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Can intensification reduce emission intensity of biofuel through optimized fertilizer use? Theory and the case of oil palm in Indonesia

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Abstract

Closing yield gaps through higher fertilizer use increases direct greenhouse gas emissions but shares the burden over a larger production volume. Net greenhouse gas (GHG) footprints per unit product under agricultural intensification vary depending on the context, scale and accounting method. Life cycle analysis of footprints includes attributable emissions due to (i) land conversion ('fixed cost'); (ii) external inputs used ('variable cost'); (iii) crop production ('agronomic efficiency'); and (iv) postharvest transport and processing ('proportional' cost). The interplay between fixed and variable costs results in a nuanced opportunity for intermediate levels of intensification to minimize footprints. The fertilizer level that minimizes the footprint may differ from the economic optimum. The optimization problem can be solved algebraically for quadratic crop fertilizer response equations. We applied this theory to data of palm oil production and fertilizer use from 23 plantations across the Indonesian production range. The current EU threshold requiring at least 35% emission saving for biofuel use can never be achieved by palm oil if produced: (i) on peat soils, or (ii) on mineral soils where the C debt due to conversion is larger than 20 Mg C ha^{-1}, if the footprint is calculated using an emission ratio of N_2O–N/N fertilizer of 4%. At current fertilizer price levels in Indonesia, the economically optimized N fertilizer rate is 344–394 kg N ha^{-1}, while the reported mean N fertilizer rate is 141 kg N ha^{-1} yr^{-1} and rates of 74–277 kg N ha^{-1} would minimize footprints, for a N_2O–N/N fertilizer ratio of 4–1%, respectively. At a C debt of 30 Mg C ha^{-1}, these values are 200–310 kg N ha^{-1}. Sustainable weighting of ecology and economics would require a higher fertilizer/yield price ratio, depending on C debt. Increasing production by higher fertilizer use from current 67% to 80% of attainable yields would not decrease footprints in current production conditions.

Keywords: biofuel policy, carbon emission, fertilizer price, intensification, land sparing/sharing, net emission saving, palm oil

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Introduction

The Borlaug or 'land sparing' hypothesis (Sanchez, 1994; Rudel *et al.*, 2009) states that intensifying agriculture will have net positive effects on the environment, regardless of any directly negative environmental impacts of 'green revolution' production technology, as it reduces the land base needed to meet world market demand for agricultural products (food, fibre and fuel) and thus reduces biodiversity loss (Green *et al.*, 2005) as well as emissions from deforestation and forest degradation. The 'land sharing' or ecological agriculture hypothesis that forms its counterpart suggests that a careful balancing of productivity and environmental services in integrated production systems can contribute to multifunctionality of integrated landscapes that is superior to the 'segregated' agriculture-*plus*-forest perspective of the Borlaug hypothesis (van Noordwijk *et al.*, 1995; Tomich *et al.*, 1998; Angelsen & Kaimowitz, 2001; Lee & Barrett, 2001). Choices for an optimal level of intensification may depend on location ('theory of place') (van Noordwijk *et al.*, 2015), type of environmental services considered (Grau *et al.*, 2013) and scale (Minang *et al.*, 2015). As currently framed (Minang & van Noordwijk, 2013), the sparing *plus* sharing debate considers the wider policy context that is needed to turn a 'necessary' to a 'sufficient' condition: environmental issues and deforestation cannot be resolved without an increase in yield levels that exceeds the growth in global demand for food, fibre and energy. However, it is naive to expect markets to directly effectuate environmental benefits through a pathway of reducing the profitability of less-efficient production modes. Both 'sparing' and 'sharing' approaches will only achieve environmental

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benefits if the opportunity for such benefits is utilized in active 'caring' approaches (Jackson *et al*., 2012). As earlier assumptions about direct links between yield and the efficiency gap are not supported by evidence (van Noordwijk & Cadisch, 2002; van Noordwijk & Brussaard, 2014), there is space for intermediate intensity solutions to be optimal from a societal perspective. Regulation of land use, however, cannot easily incorporate the fine-tuning needed to minimize environmental effects of land use change (Lambin *et al*., 2014). Pricing of input costs deserves further analysis as possible fine-tuning policy instrument. We here provide a quantitative analysis of intermediate optimum intensification levels, applicable to biofuels as costs and benefits can both be expressed in terms of net greenhouse gas emissions. For nonbiofuel crops, a similar analysis will require further steps to bring alternative options onto a single denominator.

In the biofuel debate, the interest has shifted from single characteristics of feedstock types (e.g. comparing soybean, palm and *Jatropha* oil), to recognition of the management swing potential (Davis *et al*., 2013; Creutzig *et al*., 2015) where the footprint of any feedstock depends on where and how it is produced as much as on what crop it is. The widest swing potential, according to current data, exists for palm oil, with both the best and worst emission intensities per unit product. Oil palm (*Elaeis guineensis* Jacq.) (Corley & Tinker, 2016) expansion is a 'Pandora's box' example (Tomich *et al*., 1998) of intensified tree crop production that attracts new activities in the tropical forest margins and increases forest conversion rather than reducing it. In public debate, oil palm expansion is held responsible for much of the loss of biodiversity and flagship species, but also as a cause of increased greenhouse gas emissions (Sheil *et al*., 2009). The yield gap in oil palm production is considerable for large-scale plantations and even larger for smallholder production systems (L.S Woittiez 2016), indicating a land equivalent ratio of below 1.0. Existing self-regulation in the industry is based on recommended 'good agricultural practice' without quantification of existing yield and efficiency gaps (von Geibler, 2013). There is little clarity on the level of fertilizer use that is considered good practice, from both a farm-level profitability and an environmental perspective.

The irony of biofuel use increasing rather than decreasing net anthropogenic greenhouse gas emissions in the 'biofuel boom' of the 2000s has led to a rapid regulatory response. A review of recent life cycle assessment (LCA) studies (van der Voet *et al*., 2010) showed considerable variation in outcomes, due to real-world differences, data uncertainties and methodological choices. If fossil fuel is partially substituted by 'biofuel', there are costs as well as benefits in terms of greenhouse gas emissions (Wicke *et al*., 2008). European regulation of the minimum emission reduction factors compared to the use of fossil fuel (emission saving) due to biofuel use in the Renewable Energy Directive of 2008 (European Communities Commission, 2008) has drawn attention to three types of emission costs (Hoefnagels *et al*., 2010): (i) the carbon debt due to land conversion from higher to lower time-averaged C stock, (ii) emissions associated with the production phase of the biofuel, part of which are on-site and part in the industry producing the inputs used, and (iii) emissions due to processing and transport. For emissions associated with the production phase, the issue of 'optimum intensification' levels is relevant (Jackson *et al*., 2012): Is there any merit in not fully utilizing the biophysical production opportunity of land that is already in agricultural use by moderating the use of fertilizer and similar inputs? Are current costs of fertilizer and other inputs sufficient to induce their wise and efficient use and low enough to allow 'environmentally optimum' levels of intensification?

The relationship between agricultural yields, environmental impacts of production and optimized use of inputs has been debated since the 1980s or earlier (van Noordwijk & de Willigen, 1986; de Wit, 1992; Zoebl, 1996). Increased yields increase the denominator of an efficiency (output/input) metric, but increased environmental impacts increase the numerator; the outcome depends on the shape of the yield and environmental impact response curves. The shape of these response curves themselves depends on factors such as the within-field spatial variability (Cassman & Plant, 1992; van Noordwijk & Wadman, 1992) and the degree of 'precision farming' adjustment of input levels to patch-level production conditions (Heege, 2015). de Wit (1992), showed that in the presence of multiple yield-limiting factors, the overall response of yield to aggregated input levels (or their associated environmental consequences) can be multiphasic. Neither of the extreme positions in the agriculture–environment debate ('Optimize yields economically and environmental impacts per unit product will be minimal' or 'Minimize inputs to maximize efficiency and minimize environmental impacts per unit product') are tenable as generalizations (Wicke *et al*., 2008). High yield levels can be achieved in combination with low and high efficiency; high efficiency can be coupled to low and high yields (van Noordwijk & de Willigen, 1987). The relationship between efficiency and yield depends on the finer details of the yield and environmental impact responses to input use, requiring empirical study for each crop and its specific physiology and agronomy (Corley *et al*., 1971).

