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Mapping monetary values of ecosystem services in support of developing ecosystem accounts

Elham Sumarga [a,b,*], Lars Hein [a], Bram Edens [c,1], Aritta Suwarno [a]

[a] Environmental Systems Analysis Group, Wageningen University, PO Box 47, 6700 AA Wageningen, The Netherlands
[b] School of Life Sciences and Technology, Institut Teknologi Bandung (ITB), Jalan Ganesa 10, Bandung 40132, Indonesia
[c] National Accounts Department, Statistics Netherlands, Henri Faasdreef 312, 2492 JP Den Haag, The Netherlands

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ABSTRACT

Ecosystem accounting has been proposed as a comprehensive, innovative approach to natural capital accounting, and basically involves the biophysical and monetary analysis of ecosystem services in a national accounting framework. Characteristic for ecosystem accounting is the spatial approach taken to analyzing ecosystem services. This study examines how ecosystem services can be valued and mapped, and presents a case study for Central Kalimantan, Indonesia. Four provisioning services (timber, palm oil, rattan, and paddy rice), one regulating service (carbon sequestration), and two cultural services (nature recreation, and wildlife habitat) are valued and mapped in a way that allows integration with national accounts. Two valuation approaches consistent with accounting are applied: the resource rent and cost-based approaches. This study also shows how spatial analysis of ecosystem accounting can support land use planning through a comprehensive analysis of value trade-offs from land conversion.

1. Introduction

Ecosystem accounting is a new area of environmental economic accounting that aims to measure ecosystem services in a way that is aligned with national accounts (Boyd and Banzhaf, 2007; European Commission (EC) et al., 2013; Edens and Hein, 2013). The System of National Accounts (SNA) (European Commission (EC) et al., 2009) provides the global standard for national accounting, and the Central Framework of the System for Environmental Economic Accounts (SEEA-CF) was designed as a satellite account of the SNA (United Nations (UN), 1993; United Nations (UN) et al., 2003), with a global standard for the SEEA-CF adopted in 2012 (United Nations (UN) et al., 2014). Ecosystem accounting involves an extension of the production boundary of the System of National Accounts (European Commission (EC) et al., 2013). This allows the inclusion of a broader set of ecosystem services types such as regulating services and cultural services as well as the natural growth of biological assets such as timber in measures of economic activity. In turn, this allows a more comprehensive recording of changes in ecosystem capital, i.e. the stock of ecosystems that provides a foundation for future well-being, and provides a more complete dataset for environmental policy making (Campbell and Tilley, 2014).

Ecosystem accounting involves approaches to measuring ecosystem capital and comprises the monitoring of ecosystem services flows, the capacity of ecosystems to generate these services, and the condition of ecosystems (European Commission (EC) et al., 2013). Ecosystem condition determines the capacity to generate services, as in the case of standing timber stock, species composition, soil fertility, rainfall, etc. determining the capacity to supply timber at present as well as over time. There remain considerable challenges in implementing ecosystem accounting (Edens and Hein, 2013). One of the main issues is if, how and to what degree ecosystem capital can be valued in monetary terms. In particular, it is still being discussed if ecosystem services flows and the capacity of ecosystems to generate services can be valued in monetary terms in a way that is both consistent with accounting, and that is sufficiently robust for the purpose of accounting (United Nations (UN) et al., 2014). Note that ecosystem condition is not directly connected to human benefits and can therefore not be valued in monetary terms.

Spatial explicitness is a distinguishing property of ecosystem accounts (all with the exception of the land account that provides indications of acreages of land in specific classes potentially combined with ownership information of the land). Both ecosystem services flow, ecosystem capacity and ecosystem condition are spatially heterogeneous (Schröter et al., 2014). There is a wide range of experience with

* Corresponding author at: Environmental Systems Analysis Group, Wageningen University, PO Box 47, 6700 AA Wageningen, The Netherlands.
Tel.: +31 626249922; fax: +31 317 419000.
E-mail address: elham.sumarga@wur.nl (E. Sumarga).

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mapping the values of ecosystem services (Plieninger et al., 2013; van Berkel and Verburg, 2014; Palomo et al., 2014), and very limited experience with mapping the values of the capacity of ecosystems to supply ecosystem services (Chen et al., 2009; Ericksen et al., 2012). Values have been mapped among others in support of land use planning (Fisher et al., 2011; Ruiz-Frau et al., 2012; Scolozzi et al., 2012) and to monitor the impacts of land use change (Kreuter et al., 2001; Li et al., 2010; Mendoza-González et al., 2012). However, at present, there have been few if any analyses involving the mapping of ecosystem service values in the context of, and aligned with environmental-economic accounting.

The objective of this paper is to examine how ecosystem services can be valued and mapped in a manner aligned with national accounts. In particular, we analyze and map the monetary value of a comprehensive set of ecosystem services in Central Kalimantan province, Indonesia. The novelty of our paper is in the application of a valuation approach consistent with accounting, and in the application of valuation approach to a relatively large area (around 150,000 km^2). In addition, we explore an experimental valuation approach for one specific element of biodiversity: the conservation of orangutan habitat. We selected Central Kalimantan in view of our interest in testing the ecosystem accounting approach in a developing country context, and because Central Kalimantan has been subject to rapid land use change including deforestation in the past decades (Broich et al., 2011; Miettinen et al., 2012), requiring better information on costs and benefits of different land management approaches, and on possible value trade-offs following land conversions. This study includes a specific analysis on the conversion of forests into oil palm in terms of the trade-offs that occur between ecosystem service values.

We value and map seven ecosystem services, following the classifications of Millennium Ecosystem Assessment (MEA) (2003) and TEEB (2010), in a way that permits integration with national accounts. In particular, we distinguish the following services: timber production, rattan production, oil palm production, paddy rice production, carbon sequestration, wildlife habitat, and nature recreation. Although this is not a complete set of ecosystem services generated in the study area, our set is sufficiently large and diverse to explore if and how ecosystem services valuation and mapping can be applied in the context of ecosystem accounting. We explore in our paper if valuation data and analytical approaches are sufficiently robust for integration in accounts, if not, what further steps need to be taken, and what potential other policy applications may exist for spatial maps of monetary values aligned with the system of national accounts.

The outline of the paper is as follows. In Section 2, we describe the valuation methods selected for valuing the seven ecosystem services and how the values are then mapped. In Section 3, we present the monetary value maps and the summary of multiple ecosystem services values in the main land cover classes. In Section 4, we discuss three main issues: monetary valuation and mapping of ecosystem services in support of accounting, challenges in valuation and integrating ecosystem services values in an accounting framework, and value trade-offs and their policy implications.

2. Methods

2.1. Study area

We selected Central Kalimantan province, Indonesia for this study, in view of our interest in testing accounting methods in a developing country context and for a large area. Central Kalimantan is one of the largest provinces in Indonesia, and has been appointed as pilot province for a REDD+ project enhancing data availability of some ecosystem services, in particular those related to carbon. The province covers an area of 153,500 km^2, and is located at latitude 0°45′ North–3°30′ South and longitude 110°45′–115°50′ East. Most of the area (57%) is covered by forest (Fig. 1). This province has experienced rapid land cover change, mostly conversion of forest to other uses, such as oil palm plantations. Based on a comparison of land cover maps of 2000 and 2010 (Tropenbos Indonesia, unpublished), about 14,000 km^2 areas (12.7%) have been deforested during that period. The province has a low population density with an average of 14 people/km^2 and a total population of 2149,896.

