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Ecohydrological changes after tropical forest conversion to oil palm

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Abstract

Given their ability to provide food, raw material and alleviate poverty, oil palm (OP) plantations are driving significant losses of biodiversity-rich tropical forests, fuelling a heated debate on ecosystem degradation and conservation. However, while OP-induced carbon emissions and biodiversity losses have received significant attention, OP water requirements have been marginalized and little is known on the ecohydrological changes (water and surface energy fluxes) occurring from forest clearing to plantation maturity. Numerical simulations supported by field observations from seven sites in Southeast Asia (five OP plantations and two tropical forests) are used here to illustrate the temporal evolution of OP actual evapotranspiration (ET), infiltration/runoff, gross primary productivity (GPP) and surface temperature as well as their changes relative to tropical forests. Model results from large-scale commercial plantations show that young OP plantations decrease ecosystem ET, causing hotter and drier climatic conditions, but mature plantations (age > 8−9 yr) have higher GPP and transpire more water (up to +7.7%) than the forests they have replaced. This is the result of physiological constraints on water use efficiency and the extremely high yield of OP (six to ten times higher than other oil crops). Hence, the land use efficiency of mature OP, i.e. the high productivity per unit of land area, comes at the expense of water consumption in a trade of water for carbon that may jeopardize local water resources. Sequential replanting and herbaceous ground cover can reduce the severity of such ecohydrological changes and support local water/climate regulation.

1. Introduction

Oil palm (*Elaeis guineensis*) plantations expansion has boomed over the last decades (global planted area increased from 6–16 Mha between 1990 and 2010 (Pirker *et al* 2016)), mostly in Southeast Asia (Koh *et al* 2011, Dislich *et al* 2017) at the expense of biodiversity-rich tropical forests (Koh *et al* 2011, Pirker *et al* 2016, Vijay *et al* 2016) and other land-covers such as pastures or pre-existing plantations (Gaveau *et al* 2016, Austin *et al* 2017, Furumo and Aide 2017). Oil palm (OP) is the most profitable and land-efficient

oil crop in the world (Wahid *et al* 2005, Dislich *et al* 2017, Yan 2017) thanks to low management costs and high fruit productivity per hectare, i.e. yield (t ha⁻¹) six to ten times higher than rapeseed and soy (Shay 1993, Yan 2017). Moreover, palm oil finds extensive use in both food and non-food industries for its qualities and comparatively cheap price (Dislich *et al* 2017, Pirker *et al* 2016). Hence, OP expansion is projected to increase accross the entire tropics where an additional 19 Mha of suitable land are potentially available for OP cultivation (Pirker *et al* 2016). However, such an ability to provide food and raw material

(Dislich *et al* 2017), while contributing to rural development and poverty alleviation (Sayer *et al* 2012), has driven deforestation and generated negative environmetal impacts which earned OP the label of 'world's most hated crop' (Yan 2017).