Fertilizer subsidies have a long history in developing countries as a part of policies to intensify agriculture and maintain affordable staple food provisioning to urban people. Attempts to segment the markets and subsidize fertilizer only for certain crops or types of farmers are hard to implement, as regional or local fertilizer markets function well. If fertilizer prices are (too) low, efficiency enhancement is not economical (van Noordwijk & Scholten, 1994); if they are (too) high, the economical optimum solution may well be the near-complete mining of the soil (van Noordwijk, 1999). The relevance of shifting net fertilizer subsidies towards net taxation has been debated as a measure to reduce negative environmental effects of agricultural production through groundwater and surface water pollution or emission of N_2O, a powerful greenhouse gas. van Noordwijk & Wadman (1992) defined an environmentally optimum fertilizer level by reference to tolerated levels of nitrate enrichment of ground and surface water in the Netherlands; for potential biofuels, this target can be replaced by minimizing the emission footprint.

We will here focus on the relationships between N fertilizer use and the greenhouse gas emissions per unit biofuel use, relative to the fossil fuel use it can substitute for, using oil palm production in Indonesia as case study. Data from 23 plantations across the Indonesian production conditions (Khasanah *et al.*, 2012, 2015a,b) provided insights into what is currently considered good agronomic practice, as participation in the survey was voluntary. We will provide an algebraic analysis of the problem in generic terms and then review the available quantitative data. Key policy-relevant questions are as follows:

1. Is there an 'environmental optimum' production level at which net emission savings per unit biofuel use are maximized?
2. At what fertilizer/product price ratio is the 'economic optimum' fertilization rate equal to the 'environmental optimum' that minimizes attributable emissions?
3. How do the answers to questions 1 and 2 depend on the overall emissions from the life cycle analysis: (i) C debt due to initial land conversion, (ii) CO_2 emissions due to fertilizer production, (iii) the N_2O emission factor per unit of fertilizer used, and (vi) the technical coefficients for emissions due to transport and processing?
4. Are current policies for fertilizer subsidies and taxation aligned with environmental efficiency?

Theory

Four production phases contribute to emission estimates of biofuel production in a life cycle analysis (Fig. 1): (i) carbon debt (positive in all cases where the preceding vegetation had higher C stock than the oil palm plantations themselves) and additional emissions due to conversion (e.g. use of fire on peat soils), (ii) production of external inputs, such as inorganic fertilizer, (iii) feedstock production that determines the yield per ha that relates area-based terms to product-based accounting, but that may also lead to a change in belowground C stocks, recurrent GHG emissions related to drainage and/or N_2O emissions due to fertilizer use, and (iv) transport and processing stages before the product reaches the end users. A detailed scheme to estimate biofuel production's net emissions and emission savings is provided in the supporting information [The Biofuel Emission Reduction Estimator Scheme (BERES)], with some key parameter values that are considered 'defaults' based on measurement and literature review.

The shape of the response curve describing yield as function of fertilizer input has been much debated in the literature, with many empirical results converging on a Mitscherlich curve with asymptotic approach of a maximum yield. de Wit (1992) posed that the diminishing returns interpretation of Mitscherlich curves disappears when multiple constraints are addressed simultaneously. van Noordwijk & Wadman (1992) explored how empirical Mitscherlich-type curves can be interpreted as the result of spatial variability at field-level and patch-level responses by the crop that can be described by a quadratic equation, with a maximum that can be obtained or exceeded in practice. Quadratic models represent the most optimistic perspective on nutrient use efficiency at crop level with minimum field-scale variability. We use them here and will revert to the validity of this assumption in the Discussion section.

A quadratic fertilizer (N) yield (Y) response curve ($Y = Y_0 + f\,N + c\,N^2$) has three parameters (Y_0, f and c), corresponding to the yield without fertilizer use, the initial efficiency of fertilizer use and a parameter that combines f and the maximum attainable yield, respectively. Net annual emissions per unit crop yield are $E_0 + e_0 + e_f\,N + \varepsilon\,Y$ with the parameter E_0 or annualized attribution of the C debt representing phase I, e_0 (emissions independent of fertilizer use) phase III, e_f (proportional to fertilizer use) phases II and III and ε (proportional to yield) phase IV.

The emissions per unit production have a local minimum (hence, the emission savings compared to fossil fuels use a local maximum in case of a biofuel crop) when the N fertilizer rate equals (as derived in Appendix S1):

$$N_{minem} = R(\{1 + (1 - Y_0/(fR))/B\}^{0.5} - 1)$$

$$R = (E_0 + e_0)/e_f$$

$$B = fR/(Y_{max} - Y_0) = (E_0 + e_0)/\{(e_f/f)(Y_{max} - Y_0)\}$$

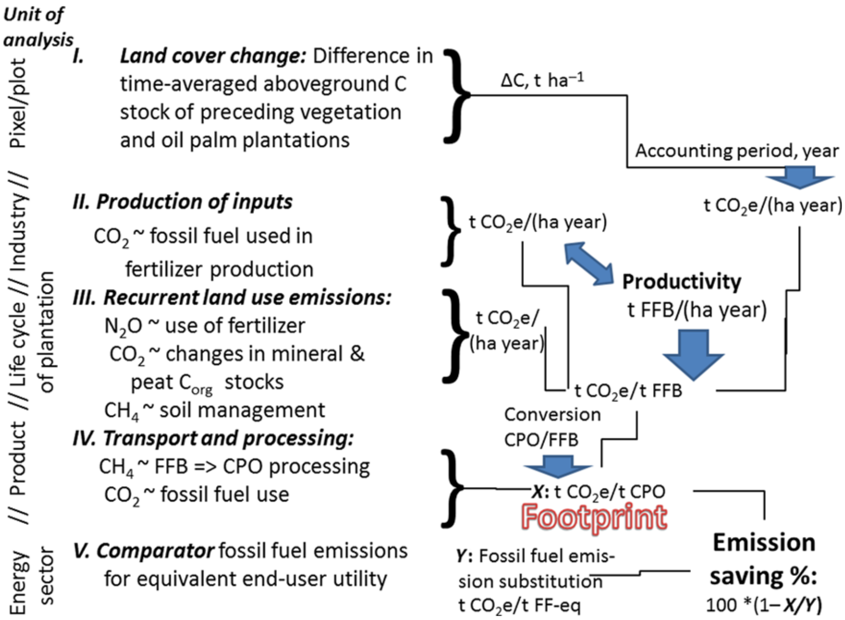

Fig. 1 Information flow in an assessment of the emission footprint per unit palm oil, and subsequent step to estimate the percentage emission saving in biofuel use.

where R and B are intermediate terms, f is initial marginal yield increment per unit of fertilizer used, E_0 is attributable CO_2eq emissions per ha per year due to initial land conversion, e_0 is attributable CO_2eq emissions per ha per year in the production stage at zero fertilizer use, e_f is attributable CO_2eq emissions per ha per year per unit fertilizer use in the production stage, Y_0 is the yield level in the absence of fertilizer use and Y_{max} is the maximum attainable yield under current circumstances beyond fertilizer use. The dimensionless B grouping is the ratio of the 'fixed cost' emissions $E_0 + e_0$ and the maximum of fertilizer-related emissions, $(Y_{max} - Y_0)(e_f/f)$. If p is the price ratio of yield products and fertilizer inputs, the economic optimum N fertilizer rate equals the fertilizer rate that minimizes emissions per unit yield, if $p = p_{SWEET}$ (SWEET = 'Sustainable Weighting of Ecology Economics Tradeoffs') (see Appendix S1):

$$p_{SWEET} = f(1 - 0.5B(\{1 + (1 - Y_0/(fR))/B\}^{0.5} - 1))$$

While this provides a generic answer to question 2, questions 1, 3 and 4 require parametrization for specific combinations of crop, attributable emissions from C debt and fertilizer use. Please note that the postharvest emissions (phase IV) represented in the term ε are not influencing the fertilizer rate that minimizes net attributable emissions and the outcome of the sparing vs. sharing debate. Phase IV emissions, however, can be an important determinant of the absolute level of emission attribution per unit final product and whether or not the overall footprint meets standards set.

