Geoderma 338 (2019) 410–421

Contents lists available at ScienceDirect

Geoderma

journal homepage: www.elsevier.com/locate/geoderma

Rates and spatial variability of peat subsidence in *Acacia* plantation and forest landscapes in Sumatra, Indonesia

Chris D. Evans[a,*], Jennifer M. Williamson[a], Febrio Kacaribu[b], Denny Irawan[b], Yogi Suardiwerianto[c], Muhammad Fikky Hidayat[c], Ari Laurén[d], Susan E. Page[e]

[a] Centre for Ecology and Hydrology, Deiniol Road, Bangor LL57 2UW, United Kingdom
[b] Faculty of Economics and Business, Universitas Indonesia, Jakarta, Indonesia
[c] Asia Pacific Resources International Ltd., Pangkalan Kerinci, Kabupaten Pelalawan, Riau 28300, Indonesia
[d] Faculty of Science and Forestry, University of Eastern Finland, Joensuu Campus, PO Box 111, Yliopistokatu 7, 80101 Joensuu, Finland
[e] Department of Geography, University of Leicester, Leicester LE1 7RH, United Kingdom

ARTICLE INFO

Handling editor: David Laird

Keywords:
Organic soils
Tropical peat
Drainage
Subsidence
Carbon
Greenhouse gases

ABSTRACT

Many peatlands in Europe and North America have been developed for agriculture for over a century, whilst in Southeast Asia development has largely occurred since 1990. Cultivation of drained peatlands now supports the livelihoods of large numbers of people, and the ongoing economic development of countries such as Indonesia and Malaysia. However, peat subsidence linked to plantation drainage represents both an environmental and a socio-economic challenge, associated with elevated CO_2 emissions, impacts on adjacent forest habitat, and long-term changes in plantation drainability. Whilst the fundamental challenges presented by peat subsidence are broadly recognised, the long-term rates and the potential for mitigation or avoidance of subsidence remain uncertain. We analysed over 2000 site-years of subsidence measurements from 312 sites in Sumatra, Indonesia, collected under *Acacia* pulpwood plantation and adjacent native forest, representing the largest peat subsidence dataset published to date. Subsidence averaged 4.3 cm yr^{-1} in the *Acacia* plantations, and extended at least 300 m into adjacent forest. Mean water table depth (*WTD*) was the best predictor of subsidence rate in both plantation and forest areas. We did not find conclusive evidence that subsidence was intrinsically faster under *Acacia* plantation than under native forest or (by comparison with previous studies) oil palm plantations for the same level of drainage. Our results suggest that raising average *WTDs* to the Indonesian Government's 40 cm target could – if practically and economically viable means of achieving this can be developed – reduce current plantation subsidence rates by 25–30%. Whilst some degree of peat subsidence under any form of plantation management may be unavoidable, these reductions would – if achieved at scale – both increase the economic lifetime of the plantations, and simultaneously deliver reductions in CO_2 emissions of national and global significance.

1. Introduction

Peatlands occupy approximately 4 million km^2 (around 3% of the global land area) and hold an estimated 630 Pg of carbon (C) (Page et al., 2011; Dargie et al., 2017). This equates to around 30% of all C held in active (soil and biomass) terrestrial pools (Ciais et al., 2013). An estimated 11% of global peat area, and 17% of total peat carbon, is located in the humid tropics, with the largest deposits found in the Southeast Asian islands of Borneo, Sumatra and New Guinea; the Congo Basin; and Western Amazonia (Page et al., 2011; Dargie et al., 2017). Tropical peatlands are naturally forested, but in the last 30 years approximately 60% of the peat swamps of Sumatra and Borneo have been deforested (Miettinen et al., 2017). Some of this area has become degraded and unproductive, whilst around 50% of the total peat area is now under some form of plantation management, either for agriculture (mainly oil palm, *Elaeis guineensis*) or wood fibre production (mainly *Acacia crassicarpa*). This transition from natural forest to plantation has contributed to regional economic development, and now supports the livelihoods of large numbers of people. One consequence of this change, however, is peatland subsidence. This occurs due to a combination of compaction (compression and shrinkage of aerated peat), consolidation (compression of peat below the water table due to loss of buoyancy of overlaying peat), and oxidation (aerobic decomposition of organic matter to CO_2) (Stephens et al., 1984; Andriesse, 1988; Hooijer et al.,