Forest conversion to OP is responsible for significant biodiversity losses and carbon emissions (e.g. Koh and Wilcove 2008, Koh *et al* 2011, Carlson *et al* 2013, Dislich *et al* 2017, Vijay *et al* 2016). Due to high fruit productivity OP plantations have been documented to uptake more carbon than tropical forests (Kotowska *et al* 2015) but carbon storage in OP biomass is insufficient to balance the carbon losses caused by forest clearing (Kotowska *et al* 2015, Dislich *et al* 2017) and, over a 30 year period, OP establishment might result in carbon emissions from 702 t CO_2 ha^{-1} to 3452 t CO_2 ha^{-1} depending on the soil type (Fargione *et al* 2008, Dislich *et al* 2017). The different structure of OP plantations, having lower and less dense canopies compared to native forests, is also known to alter local climate, by increasing air/soil temperature and modifying air humidity (Hardwick *et al* 2015, Drescher *et al* 2016, Meijide *et al* 2018), and hydrology, by increasing the risk of flooding, soil erosion and nutrient leaching (Dislich *et al* 2017). However, there is a knowledge gap on the magnitude of such ecohydrological changes and their variations accross plantation ages (Dislich *et al* 2017).
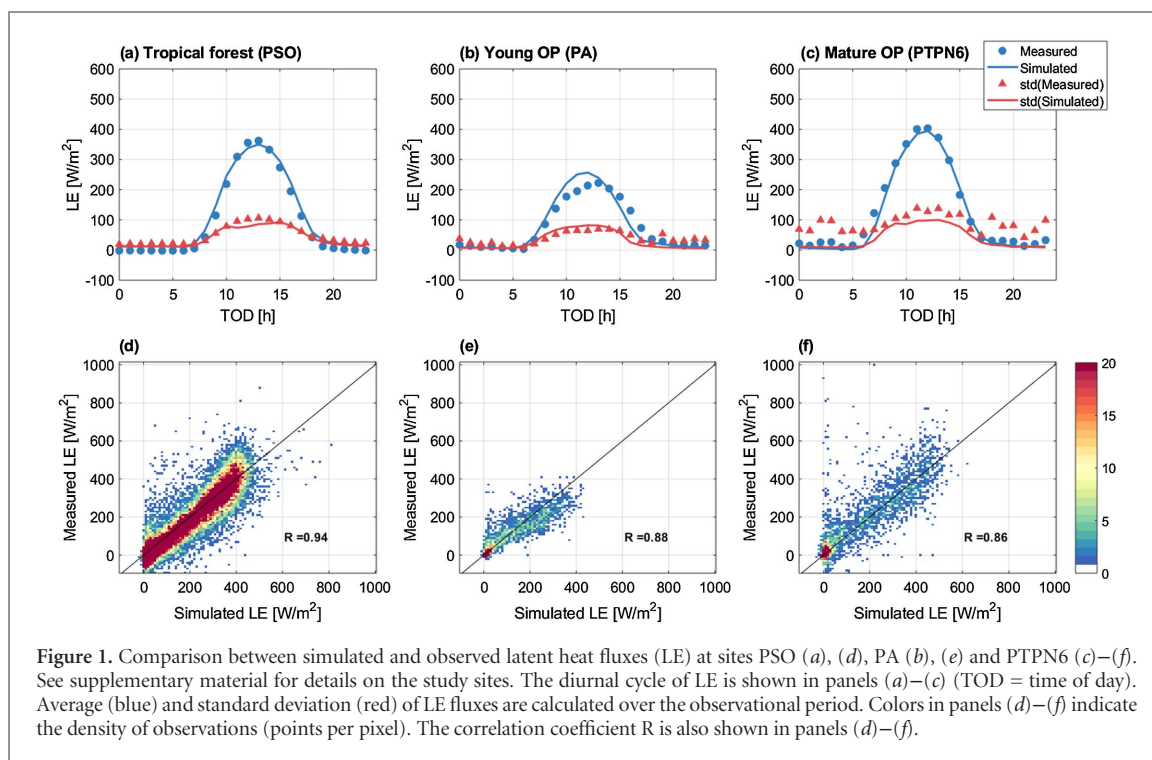
A limited literature suggest that the conversion of forests to young plantations decreases evapotranspiration (ET) and infiltration rates (Merten *et al* 2016, Dislich *et al* 2017). Decreased infiltration reduces water storage (Dislich *et al* 2017) and increases surface runoff (Comte *et al* 2012) potentially jeopardizing the access to usable water and increasing the risk of flooding. There is also anecdotal evidence that OP plantations are 'water greedy' (Merten *et al* 2016) with farmers and villagers from OP-dominated landscapes reporting water scarcity issues (e.g. decreasing water levels in wells during the dry season) as well as changes in stream flow levels and water quality (Larsen *et al* 2014, Merten *et al* 2016), which, in part, are corroborated by field observations (Carlson *et al* 2014, Merten *et al* 2016, Dislich *et al* 2017). However, contrasting results on the water requirements of OP exist in the literature and it is unclear whether young and mature plantations have similar or different ET rates when compared to native forests (Dislich *et al* 2017). In a review by Comte *et al* (2012), ET fluxes of 1000−1300 mm yr^{-1} for mature OP and 1000–1800 mm yr^{-1} for lowland forests have been reported. However, Carr (2011) indicates a palm water use of 1277−2007 mm yr^{-1} (i.e. 3.5−5.5 mm d^{-1}) with comparable transpiration rates (i.e. 2−5.5 mm d^{-1}) while Röll et al (2015) reported ET values of 1022 and 1715 mm yr^{-1} (i.e. 2.8–4.7 mm d^{-1}) with traspiration rates of 0.2 and 2.5 mm d^{-1} for a 2 and 12 year old stand, respectively.

Given the dynamic nature of OP plantations, which are cleared and replanted every two to three decades (Dislich *et al* 2017, Drescher *et al* 2016), an improved understanding of age-dependent changes is needed. Existing studies (e.g. Luskin and Potts 2011, Hardwick *et al* 2015, Röll *et al* 2015, Merten *et al* 2016, Hardanto *et al* 2017, Sabajo *et al* 2017) focused on limited observations from sites with different ages where topography, edaphic conditions, and microclimate heterogeneites potentially act as confounding effects, and the dynamic behaviour of a forest-OP plantation chronosequence is still poorly understood (Dislich *et al* 2017). Modeling efforts to simulate OP behaviour and development also exist but they are mostly focused on agronomic variables (e.g. carbon allocation, yield, fertilization) and neglect carbon/water relations and surface energy fluxes (van Kraalingen *et al* 1989, Combres *et al* 2013, Huth *et al* 2014, Hoffmann *et al* 2014, Fan *et al* 2015, Okoro *et al* 2017, Pardon *et al* 2017). Only recently, Meijide *et al* (2017) employed a land surface model adapted to OP (CLM-Palm (Fan *et al* 2015)) to simulate water/energy fluxes at two OP plantations but changes with comparison to native forests were disregarded.

