clear increase in the C stock with increased tree density. The C stocks from the shade trees' category in agroforest and monocultural

TABLE 2 Carbon footprint of CMs and CAFs at the farm level.

Source of emission (kg CO_2e ha^{-1})	CM		CAF	
	Mean ± SE*	%	Mean ± SE*	%
Emissions from fertilizer use	1,852.8 ± 217.0^a	96.7	890.8 ± 260.6^b	95.5
Emissions from pesticide use	6.2 ± 4.4^a	0.3	20.7 ± 17.3^a	2.2
Emissions from fuel use	55.4 ± 14.6^a	2.89	20.6 ± 8.7^b	2.3
Total	1,914.4 ± 212.0^a	100	9,32.1 ± 251.6^b	100

*Mean values with different superscript letters within each column denote significant ($p < 0.05$) differences between groups.

systems were 45.4 and 27 Mg ha^{-1}, respectively, with tree densities of 655 to 418 trees ha^{-1}. The trend is the opposite for C stocks in the cacao tree category, where monocultural systems had higher C stocks and tree densities (Figure 6).

The CAF systems tended to have trees with bigger diameters, contributing to higher C stocks. The graph in Figure 6 shows that the number of shade trees with diameters bigger than 10 cm were twice as common in agroforest systems compared to monocultural systems, greatly contributing to the higher C stocks. According to Albrecht and Kandji (2003), a strategy for on-farm C storage is agroforest systems that integrate annual crops to facilitate the growth of tree components, the most important source of biomass and C stocks.

3.3 Carbon footprints

The main source of C footprints in this study was from fertilizer inputs, accounting for 96.7% and 95.5% of the total in CAF and monocultural systems, respectively. Fuel and pesticide use accounted for the balance of the C footprint in systems (Table 2). The total C footprints generated in the monocultural and agroforest systems reached 1,914.4 kg CO_2e ha^{-1} or 4.47 kg CO_2e kg^{-1} of cacao beans and 932.1 kg CO_2e ha^{-1} or 2.18 kg CO_2e kg^{-1} of cacao beans ($p < 0.05$), respectively. Emissions from fertilizer and fuel use were significantly higher in monocultural systems, while agroforest systems had higher emissions from pesticide use ($p >$ 0.436) (Table 2). Referring to the total C footprint produced, the monocultural systems were significantly higher than that of the agroforest systems, producing more than twice the C footprint. The level of C footprints produced by monocultural systems in this study was greater than that of intensive cacao production under the Cabruca system in Southern Bahia, Brazil, as reported by Schroth et al. (2016). The C footprint averaged 0.36 kg CO_2e kg^{-1} of cacao beans varying from 0 to 1.76 kg CO_2e kg^{-1} of cacao beans. The study revealed that the highest emission was observed at low-to-medium yields where excessive levels of fertilizers were applied by farmers in effort to increase production. The level of C footprints varied depending on farm management practices and conditions. Excessive levels of fertilizer use did not secure high yields. Many factors such as soil condition, poor management, and the condition of cacao trees affected the yield response to fertilizer application. These factors also contribute to the variations in C stocks and C footprints.

Bivariate Pearson correlation analysis indicated that all the biophysical parameters tested were significantly related to the C stocks except for the Shannon–Wiener index, applied fertilizer, and light intensity. Only canopy openness had a significant relationship to the carbon footprint values. The correlation value varied independently related to the parameters tested (Table 3).

The relationship between shade, tree density, and soil organic values showed a significantly positive correlation to C stocks while canopy openness had a significantly negative correlation to the C footprints. These results indicate that biophysical aspects can contribute to climate-friendly cacao management systems. The diverse understory may contribute to the higher C stocks and the low level of C footprints. The increase in shade levels and density of trees may influence the biomass, producing high C stock levels. Several authors emphasize that shade trees influence C stocks (Steffan-Dewenter et al., 2007; Tscharntke et al., 2011; Schroth et al., 2016). Studies by Oke and Olatiilu (2011), for instance, report that shade trees stored up to 65% of the total tree C in cacao systems in Nigeria and Cameroon.