2.2. Spatial modeling and mapping of ecosystem services

This papers builds upon previously developed ES models (Sumarga and Hein, 2014), in which physical models for a range of ecosystem services were developed and applied to Central Kalimantan. A range of methods were applied to model and map these services in physical terms including Geostatistics, Maxent, and lookup tables. For this paper, we extend the previous paper with an additional ecosystem service, recreation. We exclude from our paper the service carbon storage since this does not constitute a flow and therefore cannot be included as an ecosystem service in an ecosystem account (even though it is highly relevant for land use planning).

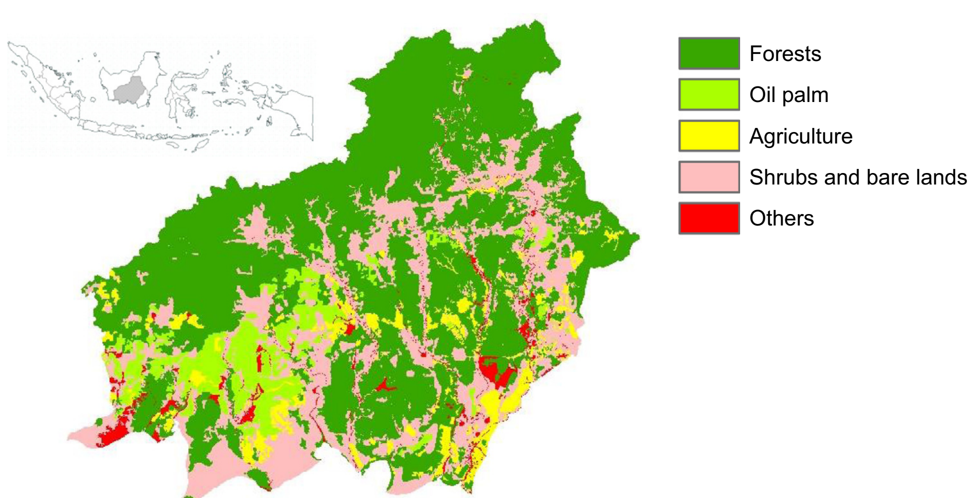

Fig. 1. Land cover map of Central Kalimantan.

2.3. Valuation methods

In this paper, we examine how the seven ecosystem services can be valued, and subsequently mapped, in a way that is aligned with national accounting. Valuation in the context of accounting requires a clear distinction between services and benefits, with services representing the contribution made by ecosystems to benefits used by people (European Commission (EC) et al., 2013). Some of these benefits are already captured in the SNA (e.g. crop harvesting), but other benefits (e.g. carbon sequestration) are not recognized in the SNA, which is why the production boundary is extended in ecosystem accounting. In the SNA, goods and services are valued at exchange values, based on representative market prices where these are available (European Commission (EC) et al., 2009). The SNA provides a standard on how products and assets can be valued in the context of the national accounts, including valuation approaches for valuing public goods (such as health care), for agricultural commodities and assets (both produced asset such as machines or non-produced assets such as land).

The SNA valuation approach has not yet been comprehensively applied to non-market ecosystem services (e.g. Edens and Hein, 2013), but the SNA does provide a number of insights in how this can be done. For instance, public goods are valued 'at-cost' (European Commission (EC) et al., 2009), which implies that an avoided-damage cost method may be appropriate for non-market ecosystem services, if it is reasonable to expect that a government, households, or firm would invest in order to mitigate the damages resulting from environmental degradation. The production factor approach, i.e. valuing an ecosystem service as a supporting factor of production, is another valuation approach that is potentially consistent with the SNA (European Commission (EC) et al., 2009), but this method is difficult to apply when an ecosystem service (such as flood control) is providing an input to a whole range of activities at the same time. This would require disentangling the contribution of flood control to a myriad of economic activities, even though in reality in some cases a loss of a flood control service would be mitigated by the construction of physical flood control infrastructure.

In this study, we apply two main valuation approaches. We analyze provisioning services and recreation based on a resource rent approach. For carbon sequestration and orangutan habitat we apply a cost based approach, as explained in more detail below.

2.3.1. Provisioning services

The four provisioning services we analyze (timber, oil palm, rattan, paddy rice) were chosen because they constitute the main products traded in Central Kalimantan. We calculate the resource rent (European Commission (EC) et al., 2013) to reveal the contribution of the ecosystem to the market outputs as presented by Equation 1. The resource rent equals the revenues minus the value of intermediate consumption, and labor and the user costs of fixed assets (United Nations (UN) et al., 2014). The user costs of fixed assets consist of consumption of fixed capital (depreciation) and the cost of capital. The latter measures an opportunity cost for the money tied up in fixed assets. The costs of capital can be estimated as the interbank lending rate plus a risk premium (Veldhuizen et al., 2009). We consider the costs of capital only for oil palm cultivation and ecotourism that require significant investments (in the case of ecotourism for instance for means of transport). Since both the lending rate and the inflation rate vary considerably between years we took an average of both for the 3 years period. For the period 2009–2011, the average interbank lending rate is 13.0% (Statistics Indonesia, 2014a), the average inflation rate is 4.5% (Statistics Indonesia, 2014b), and we assume a risk premium of 1.5% (Veldhuizen et al., 2009), resulting in a real discount rate of 10%. We use the following equation to calculate the resource rent:

$$RR = TR - (IC + LC + UCF)$$

where RR = resource rent, TR = total revenue, IC = intermediate consumption, LC = wages (labour costs), UCF = user costs of fixed assets.

We collected data from several sources to calculate the resource rent. For timber production, we analyzed data from the financial reports of two logging companies that are based in Central Kalimantan (data were given on the condition that the name of the companies would not be published). There are various taxes applied to timber production such as forest product tax and reforestation tax. In line with the SNA (European Commission (EC) et al., 2009), we excluded product based taxes from the calculation in order to obtain the output at basic prices to estimate the total revenue. For the other provisioning services, we used data (production, price, costs, and revenue) from a range of sources (Table 1). As data are obtained from diverse sources and different years, the resource rent values are standardized into 2010 values on the basis of an inflation rate of 11.1% (2008) and 2.8% (2009) (Statistics Indonesia, 2014b). The values are in Indonesian Rupiah (IDR), and then converted into Euro (€) with an

Table 1
Costs and revenue of provisioning services production (in euros).

Ecosystem service	Total revenue	Intermediate costs	Labor costs	User costs of fixed assets	Resource rent	Sources of data
Timber[a]	€ 118/m^3	€ 71.6/m^3	€ 11/m^3	€ 0.4/m^3	€ 35/m^3	Financial report of two logging companies (unpublished)
Oil palm on mineral soil (0–4 years)	€ 368/ha/year	€ 644/ha/year	€ 378/ha/year	€ 65/ha/year	€-719/ha/year	Fairhurst and McLaughlin (2009), Ismail (2010)
Oil palm on mineral soil (5–9 years)	€ 2744/ha/year	€ 626/ha/year	€ 368/ha/year	€ 132/ha/year	€ 1618/ha/year	
Oil palm on mineral soil (10–20 years)	€ 3135/ha/year	€ 641/ha/year	€ 377/ha/year	€ 56/ha/year	€ 2060/ha/year	
Oil palm on peat soil (0–4 years)	€ 368/ha/year	€ 778/ha/year	€ 457/ha/year	€ 130/ha/year	€-997/ha/year	
Oil palm on peat soil (5–9 years)	€ 2744/ha/year	€ 685/ha/year	€ 403/ha/year	€ 264/ha/year	€ 1392/ha/year	
Oil palm on peat soil (10–20 years)	€ 3135/ha/year	€ 701/ha/year	€ 412/ha/year	€ 113/ha/year	€ 1910/ha/year	
Rattan	€ 145/t	€ 20/t	€ 21/t	€ 0.2/t	€ 104/t	Iwan (2008), Martoniady (2009); involving 60 farmers
Paddy rice	€ 238/t	€ 39/t	€ 66/t	€ 3/t	€ 130/t	Nugroho (2008), Evaristy (2008), Yandi (2008); involving 88 farmers

[a] Costs are mostly allocated for intermediate costs as companies hire other parties for several main activities such as road construction and log transportation.

exchange rate of IDR 12,000 for € 1 (average for 2010). We have chosen not to use PPP adjusted rates as some of the services, in particular oil palm, timber and rattan, are produced predominantly for international markets.