Here we provide evidence of the ecohydrological impacts of forest conversion to OP by addressing the knowledge gap through model simulations constrained with field observations. With respect to OP monocultures (typical of large-scale plantations, i.e. 3000–20000 ha (Dislich *et al* 2017)), we specifically show (i) how much ecosystem ET and GPP change when a representative tropical forest is replaced by an OP plantation; (ii) we quantify how these changes are modified by plantation age; (iii) and we demonstrate the role of OP understory vegetation. This is obtained by assessing the temporal changes in water, carbon and energy fluxes relative to tropical forests, which is the typical ecosystem replaced by OP (Koh and Wilcove 2008, Dislich *et al* 2017).

2. Methods

To evaluate the ecohydrological changes induced by forest conversion to OP, model simulations by means of the ecohydrological model *Tethys* & *Chloris* (T&C) (Fatichi *et al* 2012) were combined with field data. T&C resolves surface mass and energy budgets at the land surface at the hourly-scale accounting for soil moisture dynamics and biophysical/biochemical vegetation attributes coupled with modules to simulate plant phenology, carbon allocation, and tissue turnover (Fatichi *et al* 2012, Fatichi and Ivanov 2014, Fatichi *et al* 2016, Pappas *et al* 2016, Fatichi and Pappas 2017, Mastrotheodoros *et al* 2017). The existing T&C model version has been expanded to describe OP growth by including a time-variable crown area and OP allometric relations (see supplementary material available

Figure 1. Comparison between simulated and observed latent heat fluxes (LE) at sites PSO (*a*), (*d*), PA (*b*), (*e*) and PTPN6 (*c*)−(*f*). See supplementary material for details on the study sites. The diurnal cycle of LE is shown in panels (*a*)−(*c*) (TOD = time of day). Average (blue) and standard deviation (red) of LE fluxes are calculated over the observational period. Colors in panels (*d*)−(*f*) indicate the density of observations (points per pixel). The correlation coefficient R is also shown in panels (*d*)−(*f*).

at stacks.iop.org/ERL/13/064035/mmedia, for details). OP model simulations have been calibrated and validated using short-term (∼1 yr) eddy covariance data from two sites in Indonesia (2 and 12 yr old plantations) (Röll *et al* 2015, Meijide *et al* 2017) and long-term (7−13 yr) biomass/productivity measurements from three sites in Papua New Guinea (Huth *et al* 2014) (see supplementary material).
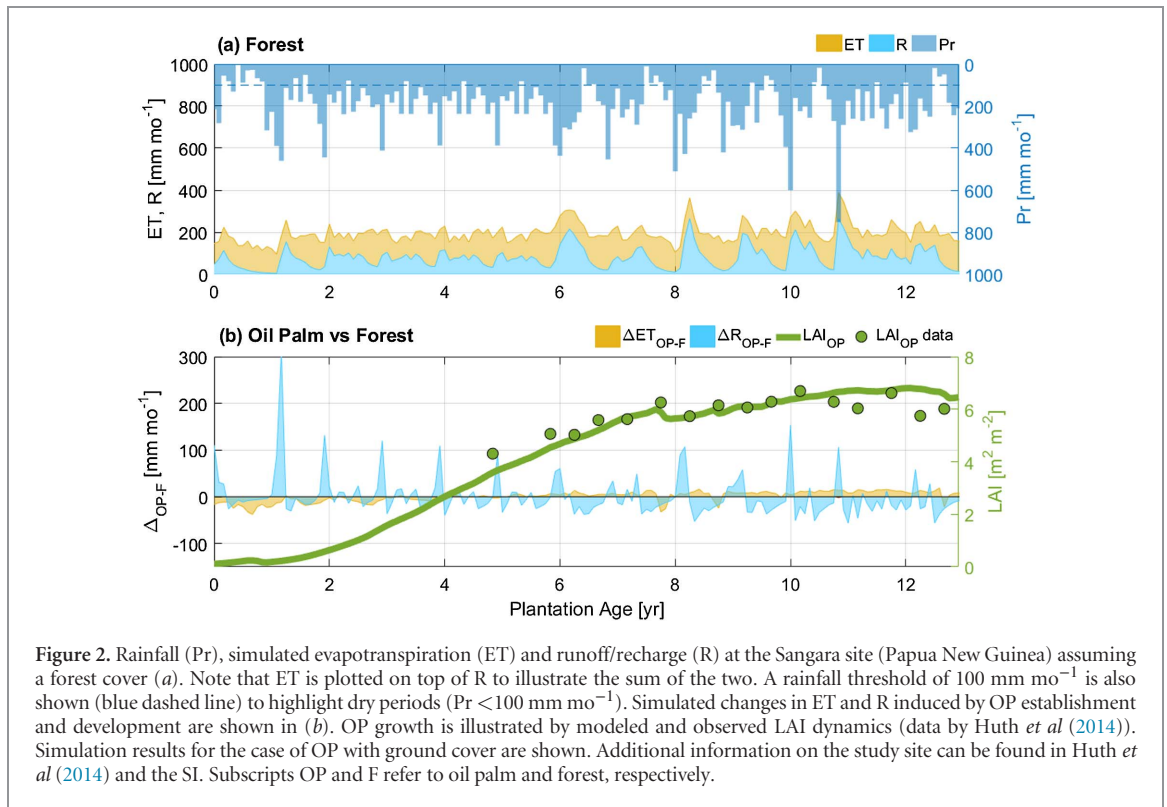
The changes in surface mass and energy fluxes following OP establishment are then evaluated using tropical forests as a reference (Fowler *et al* 2011, Dislich *et al* 2017). Hence, T&C has been also parameterized for two tropical forest ecosystems in Indonesia and Malaysia (see supplementary material). An accurate representation of carbon/water fluxes in the tropics is particularly challenging given the lack of reliable short- and long-term observations (Clark *et al* 2017) and the fact that most of existing terrestrial biosphere models systematically fail to reproduce forest seasonal dynamics (Restrepo-Coupe *et al* 2017). However, based on recent findings on the role of leaf age in regulating photosynthetic seasonality (Wu *et al* 2016), T&C has been modified to account for a mechanistic light-controlled phenology and provides an improved representation of the observed temporal dynamics of tropical forests (Manoli *et al* 2018).