* Corresponding author.
E-mail address: cev@ceh.ac.uk (C.D. Evans).

https://doi.org/10.1016/j.geoderma.2018.12.028
Received 17 August 2018; Received in revised form 13 December 2018; Accepted 15 December 2018
Available online 26 December 2018

2012; Nagano et al., 2013). Since the majority of Indonesian and Malaysian peatlands occur in low-lying regions, subsidence could eventually lead to increased coastal and river flooding risk, challenges for plantation water management, and possibly a reduction in the land area available for plantation agriculture (Aswandi et al., 2017).

At present, knowledge of the rates of and controls on tropical peat subsidence is limited. Whilst a number of studies have been undertaken in tropical peatlands, most have relied on relatively short (< 3 year) measurement periods (Hooijer et al., 2012; Wakhid et al., 2017; Ishikura et al., 2018); small numbers of measurement sites (Wösten et al., 1997; Othman et al., 2011; Nagano et al., 2013; Wakhid et al., 2017; Ishikura et al., 2018); subsidence rates inferred from surveys of peat elevation and bulk density (Kool et al., 2006); or models (Aswandi et al., 2017). The most spatially extensive field study undertaken to date was carried out by Hooijer et al. (2012), partly updated by Couwenberg and Hooijer (2013). This included 125 sites under *Acacia* with two years of data (including a subset of 14 sites with 8 years of data); 61 sites under oil palm with three years of data; and 51 sites in drainage-affected forest adjacent to the *Acacia* plantations.

Subsidence is typically rapid during the first few years following land clearance and drainage, due to initial collapse of newly dewatered peat and oxidation of the labile organic matter store, as well as the impacts of heavy machinery, and ground disturbance due to land clearance and stump removal. Following this period of 'primary' subsidence, 'secondary' subsidence tends to proceed at a slower rate. Studies of high-latitude peatlands have indicated that secondary subsidence rates also decrease over time, due to a combination of: i) unmaintained drainage; ii) changing water retention characteristics, leading to decreased aeration; iii) rising mineral content of the residual peat as organic matter is lost; and iv) accompanying increases in bulk density, as a relatively high proportion of subsidence results from compaction rather than oxidation (e.g. Hutchinson, 1980; Stephens et al., 1984; Andriesse, 1988; Pronger et al., 2014). In contrast, tropical deep peats typically have very low mineral contents, and high temperatures lead to a high ratio of oxidation to compaction (Hooijer et al., 2012). Although effective water management (such as maintaining water levels during the dry season) may slow subsidence rates in tropical plantation systems, ultimately it will be necessary to lower water levels at approximately the same rate as any subsidence that does occur, in order to maintain a constant drainage depth. This combination of factors led Couwenberg and Hooijer (2013) to argue that subsidence in tropical peat plantations will remain near-constant over time, at constant water table depths, until the water table intersects underlying mineral soils. At this point, subsidence might be expected to slow or stop as the remaining peat becomes progressively harder to drain, leaving only a shallow residual peat layer.

Increasingly, regional governments and plantation companies are recognising the challenges presented by drainage-based peatland cultivation, and seeking to develop policies and management practices to reduce CO_2 emissions from peat oxidation and fires, along with associated peat subsidence (Wijedasa et al., 2016). The El Niño fires of 2015 led the Indonesian government to establish a Peat Restoration Agency (Badan Restorasi Gambut, BRG) with a remit to reduce fire incidence and restore 2 million ha of degraded peatland by 2020. A series of recent Indonesian Government regulations (most recently SK.22/PPKL/PKG/PKL.0/7/2017) also require that water tables be maintained within 40 cm of the peat surface at the centre of each plantation block for half of the year, and within 100 cm of the surface at all times. This represents a significant change to existing operational procedures, which generally involve drainage of the peat to a target depth of around 70 cm.