2.3.2. Regulating services

In view of the fragmented nature of the various carbon markets (Lovell, 2010), and the high impact of the institutional setting of the market on the carbon prices (Michaelowa and Jotzo, 2005), we value carbon sequestration services based on the marginal social damage costs (Tol, 2008). That is, the sequestration of a ton of carbon is valued using the social cost of emitting a ton of carbon. The social cost of carbon (SCC), is "an estimate of the monetized damages associated with the increment increase in carbon emissions in a given year" (Interagency Working Group on Social Cost of Carbon and United States Government, 2013). Since these marginal damage costs indicate a present value of future damage cost estimates, the discount rate plays an important role in determining the marginal damage costs. In view of the public good character of carbon damages we apply a social discount rate of 3% (Interagency Working Group on Social Cost of Carbon and United States Government, 2013). The appropriate discount rate to use for non-market ecosystem services is not yet mainstreamed in ecosystem accounting (European Commission (EC) et al., 2013).

Consequently, we used an SCC value for 2010 at USD 32/t CO_2 that is equivalent to € 24/t CO_2 (€ 88/t C) with an exchange rate of $ 1.33 for € 1 (average in 2010). The monetary values for carbon sequestration service are negative in areas where carbon emissions are higher than carbon sequestration levels. This occurs, in particular, in drained peatlands (Sumarga and Hein, 2014). There are two ways of interpreting such negative values. One way is to conceive this as an ecosystem disservice (Zhang et al., 2007; Swain et al., 2013). However, from an accounting perspective, which does not have a notion of ´negative production´, it is preferable to separate the sequestration service (that is always positive), from the emissions, which can be presented as a degradation cost. Hence, we produce two maps, one depicting the value of carbon sequestration, and one map depicting the costs of carbon emissions. The latter can be integrated in a full ecosystem account in the form of ecosystem degradation.

2.3.3. Cultural services

2.3.3.1. Nature recreation. We estimate the monetary value of the nature recreation service based on both the entrance fees paid to the parks and on the revenue generated in the local ecotourism sector. We elicited the revenue from entrance fees of three national parks (Tanjung Puting, Sebangau, and Bukit Baka Bukit Raya) and two recreation parks (Bukit Tangkiling and Tanjung Keluang). We obtained this information from the statistics of National Parks and Conservation and Tourism offices. We also calculate the revenue from ecotourism. By far the most visited park in Central Kalimantan is Tanjung Puting National Park, which has a large orangutan population and a rehabilitation centre where orangutans can easily be spotted, in addition to a variety of other species such as the proboscis monkey. We focus on tourism revenues that can be directly attributed to the presence of the park and its biodiversity, in particular revenue from a lodge inside the park, and revenue generated from tourists booking a boat to visit the park (that can only be reached by boat). In 2010, around 50 boats were owned and operated by people from a nearby town and nearby villages, and they offer tours of one day up to a week or more in the park, including a guide, meals, operation of the boat, etc. To estimate the resource rent generated by the lodge and the boats we conducted a survey in July 2012 and in March 2013 to collect data on the cost and benefit of the tour organization and the hotel business, covering the hotel manager and 30 boat owners. In particular, we asked for data on guests, revenues, and costs (intermediate consumptions, labor costs, costs of fixed assets), that was willingly shared with us.

Subsequently, Formula (1) is applied to calculate the resource rent for recreation. We then used the resource rent of tour organization in Tanjung Puting national park (€/visitor/year) to estimate the value in the other two national parks, using a proportional (per visitor) benefit transfer. In these other parks, visitors also have to enter by boat, but the number of visitors is much smaller, see Table 2.

Table 2
Costs and revenue of tour organization and hotel inside Tanjung Puting National Park[a].

	Total revenue (€)	Intermediate costs (€)	Costs of employment (€)	User costs of fixed assets (€)	Resource rent (€)
Tour organization	€ 1300,000	€ 401,250	€ 373,750	€ 75,000	€ 450,000
Hotel	€ 129,000	€ 35,580	€ 35,000		€ 58,420

[a] Analyzed from primary interviews.

2.3.3.2. Orangutan habitat. Biodiversity is one of the most difficult aspects to analyze in an ecosystem accounting context, and it is questionable if it is feasible at all to value biodiversity, or aspects thereof, in a meaningful way using monetary indicators (Ehrenfeld, 1998; Nunes and van den Bergh, 2001). In order to explore and test what a potential approach to monetize biodiversity could look like, we applied the following valuation method, for one specific aspect of biodiversity: orangutan habitat. We select this as an indicator in view of the importance of the orangutan as a flagship conservation species (Alfred et al., 2010), because of its endangered status, and because Central Kalimantan is likely to have the world's largest population of orangutan at the provincial level (Wich et al., 2008). Consistent with accounting principles, we tested a defensive expenditure method for valuing the orangutan habitat service. We apply this method by analyzing the costs related to the reintroduction of orangutan in the forests of Central Kalimantan from an orangutan rescue center, operated by the NGO Borneo Orangutan Survival Foundation (BOSF). The reintroduction costs include pre-release cost, release cost and post-release monitoring cost. We assume, therefore, that the exchange value of the orangutan population can be approximated by analyzing the costs spent on the reintroduction of (marginal) orangutans. Once released, the orangutan will stay in the forest throughout his lifetime and the value of the orangutan habitat service needs to be converted to an annual value. This is done by considering that the average age an orangutan reaches in the wild is 50 years (Wich et al., 2004) and that orangutan are on average 10 years when they are released by BOSF (data from BOSF). We therefore divide the reintroduction costs by 40 years in order to obtain a monetary value for the presence of an orangutan in a forest habitat during one year. We acknowledge that orangutan habitat harbors many more species than orangutan alone, and that our value is a gross underestimation of the overall value of wildlife habitat. Note also that a willingness to pay (WTP) survey, a more frequently applied method for valuing conservation of wildlife, could not be applied in this study. This method elicits consumer surplus, hence it is incompatible with accounting principles.

2.4. Mapping monetary value of ecosystem services

Our monetary analysis builds upon the mapping of ecosystem services supply in biophysical terms published in Sumarga and Hein (2014). We calculated the resource rent of the production of timber, rattan and paddy rice in €/production unit/year, and multiply the values with the productivity of the provisioning services (production/ha/year). In this way, we convert the ecosystem services maps in terms of physical quantity into maps in terms of monetary value. We applied the same procedure for carbon sequestration by assigning the SCC (in ton of C) into the carbon sequestration map. To ensure consistence with accounting procedures, we separately mapped the monetary value of the carbon sequestration service (in areas with a positive net carbon flux) and the degradation costs of carbon emissions (in areas with a negative net carbon flux).

We applied the lookup tables technique, i.e. assigning a value to a mapping unit (assuming equal importance of every hectare in the mapping unit), for mapping the monetary values of oil palm production, nature recreation and orangutan habitat. First we created mapping units, followed by assigning the monetary value of those services to the related mapping units. For oil palm production, we created age classes as the mapping unit, since costs and revenue from oil palm production are highly dependent on plantation ages. Limited availability of land cover maps allowed us to create six classes only, a combination of three productivity classes: unproductive ages (0–4 years), early production ages (5–9 years) and mature production ages ($>$ 9 years) and two soil types: mineral soil and peat soil. For nature recreation, we used the five-park map as the mapping unit. For orangutan habitat, we created the habitat unit as the mapping unit. We considered the protected area map, the orangutan habitat suitability map from Maxent (Sumarga and Hein, 2014), and orangutan distribution area identified by Wich et al. (2008) to indicate the habitat unit. All our maps are in raster format with a pixel size of 100 m $\times$ 100 m.