Model simulations were therefore run for each of the five OP sites considering (i) the existing OP cover with two possible ground covers, i.e. with or without understory, and (ii) a land cover representing a tropical forest, for a total of 15 model runs. Bare soil or herbaceous/leguminous/crop plants are generally maintained as ground covers below OP and both scenarios were therefore modeled here. In the vegetated

understory scenario a tropical C4 grass (i.e. *Brachiaria brizantha*) was used. Although different species (e.g. *Mucuna bracteata* (Rutherford *et al* 2011)) can be employed as herbaceous ground cover, our choice is motivated by the availability of field observations needed to costrain model simulations (see supplementary material for details). The five analyzed sites have different soil and climate characteristics thus providing a general description of OP landscapes in the broader Southeast Asia region, where the highest density of OP plantations is observed (see figure S1 in the supplementary material). It has to be noted that peatland conditions and nutrient limitations were not considered in this study. However, the presence of peat mostly influences carbon stock changes and subsidence (Dislich *et al* 2017), the detailed modeling of which is beyond the scope of this study, while fertilizers are generally employed in large-scale plantations to ensure nutrient availability (Fowler *et al* 2011), so that considering OP in nutrient equilibrium is a reasonable assumption. Also, possible changes in soil hydraulic properties resulting from OP establishment and management (Dislich *et al* 2017) were neglected due to a lack of quantitative evidence (see discussion in section 4). Additional information on model equations, site characterisitcs and simulations setup are provided in the supplementary material.
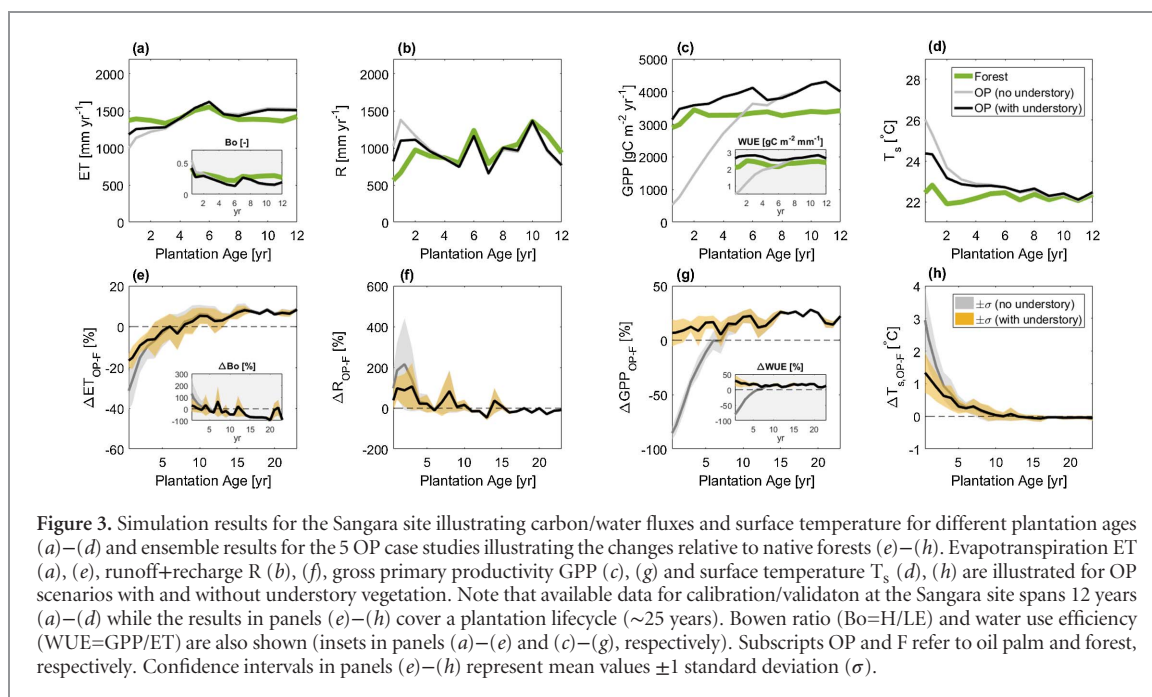
3. Results