In this study, we analyse a uniquely large and long-term set of subsidence measurements made in *Acacia* plantations and adjacent native forest areas from one of the largest peatland regions in Sumatra, Indonesia. The aims of the study were: 1) to estimate the long-term rate of subsidence in peatlands under *Acacia* plantation, and in adjacent areas of conservation-managed forest; and 2) to evaluate the factors influencing spatial variability in subsidence rate, within both plantation and adjacent forest landscapes. A subsequent paper will examine the extent to which subsidence rates have varied over time since drainage, and in response to climatic fluctuations.

2. Materials and methods

2.1. Site description

Data were obtained from peatlands in Riau province, eastern Sumatra, Indonesia. Sumatra's peatlands largely formed within the last 8000 years, and occupy ~70,000 km^2 (Dommain et al., 2014). They occur mainly in coastal areas, usually at elevations of < 10 m above sea level. Peat thickness exceeds 5 m over much of the region. Mean annual rainfall at Pekanbaru Airport, Riau is ~2900 mm with two wet seasons (March–April and October–December) and two dry seasons (January–March and May–August), and monthly mean air temperatures ranging from 29 to 32 °C (Badan Meteorologi, Klimatologi dan Geofisika, 1994–2017 data).

Since the 1990s, around two-thirds of the peat swamp forests of Riau have been cleared and drained for agriculture (primarily oil palm) and pulpwood production, but the province still retains around 7% of its original peat swamp forest in an intact condition with a further 17% classed as degraded forest (Miettinen et al., 2016). All of the sites included in the study are located within concession areas held by Asia Pacific Resources International Limited (APRIL), one of Indonesia's largest pulp and paper companies (Fig. 1). APRIL and its long-term supply partners hold concessions for around one million ha in Riau, of which 584,000 ha are on peatland. This area is divided between 262,000 ha of *Acacia crassicarpa* plantation, grown for fibre production, and 322,000 ha of native forest (both within and adjacent to plantation concessions) which is under some form of conservation management. The *Acacia* plantations were established from 1992 onwards, and are managed on a five-year rotation from planting to harvesting. Water levels in APRIL's plantations are actively managed via an extensive network of topographically-defined water management zones, controlled by outlet sluices, and supported by large-scale and continuous rainfall and water level monitoring. Water management zones comprise navigable canals, typically of 12 m width and 3 m depth, also used for transportation. Branch canals of 5–8 m width run perpendicular to these canals at a spacing of 500–800 m to form plantation compartments, which contain 1 m deep field drains at a spacing of 75 m. Following the original license conditions, the majority of the plantations are located on the outer part of peat domes, with the larger intact natural forests (notably the 300,000 ha Kampar Forest, 240,000 ha Kerumutan Forest, and smaller forests on the islands of Pulau Padang and Pulau Rupat) occupying the inner domes. Plantation concession margins are largely managed as 'buffer zones' of 300 m to 2 km width, most of which comprise native forest, although some areas have been planted with a native plantation species (*Melaleuca* sp.). These areas are not intentionally drained, but may be affected by adjacent plantation drainage, with associated dieback of some (typically larger) trees close to canals. Smaller/marginal areas of forest within or adjacent to the plantation concessions typically comprise regenerating degraded secondary forest.

2.2. Field measurements

Peat subsidence was measured with hollow, perforated 5 cm diameter hollow PVC poles, inserted vertically into the peat and anchored into underlying mineral subsoil. Ground elevation relative to a local datum was recorded quarterly by measuring distance from the top of a pole (the datum point) to the ground surface. Subsidence was recorded as negative where the ground surface was falling, and positive (i.e. growth rather than subsidence) where the surface was rising. Boards