Materials and methods

Sampling design

The IPOC/ICRAF survey of Indonesian palm oil production in 2010 was designed to estimate greenhouse gas emissions due to palm oil production across the major stratifying production factors in Indonesia (Khasanah *et al.*, 2015a,b). The three primary stratifiers of the survey were defined at national level as mineral vs. peat soils, plantations directly derived from forest or other land cover types and three levels of the prevalence of oil palm at provincial level (<1%, 1–5% and >5%). These act as indicators of the fact that the areas that first developed oil palm are probably most suited to it climatically and have the most advanced input and output markets. Not all 12 factorial combinations are important in practice, as oil palm on peat has mostly been directly derived from forest. The sampling design followed a stepwise cluster approach, soliciting self-nomination of companies to involve in learning the method while involving in data collection (Khasanah *et al.*, 2012, 2015a,b). Candidate companies were asked to describe land history, soil type and the scale of management (plantations, outgrowers, independent smallholders). A total of 23 plantations were selected for study, representing nine of the 12 clusters (Table 1). Figure 2 presents the spatial distribution of the selected samples by relative oil palm density in a province.

Data collection and analysis

In the 23 selected oil palm plantations, we collected the main parameters needed for the Biofuel Emission Reduction Estimator Scheme (BERES) (van Noordwijk *et al.*, 2013; see Appendix S2) to calculate the net emission of biofuel production and emission savings using a life cycle approach. The scheme is aligned with the way the EU RED policy requires life cycle data on the biofuel value chain. In the application, however, we did not use the 2008 'grandfather' rule that ignores C debts for land converted before the rules were made. It requires data for (i) C stock (Mg C ha^{-1}) of land cover preceding oil palm plantation (with the concept of time-averaged C stock applicable to rotations, and the current one to land cover types that are supposed to be in equilibrium), (ii) time-averaged C stock of the oil palm plantation, Mg C ha^{-1}, (iii) nitrogen (N) fertilizer level, kg N ha^{-1} and production level of fresh fruit bunches (FFB), Mg ha^{-1} yr^{-1}, (iv) oil extraction rate (OER) of crude palm oil (CPO) and kernel extraction rate, (v) soil CO_2 loss, (vi) emission factors due to fertilizer production and application and (vii) emissions due to postharvest commodity transport and processing before the product reaches the end-user (Germer & Sauerborn, 2008; Kamahara *et al.*, 2010; Wicke *et al.*, 2008; Alkabbashi *et al.*, 2009).

Biomass C stock of land cover preceding oil palm plantation. The 'time-averaged aboveground C stock' is the sum of the average over a production cycle of C pools (aboveground tree biomass, understorey vegetation and surface necromass). The belowground part of biomass is usually considered to be a proportion of aboveground biomass, with land cover-specific data hard to obtain. Data from the survey were used to establish estimates of the time-averaged aboveground C stock of oil palm plantation of around 40 Mg C ha^{-1}, as described in Khasanah *et al.* (2015b). The time-averaged aboveground C stock of forests and other preceding land cover types were assessed following the rapid carbon stock assessment (RaCSA) methodology and technical manuals (Hairiah *et al.*, 2011). Root biomass

Table 1 Sample distribution of oil palm plantations in the IPOC/ICRAF survey across preceding vegetation, soil type, oil palm prevalence in the surrounding province, and plantation management (Khasanah et al., 2015b)

Plantation parameters				Number of plantation or landscape	Plantation management*	Number of sampled plots per age category (year)			
Preceding land cover	Soil	Prevalence of oil palm (% of area in province)	Cluster			0–8	9–16	17–25	Total
Forest	Peat	5–15	1	2	N	2	2	4	8
					P	1	–	–	1
					I	1	–	–	1
		1–5%	2	2	N	4	–	–	4
					P	–	–	–	–
					I	–	–	–	–
		<1%	3	1	N	5	4	1	10
					P	–	1	–	1
					I	–	–	–	–
	Mineral	5–15	4	3	N	2	5	10	17
					P	–	2	2	4
					I	–	–	–	–
		1–5%	5	3	N	6	8	7	21
					P	1	2	–	3
					I	2	1	–	3
		<1%	6	9	N	16	20	7	43
					P	4	4	1	9
					I	10	2	–	12
Nonforest	Peat	5–15	7	–	–	–	–	–	–
		1–5%	8	–	–	–	–	–	–
		<1%	9	–	–	–	–	–	–
	Mineral	5–15	10	2	N	4	5	2	11
					P	–	–	–	–
					I	–	–	–	–
		1–5%	11	3	N	2	8	6	16
					P	4	6	3	13
					I	2	1	–	3
		<1%	12	–	–	–	–	–	–
Total				25		66	71	43	180

*N, nucleus; P, plasma; I, independent.

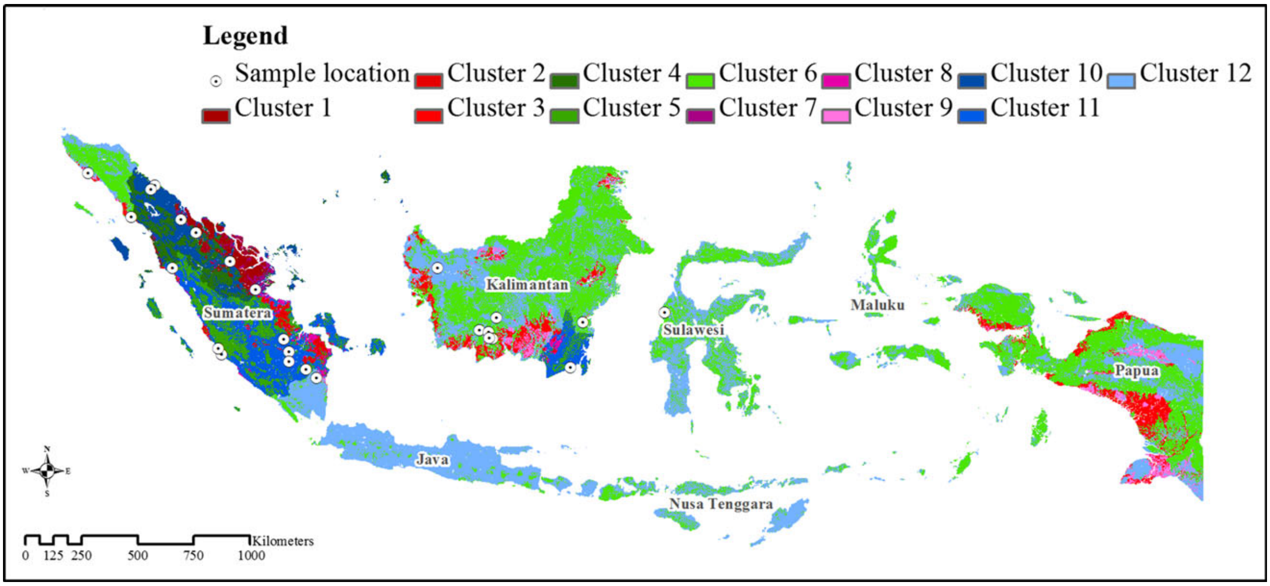

Fig. 2 Sample distribution of oil palm plantations in the IPOC/ICRAF survey.

was estimated as 25% of aboveground biomass for all land cover types. Identification of land cover type preceding oil palm used the analysis of land use and cover trajectory (ALUCT) protocols (Dewi & Ekadinata, 2013). Changes in time-averaged soil carbon stock when forest or other land cover types were converted into oil palm were analysed by Khasanah *et al.* (2015a). Khasanah *et al.*, 2015a,b discussed how life cycle inferences could be made for soil C_{org} and the oil palm biomass, respectively, despite the incomplete data for certain age classes in the various clusters (Table 1).