Model simulations successfully reproduce the observed daily and seasonal dynamics of water/energy fluxes of tropical forests, young, and mature OP (figure 1

Figure 2. Rainfall (Pr), simulated evapotranspiration (ET) and runoff/recharge (R) at the Sangara site (Papua New Guinea) assuming a forest cover (*a*). Note that ET is plotted on top of R to illustrate the sum of the two. A rainfall threshold of 100 mm mo^{-1} is also shown (blue dashed line) to highlight dry periods (Pr <100 mm mo^{-1}). Simulated changes in ET and R induced by OP establishment and development are shown in (*b*). OP growth is illustrated by modeled and observed LAI dynamics (data by Huth *et al* (2014)). Simulation results for the case of OP with ground cover are shown. Additional information on the study site can be found in Huth *et al* (2014) and the SI. Subscripts OP and F refer to oil palm and forest, respectively.

and supplementary material) as well as the long-term evolution of leaf area index (LAI), yield, and above ground biomass measured at the OP sites (figure 2 and supplementary material). The simulated hydrological regime of a tropical forest and the changes induced by OP establishment are illustrated in figures 2 and 3. After forest logging, OP establishment decreases ET due to the open canopy made of young and small palms (figure 2(*b*)). The reduction in ET increases the amount of rainfall that either infiltrates into the soil (recharge) or flows on the surface as excess water (runoff). While the ET reduction is rather evenly distributed across months, changes in runoff and recharge are concentrated in a few wet months and have larger magnitude. On average, simulated ET fluxes are between 1000–1600 mm yr^{-1} for young OP (age <5 yr) and 1200−1800 mm yr^{-1} for mature plantations (age >8 yr) with recharge+runoff (R) varying between 300 and 2800 mm yr^{-1} depending on the local rainfall regime (see supplementary material). Note that infiltration/runoff partitioning depends on soil hydraulic characteristics and catchment geomorphology (e.g. Ivanov *et al* 2004, Fatichi *et al* 2014). Therefore, given the focus on plot-scale processes and the lack of knowledge on how soil properties change after forest conversion, recharge and runoff are here combined and illustrated together. However, monthly dynamics in figure 2 reveal that changes in recharge/runoff peak after very wet months, suggesting that both infiltration and saturation excess mechanisms may act to increase runoff, thus supporting the increased risk of flooding reported in the literature (Bradshaw *et al* 2007, Dislich *et al* 2017).

The results in figure 3 show that the impacts of OP on local climate and hydrology depend on plantation age and become less pronounced as plantations grow (Comte *et al* 2012, Sabajo *et al* 2017). Compared to forests, young plantations show low ET (−40% without understory, −20% when ground cover is considered), and increased recharge/runoff (up to 400% without understory, consistently with the changes in water yield reported in the literature Dislich *et al* (2017), while mature plantations increase ET up to 10% (figure 3(*e*)) thus reducing infiltration/runoff. This transition occurs at a plantation age of 8−9 yr, i.e. when plantations become mature and LAI reaches a plateau at values of 5−7 m^2 m^{-2}, and is consistent across sites/climates (figure 3). As a consequence of such age-related changes, the overall difference in the long-term water balance over the plantation lifecycle is small with cumulative changes in ET and R approaching zero at a plantation age of ∼20 yr (see supplementary material). Simulation results also confirm that OP plantations are susceptible to water stress (Carr 2011, Dislich *et al* 2017) and a mean annual precipitation (MAP) of at least 2000−2500 mm spread evenly during the year (i.e. monthly precipitation >100 mm) is required to support OP productivity (Carr 2011) as illustrated by the decrease in LAI for lower MAP values (figure 2(*b*)) and results in the supplementary material. ET fluxes by mature OP reach maximum values (up to 1800−2000 mm yr^{-1}) at MAP ∼2500 mm yr^{-1} and decrease linearly for drier conditions (−15% at a MAP of 1500 mm yr^{-1}, see supplementary material). Forest ET is mostly lower than for mature OP but the water fluxes are sustained even during dry periods suggesting