The present study also shows that shade has a negative significant correlation to the C footprint levels, with increasing shade levels linked to decreasing C footprints. The canopy level may contribute not only to soil accumulation but also nutrients. The greatest contributor to the C footprint in this study was fertilizer use. By increasing the shade level, the C footprint level can be reduced. Studies by Beer et al. (1998) and Fassbender et al. (1991) indicated the role of shade trees in cacao and coffee systems to reduce soil erosion and nutrient leaching from the impact of raindrops, improve soil structure, increase soil nitrogen content, and enhance nutrient retention.

3.4 The implications of each system for climate friendliness and generation of livelihoods

Agriculture is the dominant source of livelihood in Polewali Mandar, with cacao production being the main agricultural activity for the farmers in this study. Local farmers often convert traditional cacao agroforest systems to monocultural systems by decreasing the number and species of shade trees. This intensification pattern is financially favorable in the short term but risky in terms of ecological and livelihood sustainability. Monocultural systems may have greater cacao productivity in the short term but comparatively lower long-term prospects for smallholders (Rice and Greenberg, 2000; Tscharntke et al., 2011; Mithofer et al., 2017). Our study indicates that the monocultural systems have 9.5% higher cacao yields compared to agroforest systems; however, this difference is

TABLE 3 Pearson correlation between carbon measurements and biophysical aspects.

Independent variable	Dependent variable	Correlation value	*p*-value ($p < 0.05$)
Carbon stock	Canopy cover	0.462	0.040*
	Shannon–Wiener	0.147	0.535
	Density	0.728	0.000*
	Soil organic	0.786	0.000*
	Applied fertilizer	−0.324	0.163
	Light intensity	−0.361	0.118
Carbon footprint	Shannon–Wiener	−0.428	0.060
	Area cultivated	0.010	0.966
	Canopy cover	−0.473	0.035*
	Tree density	−0.283	0.226

*A *p*-value shows significance ($p < 0.05$) according to Pearson's correlation.

not statistically significant (Table 1). This high yield was influenced by the density of cacao trees and management practices featuring more inputs. Consequently, there was more income from the cacao yields but a higher capital burden for agricultural inputs. Armengot et al. (2016) also reported that cacao monocultures required more agricultural inputs than agroforests. Monocultural systems tend to produce high C footprints as a result of the intensive use of inputs. Therefore, it is considered less climate friendly. Our results indicate that cacao monocultures are less desirable regarding climate friendliness and potential threats to the environment. Comparatively, cacao agroforest systems have lower C footprints and higher C stocks (Figure 4).

There was a momentous difference between cacao monocultural and agroforest systems regarding climate friendliness and generation of livelihoods. Our data suggest that there are possible pathways for carbon-friendly intensification in cacao agroforest systems through lower C footprints and higher C stocks. The cacao agroforest systems are gradually established under sustainable land use guidelines that meet biological, ecological, and economic objectives. This approach also maintains other valuable crops, which contribute to improving smallholders' livelihoods (Schroth et al., 2015; Mithofer et al., 2017). A study conducted by Schroth et al. (2016) reported that cacao intensification under tree canopies with shade levels of 20%–90% (called the Cabruca system) is compatible with climate friendliness and increases productivity. The productivity of Cabruca systems (Southern Bahia, Brazil) can be doubled through the proper use of mineral and organic fertilizer, while also maintaining low input-related C footprints.

The diverse shade tree and companion tree species that exist in cacao agroforests have certain benefits for the ecosystem. Tscharntke et al. (2011) claimed that shade trees have environmental, social, and economic values as well as play an important role in reducing household vulnerability to climatic stress and food insecurity. Products obtained from shade trees include firewood, medicine, resins, honey, fiber, and construction materials, all of which can provide alternative sources of income in both the short and long terms, thus delivering farm income variability throughout the year and providing resiliency to declines in cacao prices (Steffan-Dewenter et al., 2007; Tscharntke et al., 2011; Somarriba et al., 2013). This high potential of cacao agroforests offers opportunities to improve income generation at the household and even community level. It is necessary to recognize the importance of these diverse systems as sources of biodiversity conservation, reduced emissions, and climate change resilience. Our findings confirm that livelihoods are positively impacted by the conservation of plant biodiversity that can simultaneously serve as alternative sources of products and income. Even when cacao agroforests are not the main generator of income, they are still suitable for securing local culture and tradition.