FFB and CPO production in relation to N Fertilizer level. The companies participating in the study provided time series data of their fertilizer use and production level of FFB across the age range of plantations under their management control. For each company, we developed a quadratic equation of FFB production (Y, Mg ha^{-1} yr^{-1}) as a function of age (years after planting; T, year) to estimate (by integration) the time-averaged FFB production level over the life cycle: $Y = a + b\ T + c\ T^2$. Total N input was calculated across the various fertilizer types reported. The data of N fertilizer application did not show clear correlations with the age of oil palm. Therefore, an average rate of N fertilizer application over the whole life cycle was calculated and used. Time-averaged yield (Y) was related to this average fertilizer rate by regression analysis for a quadratic response model [$Y = Y_0 + f\ N + c\ N^2$, with $c = -f^2/(4(Y_{max} - Y_0))$; see equation [5] in Appendix S1]. While a range of fertilizer types was reported, we focussed on the N content as basis for expected yield response, but used the most commonly used compound fertilizer (15-15-15) as a basis for fertilizer costs.

The companies also provided data on their CPO and kernel extraction rate. As variation in these two parameters was limited, an average value of CPO and kernel extraction rate extraction rates was calculated and used in the subsequent analysis.

Emission factors due to postharvest transport and processing. Emission factors due to postharvest transport and processing were based on fossil fuel use and technical design of the mills, and processing steps before the product reaches the end-user (Demirbas, 2007; Kamahara *et al.*, 2008; Wicke *et al.*, 2008; Alkabbashi *et al.*, 2009).

Sensitivity analysis. To understand the responses of emission saving to changes of carbon debt, N fertilizer application and ratio of N_2O–N/N fertilizer, a sensitivity analysis was carried out. Five carbon debts, 0, 20, 30, 40 and 60 Mg C ha^{-1}, were combined with N fertilizer applications in the range of 0–550 kg N ha^{-1}, with an interval of 5 kg N ha^{-1}. The Intergovernmental Panel on Climate Change's National Greenhouse Gas Inventory Guidelines suggest that the ratio of N_2O–N/N fertilizer is 1% (IPCC, 2006). Other literature suggests this can be 4% (Crutzen *et al.*, 2008). In the absence of site-specific measurements, both assumptions were compared for impact on the end result. The IPCC national greenhouse gas inventory framework (IPCC 2007; Bentrup & Pallière, 2008) includes the CO_2 emissions involved in fertilizer production under industrial processes, rather than land use sections. With a default value of 3.5 kg CO_{2eq} per kg N fertilizer, the net effect of these CO_2 costs of fertilizer production is less than the 4.65 kg CO_{2eq} per kg due to the associated N_2O emissions in land use for an N_2O–N/N fertilizer ratio of 0.01, with global warming effect of a molecule of N_2O calculated as 296 times that of a molecule of CO_2.

Results

Time-averaged aboveground C stock of land cover preceding oil palm

We found 21 types of land use systems surrounding the 23 oil palm plantations, which were further classified into three larger categories ‘forest’, ‘tree-based systems’ and ‘non-tree-based systems’ (Khasanah *et al*., 2012). The range of time-averaged aboveground C stock values was 150–250 Mg C ha^{-1} for ‘forest’, 50–150 Mg C ha^{-1} for ‘tree-based systems’ and less than 50 Mg C ha^{-1} for non-tree-based systems (Fig. 3). These figures were derived from 924 measured plots, 800 of which came from the ICRAF database of earlier studies in Indonesia.

Level of N Fertilizer and production of FFB and CPO

Based on a survey of 23 plantations throughout the oil palm production domain in Indonesia (Khasanah *et al*., 2012), we found an average fresh fruit bunch (FFB) yield of 18.8 Mg ha^{-1} yr^{-1} and an average fertilizer use of 141 kg N ha^{-1} yr^{-1} (Fig. 4 and Table 2). FFB yield and N fertilizer use were closely associated, with an apparent fertilizer response curve of $Y(\text{FFB}) = 8.23 + 0.0889\text{N} - 0.0001\text{N}^2$, with 74.2% of variance accounted for (Fig. 5a), suggesting $Y_0 = 8.23$ and $Y_{\max} = 27.98$ Mg ha^{-1} yr^{-1}. This apparent fertilizer response is derived from survey data, rather than randomized experiments. While we used the N fertilizer rate as basis for the regression, we can assume that other plant nutrients were provided in proportion and/or that residual variation in Fig. 5a is due to such factors. For oil yield (CPO) per ha, the relation was $Y(\text{oil}) = 1.47 + 0.0298\text{N} - 5\text{E-}05\text{N}^2$, with 70.2% of variance accounted for (Fig. 5b). With the average N fertilizer level reported, the yield is expected to be 67% of the maximum FFB yield that can, apparently, be obtained with existing germplasm and plantation management represented in the data set.

Emission saving and sensitivity analysis

A default estimate of 40 Mg C ha^{-1}(Khasanah *et al*., 2015b) of aboveground C stock and no mineral soil loss (Khasanah *et al*., 2015a) was used to estimate emission saving. When the preceding vegetation had a higher C stock (and conversion took place after the cut-off date of applicable standards, e.g. 2008 for the EU RED), the plantation started with a ‘carbon debt’. If preceding C stock was less, the calculation can reflect a net emission saving for the first production cycle. Rather than a single ‘typical’ value, the IPOC/ICRAF data set shows wide variation in C debt (phase I), yield levels and N fertilizer use (phases II and III). Our data support the conclusion that peatland emissions are off the scale and preclude attainment of the emission saving standards (Fig. 6) (Couwenberg *et al*., 2010; Maswar, 2011).