Figure 3. Simulation results for the Sangara site illustrating carbon/water fluxes and surface temperature for different plantation ages $(a)-(d)$ and ensemble results for the 5 OP case studies illustrating the changes relative to native forests $(e)-(h)$. Evapotranspiration ET (a), (e), runoff+recharge R (b), (f), gross primary productivity GPP (c), (g) and surface temperature T_s (d), (h) are illustrated for OP scenarios with and without understory vegetation. Note that available data for calibration/validaton at the Sangara site spans 12 years $(a)-(d)$ while the results in panels $(e)-(h)$ cover a plantation lifecycle ($\sim$25 years). Bowen ratio (Bo=H/LE) and water use efficiency (WUE=GPP/ET) are also shown (insets in panels $(a)-(e)$ and $(c)-(g)$, respectively). Subscripts OP and F refer to oil palm and forest, respectively. Confidence intervals in panels $(e)-(h)$ represent mean values ± 1 standard deviation (σ).

a higher tolerance to drought due to water access by deep roots (results shown in figures S10 and S12 in the supplementary material).
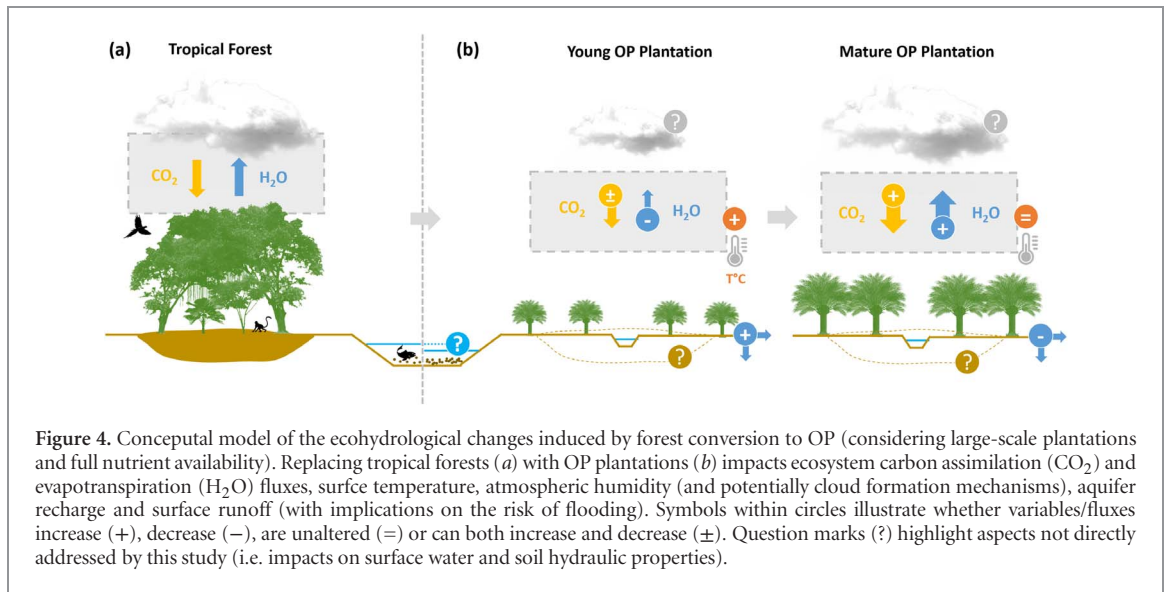
The fact that mature OPs transpire more water than native forests is directly linked to the high productivity of palms and the high fertilization rates at commercial plantations. At maturity, the GPP of OP ranges between 3000 and 4400 $gC\,m^{-2}\,yr^{-1}$, i.e. up to 25% higher than forest (figure 3(c), (g)) and the fresh fruit bunch yield is between 20 and 40 $t\,ha^{-1}$ (see supplementary material). However, the water use efficiency (WUE) is constrained between 2.2 and 3.2 $gC\,m^2\,mm^{-1}$, i.e. only 10%−20% increase with respect to forest (figure 3(g)), and the carbon-water trade-off of photosynthesis results in higher water losses. These changes in ET also affect the surface energy budget. Young plantations have lower latent heat (LE) and higher sensible heat (H) fluxes (see supplementary material), resulting in a higher surface temperature (+2−3 °C, figures 3(d),(h)) and a different partitioning of energy at the land surface (i.e. higher Bowen ratio, $Bo = H/LE$). Mature plantations show reversed conditions when compared to forests, with lower Bo and small changes in temperature (as also found by Sabajo *et al* (2017)).

Simulated changes during the initial stages of OP development are mitigated when ground cover is maintained (figure 3). Understory vegetation reduces the impact of young plantations on ET, recharge/runoff and temperature and maintains an overall higher GPP. C4 grasses have higher WUE and GPP than forests (e.g. Ludlow 1985, Wolf *et al* 2011) if water and nutrients are available. However, as OPs grow, competition for light and water significantly suppresses the vegetated ground cover which becomes negligible when OPs reach a mature state at LAI $>5\,m^2\,m^{-2}$.