4 Conclusion

The cacao farms in our study varied greatly in characteristics and biophysical aspects, depending on the management practices and site conditions. Nonetheless, results suggest that cacao production systems vary extensively in their contribution to climate change in terms of both C storage and C footprint. Cacao agroforests, with characteristically lower agricultural inputs and higher densities of shade trees and biodiversity, can play an important role in storing greater C stocks. Their shade trees and other environmental services also contribute to a lower need for agricultural inputs, such as mineral and organic fertilizers, resulting in lower C footprints. Our study shows that cacao agroforests store larger C stocks than monocultural systems, (134.43 Mg ha^{-1} compared to 104.7 Mg C ha^{-1}, respectively) and have lower C footprints (932.1 CO_2e ha^{-1} compared to 1,914.4 kg CO_2e ha^{-1}, respectively). Consequently, cacao agroforest systems are considered more climate friendly.

Cacao production systems cannot achieve zero emissions since the use of some agrochemical inputs is common. Nevertheless, there are some practices that help reduce emissions during the production process. The efficient use of agricultural inputs, in accordance with expert recommendations, especially regarding fertilizer application, can reduce emissions, the waste of fertilizers, and input costs. The study found that the application of chemical fertilizer varied greatly at the farm level and directly affected C footprints. Most farmers applied fertilizers inappropriately, many often exceeding recommended application rates.

To complement this study, further research should be conducted to fully understand the potential of climate-friendly smallholder cacao production systems under different climatic conditions and management practices. Understanding the impacts of farm management options on C stocks and C footprints could enhance cacao production, improve farmer livelihoods, and reduce GHG emissions. All this information would be useful for advancing the development of an environmentally friendly cacao industry.

Data availability statement

The raw data supporting the conclusion of this article will be made available by the authors, without undue reservation.

Author contributions

All authors contributed to the development of the research concept; TM contributed to the data analysis; data curation and review were performed by IS and NW; JR contributed to the sampling and experimental design; original draft was written by TM and IS; and writing, review and editing was performed by all authors.

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Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

Publisher's note

References

Abou Rajab, Y., Leuschner, C., Barus, H., Tjoa, A., and Hertel, D. (2016). Cacao cultivation under diverse shade tree cover allows high carbon storage and sequestration without yield losses. *PLOS ONE* 11 (2), e0149949. doi:10.1371/journal.pone.0149949

Albrecht, A., and Kandji, S. T. (2003). Carbon sequestration in tropical agroforestry systems. *Agric. Ecosyst. Environ.* 99 (1–3), 15–27. doi:10.1016/S0167-8809(03)00138-5

Arifin, J. (2001). *Estimation of carbon stock in the different land-use system in ngantang, malang. Thesis*. Indonesian: Brawijaya University.

Armengot, L., Barbieri, P., Andres, C., Milz, J., and Schneider, M. (2016). Cacao agroforestry systems have higher return on labor compared to full-sun monocultures. *Agron. Sustain. Dev.* 36 (4), 70. doi:10.1007/s13593-016-0406-6

Asare, R., Bo, M., Rebecca, A. A., Gilbert, A., and Anders, R. (2019). On-farm cocoa yields increase with canopy cover of shade trees in two agro-ecological zones in Ghana. *Clim. Dev.* 11 (5), 435–445. doi:10.1080/17565529.2018.1442805

Atangana, A., Khasa, D., Chang, S., and Degrande, A. (2014). *Tropical agroforestry*. Netherlands: Springer. doi:10.1007/978-94-007-7723-1

Audsley, E. (1997). *Harmonisation of environmental life cycle assessment for agriculture. Final report, concerted action AIR3-CT94-2028*. Brussels: European Commision, 139.

Beer, J., Muschler, R., Kass, D., and Somarriba, E. (1998). "Shade management in coffee and cacao plantations," in *Directions in tropical agroforestry research*. Editors P. K. R. Nair and C. R. Latt (Netherlands: Springer), 53, 139–164. doi:10.1007/978-94-015-9008-2_6

Bentley, J. W., Boa, E., and Stonehouse, J. (2004). Neighbor trees: Shade, intercropping, and cacao in Ecuador. *Hum. Ecol.* 32, 241–270. doi:10.1023/b:huec.0000019759.46526.4d

Bouwman, A. F., Boumans, L. J. M., and Batjes, N. H. (2002). Emissions of N2O and NO from fertilized fields: Summary of available measurement data. *Glob. Biogeochem. Cycles* 16 (4), 6–1.