3. Results

3.1. Monetary value maps

3.1.1. Timber production

We analyzed data from two timber logging companies. The average resource rent obtained by logging companies from timber production is € 35/m^3. The costs and revenue of timber production is summarized in Table 1. The monetary value map of timber production generated based on the resource rent is presented in Fig. 2a. The estimated total value for Central Kalimantan in 2010 is € 183 million with an average of € 30 per ha and a standard deviation of € 6 per ha.

3.1.2. Oil palm

We analyzed the resource rent of oil palm production in terms of fresh fruit brunch (FFB) production. The average resource rent for six classes of plantations is summarized in Table 1, and the resulting map is presented in Fig. 2b. The average resource rents are negative in the first four years as there is no FFB production. The highest production level is achieved after year 10, corresponding to a resource rent of on average € 2060/ha/year for plantations in mineral soils and € 1910/ha/year for plantations in peat soils. Plantations in peat soil generate lower resource rents due to higher costs of land management, particularly for constructing and maintaining drainage systems.

3.1.3. Rattan production

Due to the policy of the Indonesian Ministry of Trading in opening or closing raw rattan export, the rattan price in Indonesia is highly volatile. In 2012 for example, raw rattan export was stopped in order to promote production of processed rattan. Due to the un-readiness of the domestic market in absorbing the rattan supply, this policy led to a sharp decline in the rattan price. We analyze the resource rent of rattan for 2010, when no export limitations were in place, and base our analysis on studies conducted in periods without a ban on rattan export. The average resource rent of rattan production is €104/t, with the cost and revenue summarized in Table 1. The monetary value map of rattan production is presented in Fig. 2c. The total resource rent of rattan in 2010 is about € 390 million with an average of € 82 per ha and a standard deviation of € 15 per ha.

3.1.4. Paddy rice production

The average resource rent of paddy rice production is € 130/t with the cost and revenue summarized in Table 1. The monetary value map of paddy rice production is presented in Fig. 2d. The total monetary value of paddy rice production in the province in 2010 is € 222 million with an average of € 289 per ha and a standard deviation of € 47 per ha.

3.1.5. Carbon sequestration

The monetary value map of carbon sequestration is presented in Fig. 2e, and the map of the degradation costs of carbon emissions is presented in Fig. 2f. We identify 8.2 million ha contributing to sequestration, with a total monetary value of € 1990 million in 2010; and 6.7 million ha with a 'negative sequestration' (i.e. emission), with a total social costs of € 2113 million in 2010. At the scale of the province, there is therefore a net cost in 2010 of € 123 million, based on the factors that we considered in our analysis. We did not consider, however, carbon emissions due to land use change (since we value sequestration and emissions for the average situation in 2010), hence we are likely to underestimate carbon emissions given the rate of land degradation in Indonesia (Gunarso et al., 2013). Also note that because of rapid land use change (and associated ongoing drainage (Hooijer et al., 2012)), the values retrieved for 2010 may not be representative for other years.

3.1.6. Nature recreation

Table 3 provides data on revenues obtained by five parks from the visitors' entrance fees in 2010, as well as the resource rent from tour organizations and the hotel business inside Tanjung Puting National Park. We assume that the revenues from entrance fees are fully considered as resource rent as the costs of recreation provision can be neglected because of the minimum availability of infrastructure and facilities inside the national parks. There were around 50 boats operated by local people for tours in Tanjung Puting National Park in 2010. The peak season for the tour is from June to August. On average, a boat is rented 156 days/year. Our survey showed that the aggregated resource rent of the tour organizations is in total € 450,000 for 50 boats in 2010. The costs and revenues of the tour organizations are summarized in Table 2. We consider this value as an additional resource rent obtained from nature recreation in Tanjung Puting National Park. The resource rent per visitor (€ 42) is used to estimate the benefits in the other two national parks that experience a similar type of tourism. There is a hotel (Rimba Lodge) inside Tanjung Puting National Park. The hotel had on average 1032 guests/year in the period 2011–2012, and they stayed on average for 1.5 nights. The resource rent of the hotel business is € 58,417/year with the cost items summarized in Table 2. The monetary value map of nature recreation in the five parks is presented in Fig. 2h with a total estimate of resource rent of around € 644,000/year.

3.1.7. Orangutan habitat

BOSF released a total of 64 orangutans during 2012 and 2013. The reintroduction costs vary depending on the distance between the rehabilitation center and the release area. The average cost for orangutan reintroduction is € 7300/orangutan, which, based on an orangutan's average lifespan following release, is equivalent to € 183/orangutan/year. We use this value to estimate the avoided costs of orangutan habitat in areas where the occurrence of wild orangutan were identified. The population size of wild orangutan in Central Kalimantan is about 33,000 individuals, 61% of which live in protected areas (Wich et al., 2008). The orangutan population and the estimate of avoided