4. Discussion

4.1. Impacts on water resources and microclimate

The ecohydrological impacts of forest conversion to OP are summarized in figure 4. Young plantations transpire less water than native forests, thus increasing surface temperature (+2 °C−3 °C), soil moisture and the risk of flooding. On the contrary, mature plantations increase ET fluxes (up to 10%), reproducing a microclimate similar to forests (in terms of T_s as potential changes in air humidity are not modeled here) but depleting more water and thus reducing recharge/runoff. These results are consistent with field observations, e.g. Hardwick *et al* (2015) reported temperature changes up to +6.5 °C, Fowler *et al* (2011) and Meijide *et al* (2017) reported lower Bo in mature compared to young plantations, and Sabajo *et al* (2017) observed much higher surface temperatures in young plantations compared to forests (+6.0 ± 1.9 °C) but similar values in mature plantations (+0.8 ± 1.2 °C). The magnitude of the simulated impacts is also in agreement with variations in surface temperature observed at the global scale after changes in land use and land management (Luyssaert *et al* 2014). Our simulations confirm that: (i) impacts on surface climate are enhanced in young plantations (Luskin and Potts 2011) and (ii) this type of ecosystem functioning recovers as plantations mature (Dislich *et al* 2017). However, our results also corroborate local knowledge and perception of water scarcity reported in Indonesia and attributed to OPs, considered 'water-greedy' by farmers and villagers in OP-dominated landscapes (Merten *et al* 2016). The fact that cumulative changes in ET and R decrease with plantation age and become negligible over a plantation lifecycle should not be misleading since water run off in a

Figure 4. Conceptual model of the ecohydrological changes induced by forest conversion to OP (considering large-scale plantations and full nutrient availability). Replacing tropical forests (*a*) with OP plantations (*b*) impacts ecosystem carbon assimilation (CO_2) and evapotranspiration (H_2O) fluxes, surfce temperature, atmospheric humidity (and potentially cloud formation mechanisms), aquifer recharge and surface runoff (with implications on the risk of flooding). Symbols within circles illustrate whether variables/fluxes increase (+), decrease (−), are unaltered (=) or can both increase and decrease (±). Question marks (?) highlight aspects not directly addressed by this study (i.e. impacts on surface water and soil hydraulic properties).

young plantation will not be available for ET in a old plantation few years later. Hence, short-term disturbances of the hydrological balance increase the risk of flooding (at early plantation stages) and water scarcity (after maturation) creating two distinctive hydrological regimes that differ considerably from the hydrology of forest ecosystems and the impacts of which do not sum up to zero. Note that similar issues of high water use and potential impacts on local water resources have been reported for other agroforestry systems, such as Eucalyptus plantations in tropical and subtropical regions (Liu *et al* 2017, Maier *et al* 2017). In general, model simulations support previous findings that OPs are considerably vulnerable to water stress (Carr 2011). This is due to a large use of water through ET to sustain fruit productivity and to relatively shallower rooting systems when compared to forests (Carr 2011, Ivanov *et al* 2012). Irrigation is a valuable option to sustain yield under dry periods (Carr 2011), but the use of water from aquifers for irrigation may exacerbate the depletion of water resources (Famiglietti 2014, Dislich *et al* 2017). Hence, given the projected increase of atmospheric droughts (Novick *et al* 2016) and dry season length in the tropics (Neelin *et al* 2006), the very large OP water requirements (ET ∼ 5−6 mm d^{-1}) may constrain the ongoing expansion of OP across the tropics and reduce productivity (Pirker *et al* 2016).

The simulated changes in the Bowen ratio also suggest that convective clouds (and thus rainfall) can be suppressed by young plantation but sustained by mature ones (Bonetti *et al* 2015, Manoli *et al* 2016). Hence, the deforestation-induced suppression of rainfall observed in Borneo (Kumagai *et al* 2013), where rainfall recycling from terrestrial ET plays a dominant role, could be mitigated by mature OP plantations. Given the relatively limited global extent of OP cultivated area (16 Mha in 2010 (Pirker *et al* 2016)), impacts on global climate dynamics are unlikely but further research should clarify possible implications of

OP expansion on land-atmosphere interactions at the regional scale. In addition, the results here demonstrate that the current best practice of planting ground covers is a considerable mitigation option to reduce local warming, confirming the role of land management in regulating surface temperature (Davin *et al* 2014, Luyssaert *et al* 2014). Adopting sequential replanting rather than clear-cutting the entire plantation can also reduce negative hydrological and microclimatic effects by maintaining a range of palm ages (Luskin and Potts 2011). Our simulations corroborate the benefit of this practice also providing a quantification of its impact. However, the same practice can spread diseases and reduce agronomic efficiency prompting the need for a more extensive cost-benefit analysis (Luskin and Potts 2011).