Brown, S. (1997). *Estimating biomass and biomass change of tropical forests: A primer*. Rome, Italy: Food and Agriculture Organization of the United Nations.

Cairns, M., Brown, S., Helme, E. H., and Baumgardner, G. A. (1997). Root biomass allocation in the world's upland forests. *Oecologia* 111 (1), 1–11. doi:10.1007/s004420050201

Chapman, S. R., and Carter, L. P. (1976). *Crop production: Principles and practices*. San Francisco: W.H. Freeman and Company, 146–163.

Dawoe, E., Asante, W., Acheampong, E., and Bosu, P. (2016). Shade tree diversity and aboveground carbon stocks in theobroma cacao agroforestry systems: Implications for REDD+ implementation in a west african cacao landscape. *Carbon Balance Manag.* 11 (1), 17. doi:10.1186/s13021-016-0061-x

De Beenhouwer, M., Aerts, R., and Honnay, O. (2013). A global meta-analysis of the biodiversity and ecosystem service benefits of coffee and cacao agroforestry. *Agric. Ecosyst. Environ.* 175, 1–7. doi:10.1016/j.agee.2013.05.003

Effendy (2015). Application of side-grafting technology to increase cocoa productivity: Case study in sigi regency, Indonesia. *J. Appl. Sci.* 15 (4), 715–718. doi:10.3923/jas.2015.715.718

Fairbairn, W. A. (1958). Method of light intensity measurement in forest stands: II. The use of light measurement instruments in the field. *Int. J. For. Res.* 31, 155–162. doi:10.1093/forestry/31.2.155

Fassbender, H. W., Beer, J., Heuveldop, J., Imbach, A., Enriquez, G., and Bonnemann, A. (1991). Ten year balances of organic matter and nutrients in agroforestry systems at

CATIE, Costa Rica. *For. Ecol. Manag.* 45 (1–4), 173–183. doi:10.1016/0378-1127(91)90215-H

Frischknecht, R., Jungbluth, N., Althaus, H.-J., Doka, G., Dones, R., Heck, T., et al. (2007). *Overview and methodology*. Ecoinvent report No. 1. Dübendorf, Switzerland: Swiss Centre for Life Cycle Inventories.

Hairiah, K., Sitompul, S. M., Noordwijk, M. V., and Cheryl, P. (2001). *Methods for sampling carbon stocks above and below ground. ASB lecture note 4B. Bogor, Indonesia*. Uttar Pradesh: International Centre for Research in Agroforestry.

Hartemink, A. E. (2005). Nutrient stocks, nutrient cycling, and soil changes in cocoa ecosystems: A review. *Adv. Agron.* 86, 227–253. doi:10.1016/S0065-

Hillier, J., Walter, C., Malin, D., Garcia-Suarez, T., Mila-i-Canals, L., and Smith, P. (2011). A farm-focused calculator for emissions from crop and livestock production. *Environ. Model. Softw.* 26 (9), 1070–1078. doi:10.1016/j.envsoft.2011.03.014

IPCC (2006). *Interngovermental panel on climate change (IPCC)*. Tokyo, Japan: Institute for Global Environmental Strategies.Revised good practice guidelines for greenhouse gas inventories

Isaac, M. E., Timmer, V. R., and Quashie-Sam, S. J. (2007). Shade tree effects in an 8-year-old cocoa agroforestry system: Biomass and nutrient diagnosis of theobroma cacao by vector analysis. *Nutrient Cycl. Agroecosyst.* 78, 155–165. doi:10.1007/s10705-006-9081-3

Ishida, M. (2004). Automatic thresholding for digital hemispherical photography. *Can. J. For. Res.* 34, 2208–2216. doi:10.1139/x04-103

Janudianto, J., Roshetko, J. M., and Mahrizal, J. (2014). "Cacao agroforestry system (CAS): Improving productivity and profitability of smallholder cacao in Sulawesi. Bogor. Indonesia. World agroforestry Centre (ICRAF) Southeast Asia regional office." In Presentation at World Agroforestry Congress, India, February 9–14, 2014.