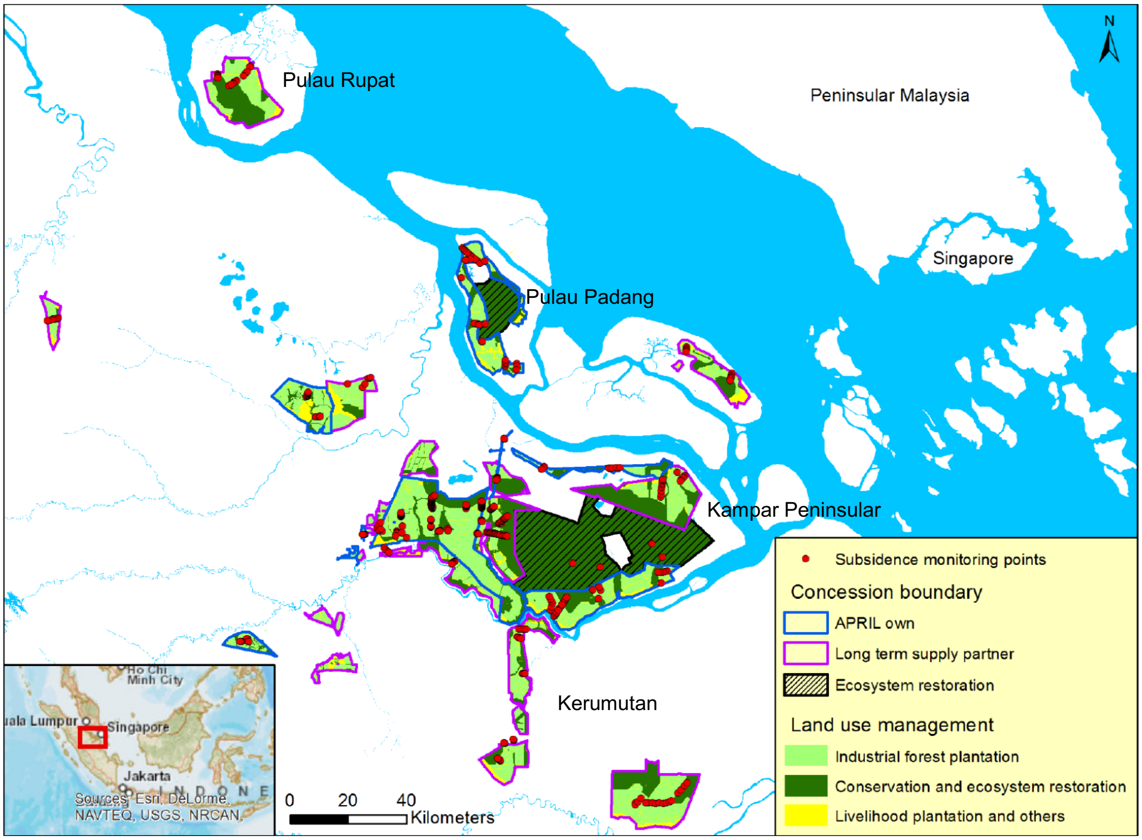

Fig. 1. APRIL concession areas and subsidence monitoring locations, Riau, Sumatra.

were used to minimise ground disturbance during installation, and care taken during subsequent visits to avoid artificially compacting the peat surface by remaining outside a 0.5 m radius of the pole. During harvesting and replanting periods, efforts were made to avoid disturbing the area surrounding each pole. Consequently some mechanical subsidence (i.e. compaction rather than oxidation) due to periodic plantation management activities may not be captured by the method used.

Water table depth (*WTD*) was recorded on each visit as depth from the pole top to the water surface within the pole, minus pole length above the ground surface; *WTD* was recorded as positive if below the ground surface, and negative if above. Throughout this study, we used mean *WTD* derived from all manual measurements made at each site as a measure of average drainage depth.