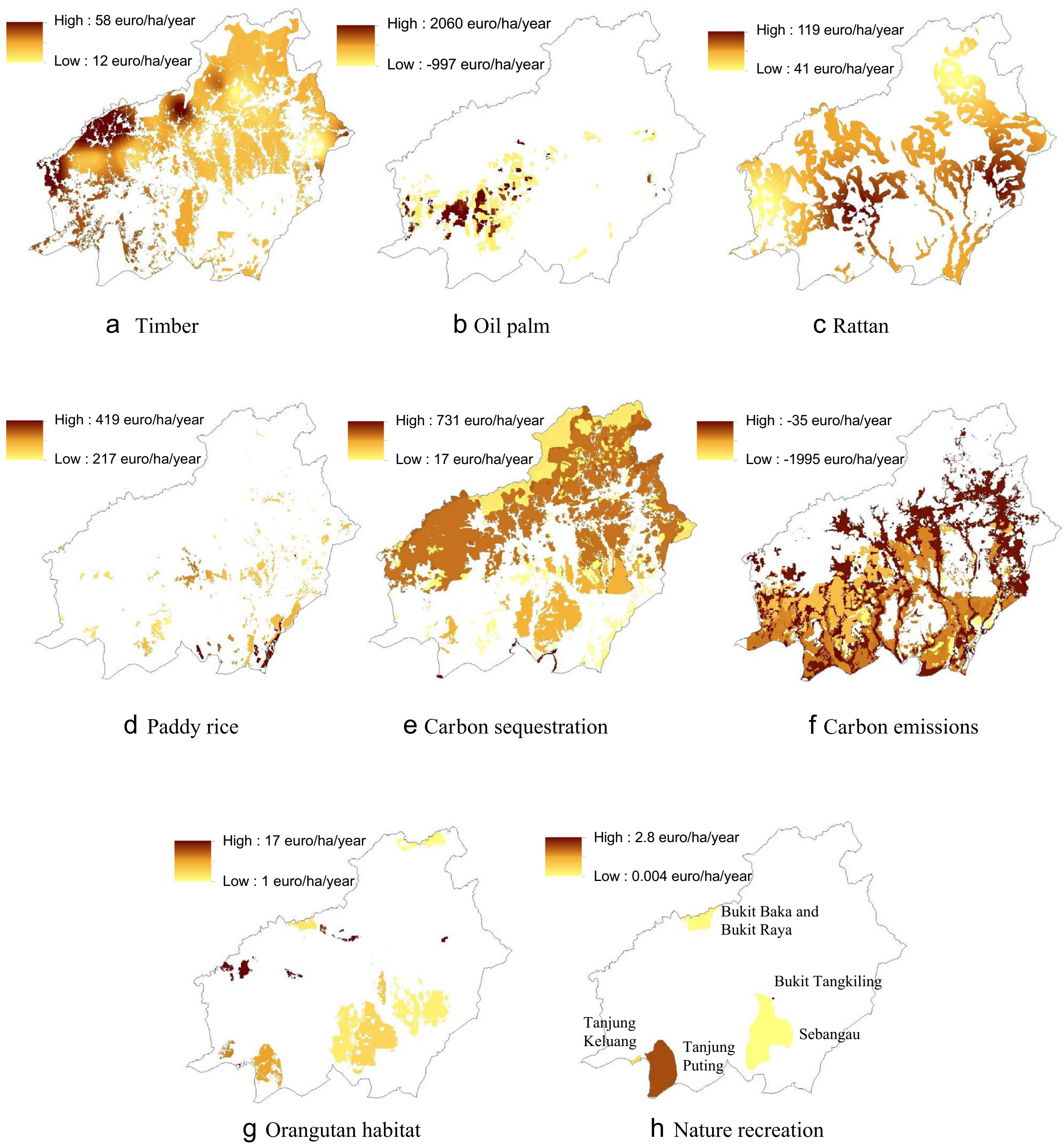

Fig. 2. Monetary value maps of ecosystem services.

Table 3
Monetary benefit from nature recreation in five parks.

Conservation parks	Number of visitors[a]		Revenue from entrance fees (€)	Revenue for tour organization (€)	Revenue for hotel (€)	Total revenue (€)
	Domestic	Foreigner				
Tanjung Puting National Park	2,300	8400	122,630	450,000	58,420	631,050
Bukit Baka & Bukit Raya National Park	10	80	320	3,860		4,180
Sebangau National Park	30	20	70	1,890		1,960
Tanjung Keluang Recreation Park	57,500	0	4,790			4,790
Bukit Tangkiling Recreation Park	33,000	0	2,750			2,750

[a] Classified into domestic and foreigner due to the difference of entrance fee.

costs in each location (habitat unit) are summarized in Appendix A. The monetary value map of orangutan habitat services is presented in Fig. 2g. Estimated in this way, the monetary value for the 33,000 wild orangutan is € 6336,000/year. Note that, as with all other value estimates that we provide, this does not represent the total economic value, since we exclude consumer surplus in our estimations in order to be consistent with accounting principles. In particular, the willingness to pay of the general public for remaining orangutan in the wild, in particular for the last populations, may far exceed the avoided release costs. Zander et al. (2014) investigated the visitors' willingness to pay to support orangutan conservation in Sarawak and found a value of € 2200/orangutan/year, which would be almost 12 times the value we estimated.

3.2. Monetary values of ES in different land cover units

Table 4 summarizes the monetary value of ecosystem services in the main land cover classes in Central Kalimantan. We distinguish between peat and mineral soils given the different implications of land use in these two soil types for ecosystem services supply, in particular with regards to carbon emissions. Note that our study is not complete, for instance we did not model hydrological services, other plantation crops (such as rubber) and other non-timber forest products. Also, there is substantial spatial variability in the trade-offs involved with land use conversion, in Table 4 we provide a provincial average value only. Note also that the resource rents of oil palm production are not extracted directly from Fig. 2b. In order to make them comparable with the other values, the values for oil palm production are the average resource rents of oil palm production during one cycle of production (20 years), with the annual values not discounted.

4. Discussion

4.1. Monetary valuation and mapping of ecosystem services in support of accounting

In this paper we present a crucial part in the development of an ecosystem account: an analysis of the monetary values of ecosystem services, both in spatial terms and in terms of an overview of the overall flow of ecosystem services generated in Central Kalimantan. In spite of Central Kalimantan being a generally data-poor environment, we managed to model seven ecosystem services both in physical terms (Sumarga and Hein, 2014) and in monetary terms (this paper). Critical in applying an accounting approach to ecosystem services mapping and valuation are: (i) distinguishing flows and assets; (ii) the use of appropriate physical and monetary assessment techniques; and (iii) sufficient accuracy and awareness of the limitations of the approach (European Commission (EC) et al., 2013). We will discuss these aspects below, focusing on the monetary aspects.

4.1.1. Distinguishing flows and assets

This paper focuses on the valuation of flows of ecosystem services. Analyzing ecosystems in terms of assets requires an additional analytical step. The ecosystem asset comprises two main components: the capacity of land cover units to generate ecosystem services and the condition of the ecosystem. The capacity reflects the maximum quantity of services an ecosystem can provide under current management (European Commission (EC) et al., 2013; Schröter et al., 2014), akin to the concept of theoretical ecosystem service flow as defined by Bagstad et al. (2014). For provisioning services, capacity depends upon the stock of ecosystem assets (e.g. standing stock of timber) and the regrowth of the ecosystem stock (e.g. mean annual increment of timber volume). For regulating and cultural services, capacity depends upon ecosystem processes and properties (e.g. riparian vegetation reducing flood risks), and the service materializes as soon as people are benefitting from the service (e.g. by having properties in the flood zone), cf. Bagstad et al. (2014). In the case of Central Kalimantan, the future flow of some services can be expected to differ substantially from the present flow. For instance, in the case of oil palm cultivation on peatlands, peat drainage leads to soil subsidence (Wösten et al., 2008; Hooijer et al., 2012), and – in the lowland environment of Central Kalimantan – to an increase in flood risks in the medium to long term (several decades) depending upon drainage depth and local hydrological conditions.

4.1.2. Physical and monetary assessment techniques

In a national accounting context, resource rent is an appropriate indicator for the monetary value of provisioning services (Campbell and Haynes, 1990; European Commission (EC) et al., 2013). We also applied it to the tourism service, calculating the resource rent approach to analyze the net revenue generated by the ecosystem service, which is providing opportunities for recreation. The resource rent approach deducts from the gross revenue in the sector all human and capital costs. By using a market interest rate to calculate the user costs of capital, the risks associated with economic activities are also

Table 4
The mean and, in brackets, standard deviation of the monetary value of ES in different land cover classes. Mean and standard deviations represent the average and the spatial variability of the values.

ES	Natural forest		Oil palm plantation		Rattan field	Paddy field	
	Dry land	Peat land	Dry land	Peat land		Dry land	Peat land
Timber (€/ha/year)[a]	28 (6)	27 (3)					
FFB of oil palm (€/ha/year)			1293	1094			
Rattan (€/ha/year)					82 (15)		
Paddy rice (€/ha/year)						290 (50)	283 (36)
Carbon sequestration in areas with net sequestration (€/ha/year)	269 (103)	214 (8)[d]	170[f]			176	
Carbon emissions in areas with net emissions (€/ha/year)		−392 (38)[e]	−20[g]	−2042			−1144
Orangutan habitat (€/ha/year)[b]	4 (3)						
Nature recreation (€/ha/year)[c]	0.6 (0.8)						

[a] In production forest.
[b] In areas suitable for orangutan habitat, mostly in protected areas.
[c] In protected areas.
[d] In primary forests.
[e] In drained forests.
[f] Established on degraded grassland.
[g] Established on forests, excluding carbon emission from land clearing.

accounted for. However, an element that is not included is the entrepreneurial reward (Carter, 2011), i.e. the reward for creating a business opportunity. This reward is in practice very difficult to calculate and it is also not explicit in the SNA 2008 (European Commission (EC) et al., 2009). By assuming it is zero there may be an overestimate of the resource rent attributed to ecosystems. The entrepreneurial risk and reward may be highest in innovative, immature business sectors, which do not include most of the sectors that we analyzed in our paper (rice production, rattan production, oil palm production), with the potential exception of nature tourism that is still a relatively new sector in Central Kalimantan.