Lin, B. B., Perfecto, I., and Vandermeer, J. (2008). Synergies between agricultural intensification and climate change could create surprising vulnerabilities for crops. *BioScience* 58 (9), 847–854. doi:10.1641/B580911

Manaker, G. H. (1996). *Interior plantscapes: Installation, maintenance, and management*. Engelwood Cliffs, NJ: Prentice Hall.

Mithofer, D., Roshetko, J. M., Donovan, J. A., Nathalie, N. E., Robiglio, V., Wau, D., et al. (2017). Unpacking 'sustainable' cocoa: Do sustainability standards, development projects and policies address producer concerns in Indonesia, Cameroon, and Peru? *Int. J. Biodivers. Sci. Ecosyst. Serv. Manag.* 13 (1), 444–469. doi:10.1080/21513732.2018.1432691

Miyaji, K. I., Silva, W. S., and Alvim, P. T. (1997). Productivity of leaves of a tropical tree, theobroma cacao, grown under shading, in relation to leaf age and light conditions within the canopy. *New Phytol.* 137, 463–472. doi:10.1046/j.1469-8137.1997.00841.x

Nair, K. P. P. (2010). *The agronomy and economy of important tree crops of the developing world*. Amsterdam, Netherlands: Elsevier. 131–181.

Nair, P. K. R., Mohan Kumar, B., and Nair, V. D. (2009). Agroforestry as a strategy for carbon sequestration. *J. Plant Nutr. Soil Sci.* 172 (1), 10–23. doi:10.1002/jpln.200800030

Oke, D., and Olatiilu, A. (2011). Carbon storage in agroecosystems: A case study of the cocoa based agroforestry in ogbese forest reserve, ekiti state, Nigeria. *J. Environ. Prot.* 02 (08), 1069–1075. doi:10.4236/jep.2011.28123

IPCCPenman, J. (2003). "IPPC national greenhouse gas inventories programme," in *Good practice guidance for land use, land-use change and forestry* (Bert Bolin: The Intergovernmental Panel on Climate Change).

Pohlan, H. A. J., and Pérez, V. D. (2010). "Growth and production of cacao. Soils, plant growth, and crop production," in *Encyclopedia of life support systems (EOLSS)* (Paris France: UNESCO).

Rice, R. A., and Greenberg, R. (2000). Cacao cultivation and the conservation of biological diversity. *Ambio A J. Hum. Environ.* 29, 167–173. doi:10.1579/0044-7447-29.3.167

Roshetko, J. M., Lasco, R. D., and Angeles, M. S. D. (2007). Smallholder agroforestry systems for carbon storage. *Mitig. Adapt. Strategies Glob. Change* 12 (2), 219–242. doi:10.1007/s11027-005-9010-9

Schneider, M., Andres, C., Trujillo, G., Alcon, F., Amurrio, P., Perez, E., et al. (2017). Cocoa and total system yields of organic and conventional agroforestry vs. Monoculture systems in a long-term field trial in Bolivia. *Exp. Agric.* 53 (3), 351–374. doi:10.1017/S0014479716000417

Schroth, G., Bede, L. C., Paiva, A. O., Cassano, C. R., Amorim, A. M., Faria, D., et al. (2015). Contribution of agroforests to landscape carbon storage. *Mitig. Adapt. Strategies Glob. Change* 20 (7), 1175–1190. doi:10.1007/s11027-013-9530-7

Schroth, G., and Harvey, C. A. (2007). Biodiversity conservation in cocoa production landscapes: An overview. *Biodivers. Conservation* 16, 2237–2244. doi:10.1007/s10531-007-9195-1

Schroth, G., Jeusset, A., Gomes, A. D., Florence, C. T., Coelho, N. A. P., Fariah, D., et al. (2016). Climate friendliness of cocoa agroforests is compatible with productivity increase. *Mitig. Adapt. Strategies Glob. Change* 21, 67–80. doi:10.1007/s11027-014-9570-7

Silatsa, F. B. T., Yemefack, M., Ewane-Nonga, N., Kemga, A., and Hanna, R. (2017). Modeling carbon stock Dynamics under fallow and cocoa agroforest systems in the shifting agricultural landscape of Central Cameroon. *Agrofor. Syst.* 91 (5), 993–1006. doi:10.1007/s10457-016-9973-4

Smith, R. L. (1990). *Elements of ecology*. 2nd edition. New York: West Virginia University Harper & Row.