In total, data from 447 subsidence poles were analysed. The first poles were installed in 2007, with further poles added periodically up until the present day. In this assessment we analysed data collected up until December 2016. Placement of subsidence monitoring points has not followed a formal or randomised survey design, with poles installed in clusters, typically along transects perpendicular to canals, and often extending from plantations into the adjacent forest. Earlier poles were largely established as part of a research programme (described by Hooijer et al., 2012) whilst subsequent poles were installed by APRIL's staff. As shown in Fig. 1, measurement locations span most of APRIL's concession areas, with the exception of interior areas of larger conservation/restoration concessions (although poles have now been installed in these areas, records are not yet sufficiently long to support analysis). At the start of the study, the majority of poles were located within *Acacia* plantation, but more recently the proportion of poles in native forest areas, as well as the number of sites remote from plantations, has been increased. Small numbers of poles in other land cover classes, including *Melaleuca* buffer strips and community land, were insufficient for analysis and therefore removed from the analysis. Early data from a subset of the sites reported on in this study formed part of a previous analysis by Hooijer et al. (2012), but data from the majority of sites (and all data since 2010) have not previously been reported.

Additional site data were collated where available, including initial peat depth, initial peat elevation, date of pole installation and current harvest rotation. For plantation sites, we used GIS data layers provided by APRIL to determine distance from each plantation monitoring pole to the nearest canal. For forest sites, we quantified distance from the measurement point to the nearest forest/plantation boundary. This measure was used on the basis that (compared to distance to the nearest plantation canal) it can be more easily and reliably derived from aerial or satellite imagery, making the resulting relationships transferrable to other areas without detailed drainage mapping data. In around two thirds of cases, the plantation/forest boundary took the form of a canal, and in all cases there was a canal close to the plantation boundary.

2.3. Data analysis and screening

We applied two separate approaches to the estimation of mean subsidence rates in the plantation and forest data. Firstly, time series of measurements from each subsidence pole were used to derive a mean subsidence rate for this pole, and these site means were then analysed across the dataset. Secondly, each individual quarterly subsidence measurement was treated separately in the analysis. Both methods required prior screening of the data to remove anomalous values, and each method had advantages and disadvantages, discussed later. By applying two methods, we aimed to provide some degree of cross-validation of the subsidence estimates obtained.

2.3.1. Site-based analysis of data (Method 1)

To derive mean subsidence rates for individual subsidence poles, we collated data from all poles with at least three years of data and at least 12 individual measurements (322 poles in total; see Supplementary Table 1 for summary information). All data were recorded as peat surface depth below the top of the pole, which provided a fixed datum point for each location. Initial depth of the peat surface below the datum point was subtracted from each measurement to give a measure of total subsidence, relative to a value of zero at the time of pole installation, for each quarterly measurement date. Data were manually screened prior to analysis to remove any measurements where damage to the pole, or re-installation of the pole at a differing depth, had occurred (damage was typically due to plantation harvesting, site preparation or planting operations). Anomalously large positive or negative changes in depth (> 30 cm) between successive quarterly measurements, which were approximately reversed in subsequent measurements, were assumed to result from recorder error and also removed. A linear regression was then fitted to the full dataset, with the regression coefficient providing an estimate of mean subsidence for that location. Subsidence values were expressed as negative where ground surface elevation was falling, and positive if the ground surface was rising.

To avoid including time series containing large step changes (considered likely to result from unrecorded changes in pole datum) affecting our analysis, we identified sites where the correlation between depth and time was non-significant (i.e. $p > 0.05$). This gave a total of nine suspect records. Of these, two had a RMSE < 2 cm, suggesting that the lack of a correlation between depth and time was due to a genuine lack of change in ground elevation, and these were retained in the analysis. The remaining seven sites showed clear step changes and were omitted. A further three records that met the significance criterion but were located on atypical land use (bare ground) were also excluded. In total, 312 records were retained in the analysis.