A range of valuation approaches have been proposed for ecosystem services that have no market prices (Boyer and Polasky, 2004; Bateman et al., 2011). Our study shows the applicability of damage costs and defensive expenditures approaches in support of ecosystem accounting (see also Brouwer et al., 2009). In line with the SNA (2008), orangutan habitat is valued 'at cost', which is the general approach prescribed in the SNA for valuing public services. Valuation of the carbon sequestration service remains prone to considerable uncertainty given the high uncertainties related to estimating the social damage costs of carbon (Anthoff and Tol, 2013). In recent papers, efforts are being made to include the effects of low-probability, high-impact effects (such as surpassing thresholds in the climate system) in the social costs of carbon (e.g. Dietz, 2011). In addition, there is an issue of the discount rate to use, which has a major effect on the SCC given the long time frame of the impacts of climate change (Guo et al., 2006). Based on the National accounting guidelines a case could be made for the use of market discount rates, however we argue that for the analysis of public services in the context of ecosystem accounting a social discount rate is more appropriate (we use an SCC based on a discount rate of 3% in our paper). This aspect needs further discussion in the context of the statistical community (see also European Commission (EC) et al., 2013).

4.1.3. Accuracy

This study notes that the effectiveness of ecosystem accounting (in a spatial context) is driven by a combination of the availability of data and the applied mapping methods. Lack of data will potentially lead to poor estimates through, among others, generalization. We have ample valuation data for rattan, paddy rice, and nature recreation using a variety of sources. In case of orangutan habitat, only three estimates of reintroduction costs are available (reintroductions happened only three times in the last couple of years). In addition, we experienced a lack of valuation data for timber and oil palm production, since financial information is typically confidential for private timber and oil palm companies. We expect that a better understanding of accuracy levels can be obtained in more data rich environment, through a better understanding of standard deviations in the factors determining the resource rent and other ecosystem service values.

In the spatial analysis of ES values, a key issue in mapping ecosystem values is the generalization error, in particular when a benefit transfer approach is used (Plummer, 2009; Liu et al., 2010). This method assumes the similarity of values in a specific land cover type, regardless of the difference in locations and the spatial variability within the mapping unit. This study shows how our mapping approaches are capable of reducing generalization error in three ways. First, by using empirical data from surveys and studies within Central Kalimantan, both in mapping ecosystem services in term of physical quantity (Sumarga and Hein, 2014) and in monetary valuation. In this way the potential error from transferring values can be minimized. Second, exhibiting the spatial variation of ecosystem services inside a land cover type (for timber, rattan, and paddy rice) by applying interpolation. We applied this approach in mapping ecosystem services in term of physical quantity; hence, this variation was maintained in the monetary value maps. We did not have sufficient data to also deal with the spatial variability in the monetary values, even though this is relevant for ecosystem accounting. For instance, the labor costs of rattan harvest increase with the distance from the river (since rattan has to be manually carried to the river side where it is loaded onto boats). Such enhanced details are highly relevant for the next steps in developing monetary ecosystem accounts. Third, by detailing the mapping units, through breaking down land cover types into sub land cover types (for carbon sequestration) and modelling habitat suitability (for orangutan habitat).

4.2. Challenges in valuation and integrating ecosystem services values in an accounting framework

Our study indicates a number of issues that require further research before a standardized approach to the mapping of ecosystem service values for the purpose of accounting can be developed.

4.2.1. Valuing perennial crops

The production of perennial crops such as oil palm depends on the age of the crop. The costs are high in the first years, and there is no revenue due to zero production. Building upon the SNA (2008), European Commission (EC) et al. (2009) suggest that the value of immature crops is attributed to the closeness to harvest, and considered to be work-in-progress (European Commission (EC) et al., 2009). The value of harvested crop is actually the accumulated value of the work-in-progress. This approach requires a comprehensive, multi-year analysis of crop production, which is in many cases based on estimates of how data can be broken down for individual years of the production cycle. In our approach, we analyzed the value of oil palm production in 6 classes, but the overall approach to value perennial crops still needs to be further discussed and agreed upon in the ecosystem accounting community.

4.2.2. Valuing wildlife habitat

We explored an innovative approach in order to test if this is feasible for the valuation of habitat and to obtain an idea of the order of magnitude of the values resulting from this approach. We focus only on the value of wildlife habitat for one species, orangutan. Orangutan is an endangered species (IUCN, 2013), and is the only remaining Asian great ape that is distributed only in Borneo and Sumatera (Nelleman et al., 2007). Its status attracts global attention for preservation; this allows us to value orangutan habitat through the costs of the release program. However, there is a large variety of wildlife in Central Kalimantan, including many IUCN Red List species such as Malayan sun bear (*Helarctos malayanus*), gibbons (Hylobates sp), maroon leaf monkey (*Presbytis rubicunda*), Sunda pangolin (Manis javanica), Sunda slow loris (*Nycticebus coucang*), Horsfield's tarsier (*Tarsius bancanus*), and Great argus (*Argusianus argus*). Hence, our valuation – even though in principle aligned with the SNA valuation principles – grossly underestimates the value of wildlife habitat by not considering all these other species as well as the value of the ecosystems as a whole. No reintroduction programs exist for these other species in Kalimantan and it seems unlikely that there are ecosystems on the planet for which there are reintroduction programs for all or most species. Hence it appears as if our species-based approach would not be suitable for scaling up. A question is if the same valuation principle, valuing biodiversity through the costs of rehabilitation, could be deployed at the ecosystem scale, in other words if the costs of ecosystem rehabilitation programs could be used as an indication of the SNA-conform value of the biodiversity (including ecosystem, species and genetic diversity) contained in that ecosystem. This is certainly not correct from a welfare economics perspective (Jobstvogta et al., 2014; Zander et al., 2014), but the consistency of this approach with accounting principles deserves further attention (Turner et al., 2010; United Nations (UN) et al., 2014). For the time being, however, biodiversity accounts may need to be developed in physical units only, and suitable indicators for biodiversity such as species status, richness and abundance indices (Keeping, 2014; Shtilerman et al., 2014; Taft et al., 2014) need to be presented alongside monetary data from ecosystem accounts.

4.2.3. Integrating ecosystem services values in an accounting framework

It is relatively straightforward to integrate values from timber, oil palm, rattan, rice, and recreation in an aggregate measure such as GDP as the benefits to which they contribute are already within scope of the SNA production boundary. Ecosystem accounting allows us to separately identify these values and make the contribution of the ecosystem to economic activity visible. In case of carbon sequestration, the benefit lies outside the SNA production boundary and its valuation would lead to an adjustment of GDP (that is sometimes called a green(ed) GDP Boyd and Banzhaf, 2007). There is an ongoing development of green GDP in Indonesia as an indicator of sustainable development; this adjusted GDP involves the values of resource depletion, degradation and pollution (Gustami, 2012). Revealing resource depletion and degradation has always been an important motivation of ecosystem accounting development, particularly for resource-dependent countries (Repetto et al., 1989; Howarth and Farber, 2002). Implementing adjusted measures would be highly relevant for Central Kalimantan, where an adjustment could consist of both additions (e.g. carbon sequestration or habitat services) and deductions (due to either resource depletion or environmental degradation) in order to provide a more comprehensive insight in the costs and benefits of land use change including the rapid spread of oil plantations in different soil types.