The curvature of the relationship between the level of N fertilizer and production of FFB plus the effect of a ‘fixed cost’ of C debt lead to an interesting shift in the shape and positions of the curves that relate the emission savings to the N fertilizer level in the production stage (Fig. 7). A net emission saving target of 35%

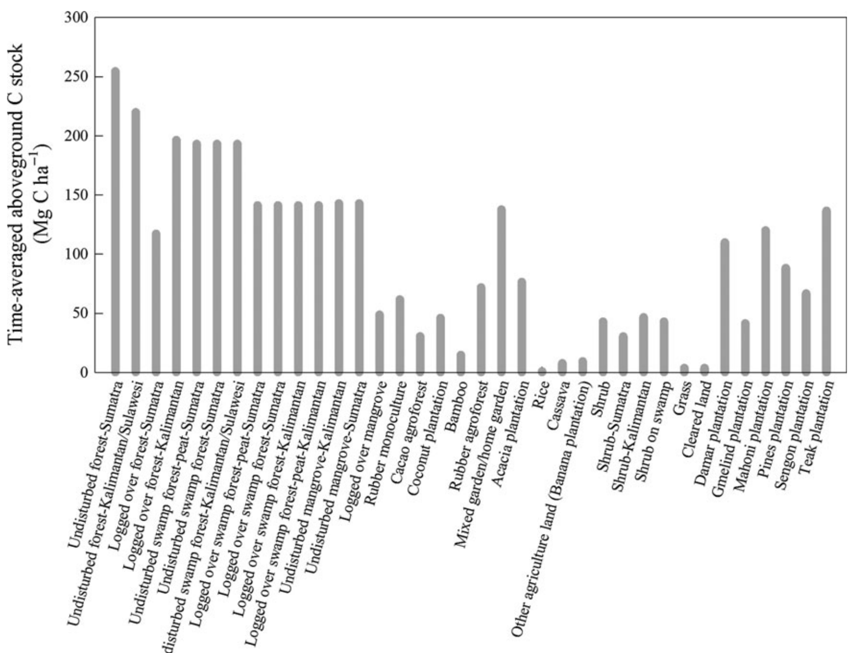

Fig. 3 Time-averaged aboveground C stock of other land uses involved in the plantations that were part of the IPOC/ICRAF survey.

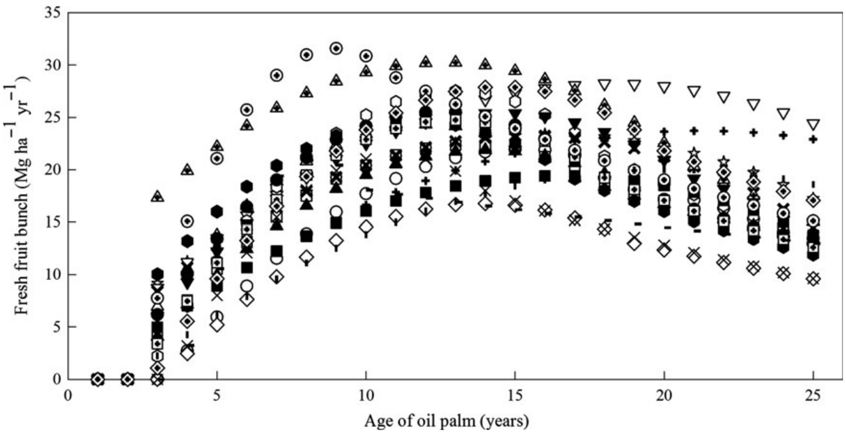

Fig. 4 Relationship between the age of the oil palm and fresh fruit bunch (FFB) production level as derived for the plantations that were part of the IPOC/ICRAF survey.

Table 2 Time-averaged N fertilizer application, yield level and oil extraction rate per plantation; plantation identity (ID) with 'a' refers to nucleus (plantation), 'b' to plasma (smallholders)

Plantation ID	N fertilizer*, kg N ha^{-1} yr^{-1}	FFB†, Mg ha^{-1} yr^{-1}	Kernel‡, %	OER of CPO‡, %	PKO§, %
001	144.96	18.30	5.16	23.63	0.5
002	121.87	19.43	4.24	24.07	0.5
005	110.49	18.16	4.97	23.97	0.5
006	91.09	15.64	4.29	20.36	0.5
007	251.87	23.01	4.75	22.48	0.5
008	124.31	16.71	4.75	20.54	0.5
010	66.98	11.77	4.87	19.92	0.5
011a	153.61	19.38	5.31	22.31	0.5
011b	151.76	18.84			0.5
013	127.83	16.51	5.73	23.02	0.5
014	139.55	18.70	5.74	24.01	0.5
015	127.91	19.47			0.5
016	113.65	15.44	4.86	23.15	0.5
017	104.39	18.49	4.80	23.99	0.5
018	257.38	24.41	4.90	23.62	0.5
019	109.84	17.76	3.87	22.49	0.5
020	163.52	22.22			0.5
021a	126.52	13.96			0.5
021b	137.07	14.46			0.5
022	76.75	14.76	4.35	22.86	0.5
023	178.45	20.25	4.24	21.68	0.5

*Time-averaged N fertilizer rates (over the life cycle, no available data for plantation ID 003, 004, 009 and 012, within emission saving estimation, default data then used (141 kg N ha^{-1} yr^{-1}).

†Time-averaged production rates (over the life cycle), no available data for plantation ID 003, 004, 009 and 012, within carbon footprint estimation, default data then used (18.8 Mg ha^{-1} yr^{-1}).

‡No available data for plantation ID 003, 004, 009, 012, 015, 020 and 021, within emission saving estimation, default data then used (23% for OER and 5% for kernel;

§PKO palm kernel oil; estimate based on Corley & Tinker, 2016.

cannot be achieved if C debt is more than 20 Mg C ha^{-1} for a N_2O/N fertilizer loss rate 4% and when C debt is more than 40 Mg C ha^{-1} for a N_2O/N fertilizer loss rate of 1%.

For many parameter combinations cases, there is a weakly defined 'optimum' N fertilizer level that maximizes the emission savings, within a rather broad range where emission savings vary less than 5%

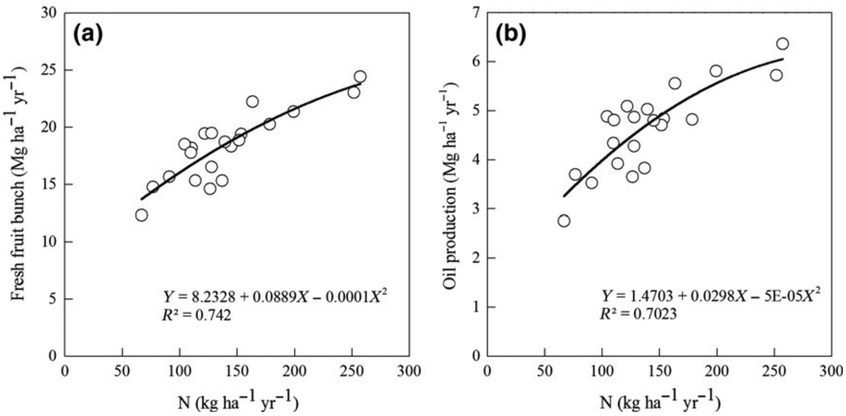

Fig. 5 Correlation between two properties assessed at life cycle level: the average yearly N fertilizer application and average yearly fresh fruit bunch (FFB) (a), the average yearly N fertilizer application and oil production (b).

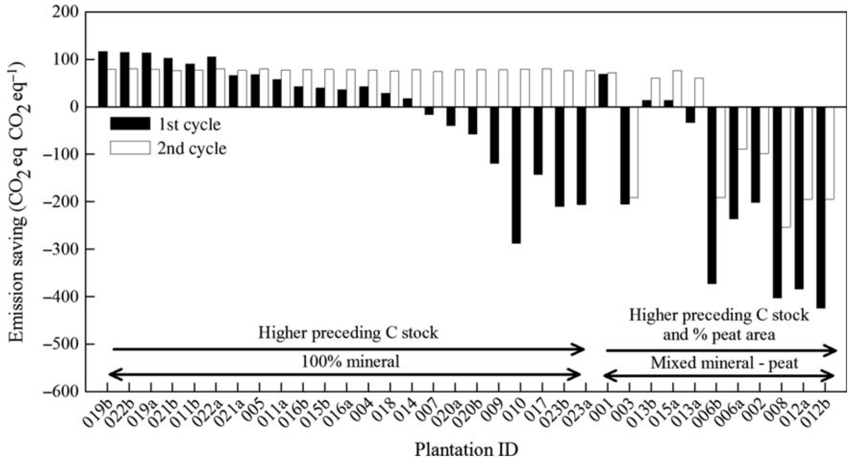

Fig. 6 Attributable emission savings in relation to preceding carbon stock and N fertilizer application; plantation identity (ID) with a refers to nucleus (company), b plasma (smallholders); C debts before 2008 are included in the calculations and N_2O/N is 1%.