4.2. Challenges and perspectives

In this study we have focused on OP temporal dynamics (i.e. growth, water consumption), assuming spatial homogeneity within large-scale plantations. Future studies need to focus on the role of spatial fragmentation (i.e. smallholder plantations) and heterogeneous OP age distributions accross the landscape, as the role of spatial heterogeneities is largely unexplored and can likely contribute to mitigate impacts on the hydrological cycle. Nutrient limitation was also neglected in this study. This is a reasonable assumption for large-scale plantations but not in the case of smallholders where fertilization rates are often suboptimal and lower yields are obtained (Huth *et al* 2014, Röll *et al* 2015). This can explain the fact that ET rates slightly lower than forests were recently observed in mature smallholder-run plantations (Sabajo *et al* 2017). Hence, the impact of different managements practices (e.g. fertilization rates, large-scale plantations vs smallholders) on OP water consumption, surface energy fluxes, and changes in local microclimate need to be further investigated.

Furthermore, at the catchment scale, OP establishment can decrease infiltration (Merten *et al* 2016) and aquifer recharge (Bruijnzeel 2004, Dislich *et al* 2017), concurrently increasing the risk of floods (Bradshaw *et al* 2007) and erosion (Guillaume *et al* 2015). The deep recharge/fast runoff partition largely depends on catchment geomorphology and soil hydraulic characteristics (Dislich *et al* 2017), which could not be fully accounted for in the plot scale analysis here. Similarly, it has been shown that different management strategies and soil compaction due to mechanical clearing can alter soil physical characteristics and reduce infiltration (Bruijnzeel 2004, Matthews *et al* 2010, Moradi *et al* 2015, Dislich *et al* 2017), but the relevant changes in hydraulic properties are still poorly quantified, prompting the need for further field investigations. With regard to the effect of the ground cover, we have considered a tropical C4 grass for the sake of convenience, but different crops and herbaceous species are often employed. According to a survey by Rutherford *et al* (2011), planting of a cover crop is a common management strategy in Southeast Asia. When the ground cover is maintained, a mixtures of legume (e.g. *Mucuna bracteata*) and grass species is often used (Rutherford *et al* 2011, Huth *et al* 2014) as nitrogen-fixing leguminous plants can improve soil fertility (Dislich *et al* 2017). However, targeted data are needed to assess the differential capabilities of understory species to regulate local microclimate, increase infiltration, and reduce soil erosion (Dislich *et al* 2017).

4.3. Broader impact

Over the last decade oil palm expansion has partly shifted from forested to non-forested lands providing an opportunity for increasing sustainability (Gaveau *et al* 2016, Furumo and Aide 2017). In Indonesia, the proportion of OP plantations directly replacing forests declined from 54%–18% between the 1995−2000 and 2010−2015 (Austin *et al* 2017) and 79% of OP expansion in Latin America occured on previously cleared land (Furumo and Aide 2017).Such a transition from non-forested land to OP seems promising for a sustainable development of OP, but a cautious assessment and management of water resources is still needed. Hence, while more research is required to understand the ecohydrological impacts of different land cover transitions to OP (e.g. pastures, croplands, savannas), our results highlight the importance of including water resources in the debate on OP sustainability and foreshadow the key role of water availability in constraining OP plantations expansion across the tropics.

More generally, given the 'hybrid' socio-natural dimension of water (Linton and Budds 2014, Merten *et al* 2016), the impact of OP expansion goes beyond the biophysical dimension illustrated here and is directly linked to population dynamics, social inequality issues, technological development and market regulation (Yan 2017, De Pinto *et al* 2017). Despite the local awareness, OP water requirements have been marginalized or neglected in the OP controversy, which mostly focused on carbon emission mitigation and biodiversity conservation (Sayer *et al* 2012, Larsen *et al* 2014). However, large scale land acquisitions can potentially drive groundwater depletion and streamflow reduction (Merten *et al* 2016, Dalin *et al* 2017), which we could quantify as remarkable. Hence, there is a need to integrate ecohydrological studies with participatory approaches (e.g. Castelletti and Soncini-Sessa 2006, Étienne 2013) to model the coupled 'hydrosocial cycle' (Linton and Budds 2014) and support sustainable development of OP landscapes by quantifying the trade-offs among different management alternatives.

5. Conclusion

In conclusion, this study provides a comprehensive quantitative understanding of the ecohydrological changes occurring in OP landscapes from forest logging to OP establishment, growth and maturity. Specifically, our results show the key role of plantation age in regulating ecosystem functioning. Young plantations cause surface warming (+2 °C−3 °C) and increase the risk of flooding while mature plantations restore local climate but consume more water (+3%−8%) than the forests they have replaced. Thus, the 'allegedly high water use' of OP (Merten *et al* 2016) is here confirmed by process-based ecohydrological simulations, demonstrating that the high land use efficiency of OP comes at the expense of water consumption in a trade of 'water for carbon' (Jackson *et al* 2005) that, in the long term, may jeopardize local water resources. The practices of retaining old palms during crop rotation and cultivating ground cover crops can mitigate some of the impacts and are recommended to support water/climate regulation in OP-dominated landscapes.

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