4.2.4. Environmental assets valuation

In national accounting as in micro-economics, environmental assets are valued on the basis of present and future returns generated by the assets (European Commission (EC) et al., 2009). Besides estimating a path of future returns, other key inputs required for calculating the net present value (NPV) are an estimation of asset life, and the selection of a discount rate (European Commission (EC) et al., 2009). Analyzing the value of the environmental assets represented by different land cover units on the basis of the expected flow of ecosystem services (European Commission (EC) et al., 2013) is a next step in developing ecosystem accounts (Edens and Hein, 2013). Because we analyzed the production of palm oil over the production cycle of the oil palm, we are able to provide the NPV of the asset ‘oil palm plantation’ following accounting conventions. We calculate the NPV for oil palm on mineral and peat land (Appendix B). We used a production cycle of 20 years with a discount rate of 10% (equal to the interest rate we applied to calculate the user costs of fixed capital). Our analysis yields an environmental asset value of € 6596 per ha on mineral soil and € 4393 on peat soil, resulting from the resource rent generated by the production of FFB. Our average NPV for Central Kalimantan falls within the range reported by Budidarsono et al. (2012), which is € 3667–€ 24,583 at a discount rate of 8%, and by Butler et al. (2009), who report the NPV to vary between € 3196–€ 8025 at a discount rate of 10%. We carried out a sensitivity analysis for the discount rate used to analyze the value of palm oil production, using the discount rates of 8% and 12%. For mineral land, we find an NPV of € 8792 per ha for a discount rate of 8%, and € 4894 for a discount rate of 12% using the data in our model. For peat soil, these values are € 6349 per ha for a discount rate of 8% and € 2889 per ha for a discount rate of 12% respectively (see Appendix B). These values demonstrate the sensitivity of the NPV to the discount rate.

4.2.5. Completing the ecosystem accounts

This study analyses how different ecosystem services generated in Central Kalimantan province can be valued and mapped in a way that is in line with national accounts. There are a range of ecosystem services in Central Kalimantan that this study does not cover, such as other crop production services (e.g. hevea rubber, vegetables), other non-timber forest products such as jelutung (*Dyera costulata*), aquaculture and fisheries, flood control, erosion control, cultural practices, and habitat for other species. This study provides an important basis for accounting for a broader set of ecosystem services by exploring spatial patterns and valuation approaches. Nevertheless, further studies are required to develop an ecosystem account covering a more comprehensive suite of ecosystem services, and to analyze the capacity of the ecosystem to generate services in both physical and monetary terms (European Commission (EC) et al., 2013), building upon this paper as well as Sumarga and Hein (2014). Key challenges include the development of suitable accounting methods for hydrological services, and other significant cultural services in Central Kalimantan such as cultural heritage, landscape beauty, and scientific and educational information, where the existing valuation approaches tend to focus on measuring consumer surplus (e.g. van Berkel and Verburg, 2014); hence, they are not consistent with national accounting valuation principles.

4.3. Value trade-offs and policy implications

Central Kalimantan is one of few provinces in Indonesia that has not yet finalized its provincial land use planning. The discussion on land use planning in this province has been ongoing for about 11 years and started when the provincial spatial planning act (Provincial Legislation no 08, 2003) was first submitted to the central government for approval. The delay in approval is related to the conflicts of interests among the sectors depending on land, including the forestry sector, the agricultural and mining industries, and the district, provincial and central government (Galudra et al., 2011). A key issue pertains to the area designated as forests, in which any land conversion, including for oil palm plantations and mining, will be prohibited (Brockhaus et al., 2012).

Provided that valuation outcomes are robust and developed using a consistent approach, valuation allows analysis of value trade-offs related to land management. In the context of Central Kalimantan, the main land management issue is deforestation, particularly in relation to oil palm expansion. In the last decade, the deforestation and oil palm expansion rate in Central Kalimantan have been among the highest in Indonesia (Broich et al., 2011; Koh et al., 2011). In the period 2000–2010, about 933,000 ha of new oil palm plantation has been established, mostly by converting forests (514,000 ha) (analyzed from changes in the land cover map between 2000 and 2010, see Sumarga and Hein, 2014). The ecosystem services trade-offs as an implication of the conversion of forests into oil palm plantation are shown in Table 4.

Table 4 shows that conversion of peat forests into oil palm plantations results in a decrease in the overall value of the ES generated. Oil palm plantations require drainage of peatlands to a water table depth of 80 to 90 cm, resulting in carbon emissions of on average around 86 t CO_2/ha/year (Hooijer et al. 2010). This leads, for Central Kalimantan, to average annual societal costs of around € 2042/ha/year. The costs of carbon emissions can be compared with the benefits of palm oil production in two ways. First, they can be compared in terms of the annual costs of the carbon emissions versus the as average resource rent over the lifetime of the palm oil plantation, which is € 1094/ha/year (based on Table B1). Second, they can be compared in terms of the NPV of the carbon emissions versus the NPV of the resource rent of the oil palm plantations. At a 10% private discount rate, the NPV of the oil palm on peat soil is € 4393 per ha (for a 20 years discounting period). For an 8% discount rate the NPV is € 6349, and for a 12% discount rate, it decreases to € 2889 per ha. This NPV can be compared to the social costs of carbon emissions, recognizing the uncertainties in the marginal damage cost estimates of carbon emissions. Key sources of uncertainty in the social costs of carbon are the assumed damage costs resulting from climate change, the occurrence of low probability, high impact events, and the selected social discount rate (Guo et al., 2006; Tol, 2008; Nordhaus, 2011; Ackerman and Stanton, 2012). The SCC used in this study (€ 88/t C; equivalent to € 24/t CO_2) is an average value derived from three integrated assessment models: DICE, PAGE, and FUND based on a discount rate of 3% (Interagency Working Group on Social Cost of Carbon and United States Government, 2013). If the social discount rate is increased to 5%, the models indicate an average SCC of € 30/t C (Interagency Working Group on Social Cost of Carbon and United States Government, 2013), which reduces the costs of carbon emission from oil palm on peatland to € 702/ha/year. At a 3% social discount rate, the NPV of the carbon emissions for one production cycle of the oil palm plantations (20 years) is € 32,422 per ha, and at a 5% social discount rate it is € 11,146 per ha.

Hence, for commonly used private discount rates (8 to 12%) and social discount rates (3 to 5%) we find that the social costs of carbon emissions of oil palm plantation on peat far exceed the private benefits of palm oil cultivation, even without considering the impacts of plantation establishment on other ecosystem services (such as timber, NTFP and biodiversity). Our study confirms the recommendations of, among others, Wicke et al. (2008) and Hooijer et al. (2010) to stop further conversion of peat lands into oil palm plantations.

Note also that our study does not address long-term hydrological effects of peat drainage that include increased flood risks because much of the drained land will irreversibly subside in the coming decades (Hooijer et al., 2012) to a level where it is prone to frequent flooding making it impossible to continue growing oil palm. This may decrease the value of environmental assets in the coming decades. It is noteworthy that such long-term environmental trends, although very relevant for environmental management, are not sufficiently included in an ecosystem account. Follow up studies are required to further analyze asset value of oil palm development on peatland, particularly in relation to flood risk due to continuous soil subsidence. This points to the need to supplement ecosystem accounts with other types of information in order to have a sufficiently comprehensive basis for decision making on ecosystem use.