(differences that may be below experimental error); in some cases, the optimum is outside the 0–500 kg N ha^{-1} yr^{-1}, and zero fertilizer use would give the highest emission reduction rate per unit biofuel derived from the production system (Table 3).

Discussion

Our analysis showed that under parameter conditions that apply to relevant subsets of palm oil production on mineral soil in Indonesia, there is an 'environmental optimum' production level at which net emission savings per unit biofuel use are maximized. The net emission savings decrease strongly with increasing C debt, but the N fertilizer rate that maximizes emission savings increases with C debt. For the production systems represented in the survey (which may not represent the real average of Indonesian palm oil production across all production conditions as the sampling design included self-nomination of companies), the reported N fertilizer rate of 141 kg N ha^{-1} yr^{-1} was substantially below the 'economic optimum' rate and likely achieving only 67% of attainable yield (as defined by the empirical Y_{max} parameter), but using much less fertilizer than would be needed to achieve the maximum (444 kg N ha^{-1}). However, the economic optimum estimate will be lowered if further risks (physical production, prices) are included in the model. A safety margin of a factor 6 on p has to be inferred to explain the average fertilizer level observed. The N fertilizer level used at each of the plantations might reflect the actual economic optimum for the type of planting material and local circumstances, which may not be the same across all plantations in the data set. Figure 5 is not the result of a controlled fertilizer experiment, but a summary of current fertilizer use and yields, where assessment of the life cycle average yield required extrapolation beyond the data (Fig. 4) and as such has some

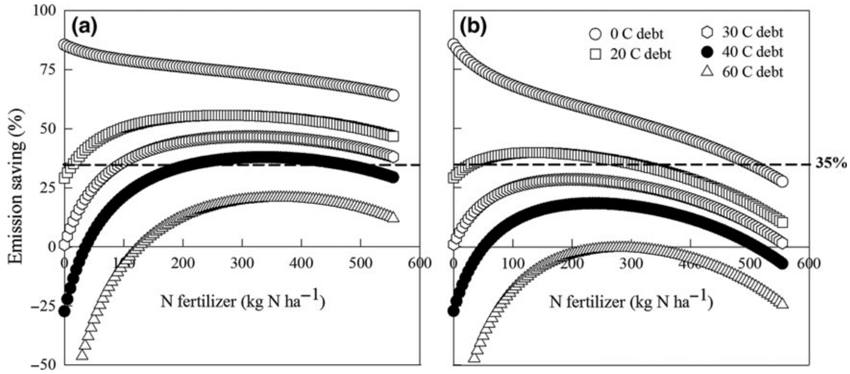

Fig. 7 Relationship between N fertilizer level and the net emission reduction if Indonesian palm oil is used as feedstock for biodiesel at default parameter conditions, for two levels of the assumed N_2O–N/N fertilizer emission ratio and five levels of carbon debt (preceding time-averaged C stock of oil palm plantations): (a) 1% N loss as N_2O, (b) 4% N loss as N_2O.

Table 3 Key characteristics of the relationship between N fertilizer and net emission reduction in Fig. 6

N_2O–N/N-fertilizer	C debt, ton C ha^{-1}	Max. emission savings (%)	Emission minimizing fertilizer, kg N ha^{-1}	Meeting 35% target*	
				Min	Max
0.01	0	85.5	0	0	>500
0.01	20	55.8	270	15	>500
0.01	30	46.7	310	95	>500
0.01	40	38.0	335	210	475
0.01	60	20.8	365	–	–
0.04	0	85.5	0	0	500
0.04	20	39.8	140	30	395
0.04	30	28.5	200	–	–
0.04	40	18.3	235	–	–
0.04	60	−0.6	280	–	–

*Acceptable fertilizer range to achieve at least 35% emission savings.

uncertainty built in. Our assumption of a quadratic response model represents the most efficient side of the spectrum, with field-scale variability likely shifting towards Mitscherlich-type response curves with higher economic optimum fertilizer rates, and lower environmentally acceptable ones (van Noordwijk & Wadman, 1992). Figure 8 shows the effects of spatial variability in the three parameters of the fertilizer–yield response. Variability in Y_{max} (e.g. through variability in plant characteristics) has a stronger depressing effect on the response curve than variation in Y_0 or f; however the effect is strongest when all three are variable. Interestingly, effects become noticeable at fertilizer rates above 200 kg N ha^{-1} and relative yield levels at 80% of Y_{max}. There may be space to increase yields from 67% to 80% of Y_{max} before negative effects on the emissions footprint emerge. The recorded fertilizer rate is in the environmental optimum range that maximizes emission savings per unit of biofuel use if a 4% N_2O–N/N fertilizer ratio is used, and below this level if a 1% N_2O emission factor applies. As default we assumed no change in CH_4 sink strength between forests and well-managed plantations, which may not be true at high N fertilizer rates (Tate, 2015).

The C debt (phase I) and N_2O emission per unit N fertilizer use (phase III) are the two dominant parameters in the calculation. The first factor had been recognized before (Agus *et al.*, 2013), the second not yet explicitly. Under the EU RED policy, conversion to oil palm before 2008 is not considered, so older plantations have zero C debt. This grandfather clause was not included in the construction of Fig. 6. Details of soil and crop management in phase III may influence results for N_2O emissions. The realistic value of N_2O–N/N fertilizer ratio may well be between the 1% estimate of IPCC (2006) and the 4% value proposed by Crutzen *et al.* (2008). Uncertainty about the true value of this parameter needs to be considered in the application of biofuel standards, but may apply across all crops. Further measurements of this ratio are a priority for research

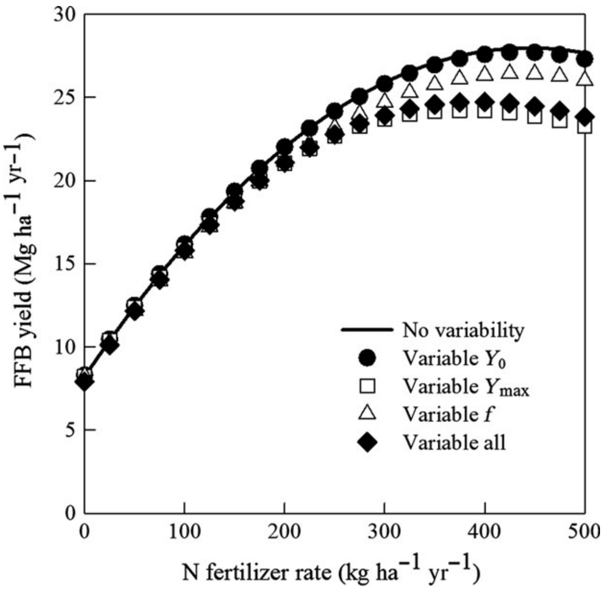

Fig. 8 Effect of spatial variability on any or all three parameters of the fertilizer response curve of Fig. 5a; average over 100 independent draws from uniform distributions in the range of 50–150% of original value.

(Reijnders & Huijbregts, 2008; Reijnders, 2011). Richards *et al.* (2016) compared a number of existing models and calculation schemes for N_2O fluxes from tropical agricultural soils and found that the substantial variation in both space and time in measured fluxes is not adequately accounted for by any current model. The IPCC emission factors are at least calibrated to global average emission data, but there is opportunity to improve on both practice and accounting method.