Somarriba, E., Cerda, R., Orozco, L., Cifuentes, M., Dávila, H., Espin, T., et al. (2013). Carbon stocks and cocoa yields in agroforestry systems of central America. *Agric. Ecosyst. Environ.* 173, 46–57. doi:10.1016/j.agee.2013.04.013

Steffan-Dewenter, I., Kessler, M., Barkmann, J., Bos, M. M., Buchori, D., Erasmi, S., et al. (2007). Tradeoffs between income, biodiversity, and ecosystem functioning during tropical rainforest conversion and agroforestry intensification. *Proc. Natl. Acad. Sci. U. S. A.* 104, 4973–4978. doi:10.1073/pnas.0608409104

Tscharntke, T., Clough, Y., Bhagwat, S. A., Buchori, D., Faust, H., Hertel, D., et al. (2011). Multifunctional shade-tree management in tropical agroforestry landscapes — a review: Multifunctional shade-tree management. *J. Appl. Ecol.* 48 (3), 619–629. doi:10.1111/j.1365-2664.2010.01939.x

Vaast, P., and Somarriba, E. (2014). Tradeoffs between crop intensification and ecosystem services: The role of agroforestry in cocoa cultivation. *Agrofor. Syst.* 88, 947–956. doi:10.1007/s10457-014-9762-x

van Rikxoort, H., Schroth, G., Läderach, P., and Rodríguez-Sánchez, B. (2014). Carbon footprints and carbon stocks reveal climate-friendly coffee production. *Agron. Sustain. Dev.* 34 (4), 887–897. doi:10.1007/s13593-014-0223-8

Vernon, A. J., and Sundaram, S. (1972). "Current cocoa research," in *Proc. 4th int. Cocoa res. Conf.* (St. Augustine: Trinidad and Tobago), 689–693.

Walkley, A., and Black, I. A. (1934). An examination of Degtjareff method for determining organic carbon in soils: Effect of variation in digestion condition and inorganic soil constituents. *Soil Sci.* 63, 251–263. doi:10.1097/00010694-194704000-00001

Wielgoss, A., Clough, Y., Fiala, B., Rumede, A., and Tscharntke, T. (2012). A minor pest reduces yield losses by a major pest: Plant-mediated herbivore interactions in Indonesian cacao: Indirect interaction between cacao pests. *J. Appl. Ecol.* 49 (2), 465–473. doi:10.1111/j.1365-2664.2012.02122.x

Yuliasmara, F., Aris, W., and Prawoto, A. A. (2009). Carbon storage in different age classes and cultivation system of cacao: Allometric approaches. *Pelita Perkeb.* 25, 86–100. (in Indonesian). doi:10.22302/iccri.jur.pelitaperkebunan.v26i3.137

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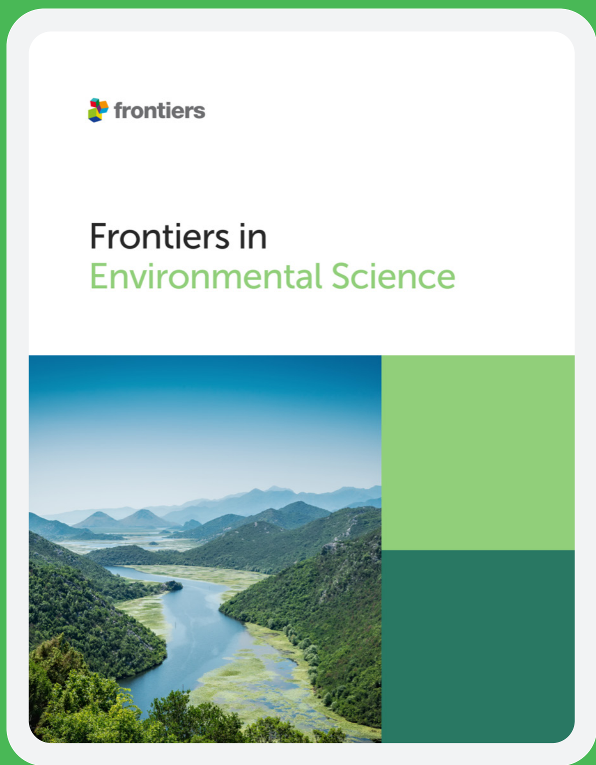

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