5. Conclusions

We valued and mapped seven ecosystem services in Central Kalimantan aligned with the principles of national accounting, using resource rents and costs approaches as principal valuation methods. Our value maps show the substantial spatial variation in the values of ES even at the level of a province, clearly indicating that any approach aimed to scale up ES values needs to consider the spatial heterogeneity of ES. For the seven services that we selected, valuation following an accounting approach proved feasible, and we recommend further testing of the monetary valuation of ES in a range of different contexts in order to develop an ecosystem accounting approach to measuring and monitoring ecosystem capital. In terms of valuing biodiversity/habitat services, we show that there are potential approaches that can be applied to analyze the value of this service in a manner that is in principle consistent with accounting, but that data limitations will restrict valuation possibilities. Our case study demonstrates a major policy application of monetary value maps, i.e. supporting land use planning. We show that the conversion of Indonesian peat land to oil palm plantations is highly inefficient from a societal perspective: the societal costs, in particular those related to greenhouse gas emissions following peat drainage, far exceed the private benefits. We conclude that ecosystem accounting including monetary valuation of ecosystem services flows and ecosystem assets – once fully developed and standardized – is a highly promising approach to support more sustainable and efficient ecosystem management.

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Appendix A. The avoided costs of orangutan habitat

Location*	Estimated population*	Estimated avoided costs (€)	Suitable area** (ha)	Estimated value (€/ha)
Bukit Baka and Bukit Raya NP	675	123,525	58,545	2.1
Tanjung Puting NP	6000	1098,000	188,679	5.8
Lamandau NR	1200	219,600	32,273	6.8
Mawas NR	3500	640,500	355,897	1.8
Sebangau NP (incl. Sebangau Kahayan)	7600	1390,800	454,937	3.1
Ketingan	3000	549,000	274,030	2.0
Rungan Kahayan	1000	183,000	46,093	4.0
Arut Belantikan	6000	1098,000	66,664	16.5
Seruyan	1000	183,000	13,308	13.8
Kahayan and Sambah	1000	183,000	16,571	11.0
Sambah and Katingan	500	91,500	9,730	9.4
Kahayan Kapuas	300	54,900	33,548	1.6
Tanjung Keluang NR	200	36,600	2,385	15.3
Pararawen NR	500	91,500	6,617	13.8
Sapat Hawung NR	500	91,500	92,072	1.0

* Based on Wich et al. (2008).

** Generated from Maxent modeling with an optimum threshold of 0.24.

Appendix B

See Tables B1 and B2.

Table B1
NPV of FFB production on peat soil, modified from Fairhurst and McLaughlin (2009) with a discount rate of 8%, 10%, and12%, values are rounded.

Year		0	1	2	3	4	5	6	7	8	9	10
FFB yield	t/ha	0	0	0	5	10	15	20	24	26	27	27
FFB price	€/t	123	123	123	123	123	123	123	123	123	123	123
Revenue	€/ha	0	0	0	613	1225	1838	2450	2940	3185	3308	3308
Planting and other farming costs	€/ha	949	1770	614	1092	639	639	639	639	639	639	639
Harvesting and transportation costs	€/ha	0	0	0	33	67	100	133	160	173	180	180
Depreciation cost	€/ha	0	0	0	83	165	165	165	165	165	165	165
Costs of fixed assets	€/ha	0	0	0	289	281	264	248	231	215	198	182
Total costs	€/ha	949	1770	614	1496	1151	1168	1185	1195	1192	1182	1166
Total costs 2010	€/ha	1083	2020	701	1707	1314	1333	1352	1364	1360	1349	1330
Resource rent	€/ha	−1083	−2020	−701	−1095	−89	505	1098	1576	1825	1959	1977
PV (discount rate 8%)	€/ha	−1083	−1870	−601	−869	−65	343	692	920	986	980	916
PV (discount rate 10%)	€/ha	−1083	−1836	−579	−823	−61	313	620	809	851	831	762
PV (discount rate 12%)	€/ha	−1083	−1803	−559	−779	−56	286	556	713	737	706	637

Year		11	12	13	14	15	16	17	18	19	20	NPV
FFB yield	t/ha	27	27	27	27	27	26	25	24	23	22	
FFB price	€/t	123	123	123	123	123	123	123	123	123	123	
Revenue	€/ha	3308	3308	3308	3308	3246	3185	3063	2940	2818	2695	
Planting and other farming costs	€/ha	639	639	639	639	639	639	639	639	639	639	
Harvesting and transportation costs	€/ha	180	180	180	180	177	173	167	160	153	147	
Depreciation cost	€/ha	165	165	165	165	165	165	165	165	165	165	
Costs of fixed assets	€/ha	165	149	132	116	99	83	66	50	33	17	
Total costs	€/ha	1149	1133	1116	1100	1080	1060	1037	1014	991	967	
Total costs 2010	€/ha	1311	1292	1274	1255	1232	1210	1183	1157	1130	1104	
Resource rent	€/ha	1996	2015	2034	2053	2014	1975	1879	1783	1687	1591	
PV (discount rate 8%)	€/ha	856	800	748	699	635	577	508	446	391	341	6349
PV (discount rate 10%)	€/ha	700	642	589	541	482	430	372	321	276	237	4393
PV (discount rate 12%)	€/ha	574	517	466	420	368	322	274	232	196	165	2889

Table B2
NPV of FFB production on mineral soil, modified from Fairhurst and McLaughlin (2009) with a discount rate of 8%, 10%, and 12%, values are rounded.

Year		0	1	2	3	4	5	6	7	8	9	10
FFB yield	t/ha	0	0	0	5	10	15	20	24	26	27	27
FFB price	€/t	123	123	123	123	123	123	123	123	123	123	123
Revenue	€/ha	0	0	0	613	1225	1838	2450	2940	3185	3308	3308
Planting and other farming costs	€/ha	949	959	614	1092	639	639	639	639	639	639	639
Harvesting and transportation costs	€/ha	0	0	0	33	67	100	133	160	173	180	180
Depreciation cost	€/ha	0	0	0	41	83	83	83	83	83	83	83
Costs of fixed assets	€/ha	0	0	0	144	140	132	124	116	107	99	91
Total costs	€/ha	949	959	614	1311	929	954	979	997	1002	1001	992
Total costs 2010	€/ha	1083	1094	701	1496	1060	1088	1117	1138	1144	1142	1132
Resource rent	€/ha	−1083	−1094	−701	−883	165	749	1333	1802	2041	2166	2175
PV (discount rate 8%)	€/ha	−1083	−1013	−601	−701	122	510	840	1052	1103	1083	1007
PV (discount rate 10%)	€/ha	−1083	−995	−579	−663	113	465	753	925	952	918	839
PV (discount rate 12%)	€/ha	−1083	−977	−559	−629	105	425	675	815	824	781	700

Year		11	12	13	14	15	16	17	18	19	20	NPV
FFB yield	t/ha	27	27	27	27	27	26	25	24	23	22	
FFB price	€/t	123	123	123	123	123	123	123	123	123	123	
Revenue	€/ha	3308	3308	3308	3308	3246	3185	3063	2940	2818	2695	
Planting and other farming costs	€/ha	639	639	639	639	639	639	639	639	639	639	
Harvesting and transportation costs	€/ha	180	180	180	180	177	173	167	160	153	147	
Depreciation cost	€/ha	83	83	83	83	83	83	83	83	83	83	
Costs of fixed assets	€/ha	83	74	66	58	50	41	33	25	17	8	
Total costs	€/ha	984	976	968	959	948	936	921	906	892	877	
Total costs 2010	€/ha	1123	1114	1104	1095	1082	1068	1051	1034	1017	1000	
Resource rent	€/ha	2184	2194	2203	2213	2165	2117	2011	1906	1800	1695	
PV (discount rate 8%)	€/ha	937	871	810	753	682	618	544	477	417	364	8792
PV (discount rate 10%)	€/ha	766	699	638	583	518	461	398	343	294	252	6596
PV (discount rate 12%)	€/ha	628	563	505	453	395	345	293	248	209	176	4894

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