Phase IV of the life cycle – transport and processing – can have a strong impact on the absolute levels of emission savings (Choo *et al.*, 2011), but does not influence the environmentally optimum N fertilizer rate in phase II, as phase III is expressed per unit product. Methane capture at the mill, an established technology that is not yet widely used, can increase emission savings by about 10% across the board. Full utilization of biomass residues and by-products for static energy production, rather than focus on biofuel, can increase emission savings (Kamahara *et al.*, 2008), but may interfere with the recycling of organic residues to the plantations, affecting the maintenance of soil organic matter. Further analysis of the IPOC/ICRAF data will clarify which current management practices risk a declining C_{org} content, while overall levels can be just about maintained (Khasanah *et al.*, 2015a). Maintaining C_{org} content of forest-derived mineral soils is probably possible (Powers *et al.*, 2011).

In the 100–400 kg N ha^{-1} yr^{-1} range that includes virtually all data points, the emission savings per unit biofuel respond weakly positive or weakly negative to the N fertilizer level. Given the uncertainties around the data, there is no strong argument for modifying fertilizer price policies as a measure to reduce emissions. The N fertilizer rates currently used are slightly below what would be 'environmental' optimum in most conditions. Overall, the data indicate that intensification through increases of fertilizer rates above current practice could increase yields from the current 67% to 80% of attainable yields without negative effects on the footprint of Indonesian palm oil. Our analysis showed that there is an intermediate level of intensification of palm oil production system, achieving between 67% and 80% of its potential, that maximizes the possible net emission savings when palm oil is used as biofuel. The C debt (phase I) and processing/transport parameters (phase IV) have an overriding impact on the net emission savings attributed to palm oil use, but phase III does not influence the level of intensification that minimizes emissions. Behavioural studies on fine-tuning management decisions matter for achieving sustainability goals in the oil palm industry (Choong & McKay, 2014), as well as elsewhere. In between the Borlaug hypothesis and ecological agriculture, intermediate levels of intensification need to be fine-tuned to match the emerging public policy standards. Fertilizer price instruments cannot, in this situation, be expected to secure environmental policy outcomes beyond what land use policies and market-based accountability for footprints can achieve.

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References

Agus F, Henson IE, Sahardjo BH, Harris N, van Noordwijk M, Killeen TJ (2013) Review of emission factors for assessment of CO_2 emission from land use change to oil palm in Southeast Asia. In: *Reports from the Technical Panels of the Second Greenhouse Gas Working Group of the Roundtable for Sustainable Palm Oil (RSPO)* (eds Killeen TJ, Goon J), pp. 7–27. RSPO, Kuala Lumpur. http://www.rspo.org/file/GHGWG2/3_review_of_emission_factors_Agus_et_al. pdf.

Alkabbashi AN, Alam MdZ, Mirghani MES, Al-Fusaiel AMA (2009) Biodiesel production from crude palm oil by transesterification process. *Journal of Applied Science*, **9**, 3166–3170.

Angelsen A, Kaimowitz D (2001) *Agricultural Technologies and Tropical Deforestation*. CABI, Wallingford.

Bentrup F, Pallière C 2008. Energy efficiency and greenhouse gas emissions in European nitrogen fertilizer production and use. International Fertiliser Society – Proceeding 639.

Cassman KG, Plant RE (1992) A model to predict crop response to applied fertilizer nutrients in heterogeneous fields. *Fertilizer Research*, **31**, 151–163.

Choo Y, Muhamad H, Hashim Z, Subramaniam V, Puah C, Tan Y (2011) Determination of GHG contributions by subsystems in the oil palm supply chain using the LCA approach. *International Journal of Life Cycle Assessment*, **16**, 669–681.

Choong CG, McKay A (2014) Sustainability in the Malaysian palm oil industry. *Journal of Cleaner Production*, **85**, 258–264.

Corley RHV, Tinker PB (2016) *The Oil Palm* (5th edn). John Wiley & Sons, Chichester, UK.

Corley RHV, Hardon JJ, Tan GY (1971) Analysis of growth of the oil palm (*Elaeis guinensis* jacq.) 1. Estimation of growth parameters and application in breeding. *Euphytica*, **20**, 307–315.

Couwenberg J, Dommain R, Joosten H (2010) Greenhouse gas fluxes from tropical peatlands in South-East Asia. *Global Change Biology*, **16**, 1715–1732.

Creutzig F, Ravindranath NH, Berndes G *et al*. (2015) Bioenergy and climate change mitigation: an assessment. *GCB Bioenergy*, **7**, 916–944.

Crutzen PJ, Mosier AR, Smith KA, Winiwarter W (2008) N_2O release from agro-biofuel production negates global warming reduction by replacing fossil fuels. *Atmospheric Chemistry and Physics*, **8**, 389–395.

Davis SC, Boddey RM, Alves BJR *et al*. (2013) Management swing potential for bioenergy crops. *Global Change Biology Bioenergy*, **5**, 623–638.

Demirbas A. (2007). Importance of biodiesel as transportation fuel. *Energy Policy*, **35**, 4661–4670.

Dewi S, Ekadinata A (2013) Analysis of Land Use and Cover Trajectory (ALUCT). In: *Negotiation-Support Toolkit for Learning Landscapes* (eds van Noordwijk M, Lusiana B, Leimona B, Dewi S, Wulandari D), pp. 157–162. World Agroforestry Centre (ICRAF) Southeast Asia Regional Program, Bogor, Indonesia.

European Communities Commission (2008) Proposal for a Directive of the European Parliament and of the Council on the promotion of the use of energy from renewable sources.

von Geibler J (2013) Market-based governance for sustainability in value chains: conditions for successful standard setting in the palm oil sector. *Journal of Cleaner Production*, **56**, 39–53.

Germer J, Sauerborn J (2008) Estimation of the impact of oil palm plantation establishment on greenhouse gas balance. *Environment, Development and Sustainability*, **10**, 697–716.

Grau R, Kuemmerle T, Macchi L (2013) Beyond 'land sparing versus land sharing': environmental heterogeneity, globalization and the balance between agricultural production and nature conservation. *Current Opinion in Environmental Sustainability*, **5**, 477–483.

Green RE, Cornell SJ, Scharlemann JPW, Balmford A (2005) Farming and the fate of wild nature. *Science*, **307**, 550–555.

Hairiah K, Dewi S, Agus F, Velarde SJ, Ekadinata A, Rahayu S, van Noordwijk M (2011) *Measuring Stocks Across Land Use Systems: A Manual*. World Agroforestry Centre—ICRAF, SEA Regional Office, Bogor, Indonesia.

Heege HJ (2015) *Precision in Crop Farming*. Springer, Dordrecht.

Hoefnagels R, Smeets E, Faaij A (2010) Greenhouse gas footprints of different biofuel production systems. *Renewable and Sustainable Energy Reviews*, **14**, 1661–1694.

IPCC (2006) *Intergovernmental Panel on Climate Change: 2006 IPCC Guidelines for National Greenhouse Gas Inventories*. IGES, Hayama, Japan.

IPCC (2007) Intergovernmental panel on climate change: technical summary. In: *Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change* (eds Solomon S, Qin D, Manning M, Chen Z, Marquis M, Averyt KB, Tignor M, Miller HL). Cambridge University Press, Cambridge, UK and New York, NY, USA.

Jackson LE, Pulleman MM, Brussaard L *et al*. (2012) Social-ecological and regional adaptation of agrobiodiversity management across a global set of research regions. *Global Environmental Change*, **22**, 623–639.

Kamahara H, Widiyanto A, Tachibana R, Atsuta Y, Goto N, Daimon H, Fujie K (2008) Greenhouse Gas Balance on Life Cycle of Biodiesel: A Case of Palm biodiesel Production in Indonesia. Paper presented at the 8th International Conference on EcoBalance, Tokyo. 10–12 December 2008.

Kamahara H, Hasanudin U, Widiyanto A, Tachibana R, Atsuta Y, Goto N, Daimon H, Fujie K (2010) Improvement potential for net energy balance of biodiesel derived from palm oil: A case study from Indonesian practice. Biomass and Bioenergy **34**, 1818–1824.

Khasanah N, van Noordwijk M, Ekadinata A *et al*. (2012) The carbon footprint of Indonesian palm oil production. Technical Brief No 25: palm oil series Bogor, Indonesia. World Agroforestry Centre – ICRAF, SEA Regional Office. 10 p.

Khasanah N, van Noordwijk M, Ningsih H, Rahayu S (2015a) Carbon neutral? No change in mineral soil carbon stock under oil palm plantations derived from forest or non-forest in Indonesia. *Agriculture, Ecosystems and Environment*, **211**, 195–206.

Khasanah N, van Noordwijk M, Ningsih H (2015b) Aboveground carbon stocks in oil palm plantations and the threshold for carbon-neutral vegetation conversion on mineral soils. *Cogent Environmental Science*, **1**, 1119964.

Lambin EF, Meyfroidt P, Rueda X *et al*. (2014) Effectiveness and synergies of policy instruments for land use governance in tropical regions. *Global Environmental Change*, **28**, 129–140.

Lee DR, Barrett CB (2001) *Tradeoffs or Synergies? Agricultural Intensification, Economic Development and the Environment*. CAB-International, Wallingford.

Maswar M (2011) Kajian Cadangan Karbon Pada Lahan Gambut Tropika Yang Didrainase Untuk Tanaman Tahunan. Sekolah Pascasarjana Institut Pertanian Bogor.

Minang PA, van Noordwijk M (2013) Design challenges for achieving reduced emissions from deforestation and forest degradation through conservation: leveraging multiple paradigms at the tropical forest margins. *Land Use Policy*, **31**, 61–70.

Minang PA, Duguma LA, Alemagi D, van Noordwijk M (2015) Scale considerations in landscape approaches. In: *Climate-Smart Landscapes: Multifunctionality in Practice* (eds Minang PA, van Noordwijk M, Freeman OE, Mbow C, de Leeuw J, Catacutan D), pp. 121–133. World Agroforestry Centre (ICRAF), Nairobi, Kenya.

van Noordwijk M (1999) Nutrient cycling in ecosystems versus nutrient budgets of agricultural systems. In: *Nutrient Disequilibria in Agro-Ecosystems: Concepts and Case Studies* (eds Smaling EMA, Oenema O, Fresco LO), pp. 1–26. CAB International, Wallingford.

van Noordwijk M, Brussaard L (2014) Minimizing the ecological footprint of food: closing yield and efficiency gaps simultaneously? *Current Opinions on Environmental Sustainability*, **8**, 62–70.

van Noordwijk M, Cadisch G (2002) Access and excess problems in plant nutrition. *Plant and Soil*, **247**, 25–39.

van Noordwijk M, de Willigen P (1986) Quantitative root ecology as element of soil fertility theory. *Netherlands Journal of Agricultural Science*, **34**, 273–281.

van Noordwijk M, de Willigen P (1987) Agricultural concepts of roots: from morphogenetic to functional equilibrium. *Netherlands Journal of Agricultural Science*, **35**, 487–496.

van Noordwijk M, Scholten JHM (1994) Effects of fertilizer price on feasibility of efficiency improvement: case study for an urea injector for lowland rice. *Fertilizer Research*, **39**, 1–9.

van Noordwijk M, Wadman W (1992) Effects of spatial variability of nitrogen supply on environmentally acceptable nitrogen fertilizer application rates to arable crops. *Netherlands Journal of Agricultural Science*, **40**, 51–72.

Noordwijk M, Tomich TP, Winahyu R, Murdiyarso D, Partoharjono S, Fagi AM (eds) (1995) Alternatives to Slash-and-Burn in Indonesia, Summary Report of Phase 1. ASB-Indonesia Report Number 4, Bogor, Indonesia, 154.

van Noordwijk M, Khasanah N, Dewi S (2013) Biofuel emission reduction estimator scheme (BERES): landuse history, production systems and technical emission factors. In: *Negotiation-Support Toolkit for Learning Landscapes* (eds van Noordwijk M, Lusiana B, Leimona B, Dewi S, Wulandari D), pp. 157–162. World Agroforestry Centre (ICRAF) Southeast Asia Regional Program, Bogor, Indonesia.

van Noordwijk M, Minang PA, Freeman OE, Mbow A, de Leeuw J (2015) The future of landscape approaches: interacting theories of place and change. In: *Climate-Smart Landscapes: Multifunctionality in Practice* (eds Minang PA, van Noordwijk M, Freeman OE, Mbow C, de Leeuw J, Catacutan D), pp. 375–386. World Agroforestry Centre (ICRAF), Nairobi, Kenya.

Powers J, Corre M, Twine TE, Veldkamp E (2011) Geographic bias of field observations of soil carbon stocks with tropical land-use changes precludes spatial extrapolation. *Proceedings of the National Academy of Sciences (USA)*, **108**, 6318–6322.

Reijnders L, Huijbregts MAJ (2008) Palm oil and the emission of carbon-based greenhouse gases. *Journal of Cleaner Production*, **16**, 477–482.

Richards M, Metzel R, Chirinda N *et al*. (2016) Limits of agricultural greenhouse gas calculators to predict soil N_2O and CH_4 fluxes in tropical agriculture. *Scientific Reports*, **6**, 26279.

Rudel TK, Schneider L, Uriarte M *et al.* (2009) Agricultural intensification and changes in cultivated areas, 1970–2005. *Proceedings of the National Academy of Sciences USA*, **106**, 20675–20680.

Sanchez PA (1994) Alternatives to slash and burn: a pragmatic approach for mitigating tropical deforestation. In: *Agricultural Technology: Policy Issues for the International Community Wallingford* (ed. Anderson JR), pp. 451–479. CAB International, Wallingford, UK.

Sheil D, Casson A, Meijaard E *et al.* (2009) *The Impacts and Opportunities of Oil Palm in Southeast Asia*. Center for International Forestry Research (CIFOR), Bogor, Indonesia.

Tate KR (2015) Soil methane oxidation and land-use change – from process to mitigation. *Soil Biology and Biochemistry*, **80**, 260–272.

Tomich TP, Van Noordwijk M, Vosti S, Whitcover J (1998) Agricultural development with rainforest conservation: methods for seeking best bet alternatives to slash-and-burn, with applications to Brazil and Indonesia. *Agricultural Economics*, **19**, 159–174.

van der Voet E, Lifset RJ, Lin Luo L (2010) Life-cycle assessment of biofuels, convergence and divergence. *Biofuels*, **1**, 435–449.

Wicke B, Dornburg V, Junginger M, Faaij A (2008) Different palm oil production systems for energy purposes and their greenhouse gas implications. *Biomass and Bioenergy*, **32**, 1322–1337.

de Wit CT (1992) Resource use efficiency in agriculture. *Agricultural Systems*, **40**, 125–151.

Woittiez. *pers comm*., 2016.

Zoebl D (1996) Controversies around resource use efficiency in agriculture: shadow or substance? Theories of C. T. de Wit (1924–1993). *Agricultural Systems*, **50**, 415–424.

Supporting Information

Additional Supporting Information may be found online in the supporting information tab for this article:

Table S1. Parameter definitions and default value in BERES.

Appendix S1. Derivation of equation for p_{SWEET}.

Appendix S2. The Biofuel Emission Reduction Estimator Scheme (BERES).