To calculate mean subsidence rates ($Subs_m$, cm yr^{-1}) by land-use category (Aim 1), data were aggregated by land-use (i.e. plantation and forest) and an arithmetic mean calculated for each land-cover type. To analyse spatial controls on subsidence rates (Aim 2) we plotted mean subsidence at each site against candidate explanatory variables including mean annual WTD (WTD_m, cm), initial peat depth and elevation, distance from nearest canal and nearest forest/plantation boundary, separating data by land-use type and fitting linear or non-linear regressions as appropriate to the data. Data were analysed using a mixed model, with transect (*tr*) as a random effect and WTD_m, land-use (*lu*), distance from forest poles to the nearest forest/plantation boundary ($d_{f\text{-}p}$, m), distance from plantation poles to the nearest canal ($d_{p\text{-}c}$, m), initial peat depth (*pd*, cm), and rotation (r) as fixed effects, using the Non-Linear Mixed Effects (*nlme*) procedure in R (Pinheiro et al., 2016; R Core Team, 2015). To avoid any potential bias due to seasonal variations in rainfall in the calculation of WTD_m at sites with missing data, mean WTD for each quarter was calculated separately and WTD_m calculated as the mean of these four values (gap-filling of missing data was not possible as values for an entire transect were normally missing at any given time point, and correlations between neighbouring transects were not sufficiently strong to support their use for gap-filling). Plantations were assigned a rotation number of 1 when subsidence poles were installed on site (which generally followed first planting of the site) and this increased each time the stand was harvested. This variable was included to determine whether any of the variation in subsidence could be explained by the time a site had been under plantation management.

2.3.2. Individual measurement-based analysis of data (Method 2)

In Method 2, we treated each individual (quarterly) subsidence measurement as a separate data point, and calculated means per land-cover category as the mean of all individual measurements in that sample set. The approach differs from Method 1 in that: i) monitoring poles with a longer run of data effectively gain greater weight in the calculation of mean subsidence rates compared to shorter records, because there are more individual measurements for those sites; ii) a more systematic approach could be taken to handle influential and outlying observations; and iii) data from recently-installed subsidence poles could be included. We analysed subsidence data from a total of 447 individual measurement poles, spanning the period 2010–2016. After accounting for gaps in observations, this gave an initial total of 8666 individual subsidence observations. This dataset was then objectively screened for outliers using the Blocked Adaptive Computationally Efficient Outlier Nominators (*bacon*) procedure in Stata (Billor et al., 2000; Weber, 2010). This procedure conducts an iterative process of multivariate outlier detection based on the Mahalanobis distance (De Maesschalk et al., 2000), which measures distance of each observation from the central point of the whole dataset. In order to maintain consistency with planned future analysis, we implemented data screening in a multivariate setting comprising five variables: subsidence rate, water table depth, rainfall, distance to nearest canal (plantation sites) and distance to plantation boundary (forest sites). The *bacon* procedure objectively defines outliers based on statistical distributions, iteratively defining a threshold Mahalanobis distance between 'nonoutlier' and 'outlier' values, and thereby avoids subjective decision-making with regard to the removal of apparently anomalous values. For this study, the threshold Mahalanobis distance was iteratively set to 7.492, giving a final screened dataset of 6328 'non-outlier' observations.

2.3.3. Comparison with previous subsidence measurements

To place data from the study within a wider regional (Southeast Asian) and global context, we collated published subsidence estimates from over 40 previous studies. These were limited to those that: i) were carried out on deep peat (> 2 m); ii) involved direct measurement of peat surface elevation change using either subsidence poles or repeated peat depth or elevation surveys; and iii) spanned a minimum period of two years. We excluded data collected within three years of initial drainage, when rapid primary subsidence was likely to be occurring. Although the number of measurement points varied widely between studies, in general there was a trade-off between the extent, duration and quality of data; for example studies based on single subsidence poles provided highly reliable, long time series of subsidence measurements but no replication, whereas extensive studies provided high replication but typically relied on less accurate elevation measurements and only two time points. We therefore aggregated data from each study in order to provide a single subsidence estimate, unless the study specifically compared different land-cover types, or presented data from two completely independent datasets. Measurements made under different experimental water levels at a single location were averaged. For a few very long subsidence records that showed marked non-linearity in subsidence in response to historical changes in drainage, we calculated subsidence from the most recent period during which water table depths were approximately stable. In all cases we recorded the number of measurement points and duration over which measurements were made, and where reported we also recorded mean WTD, following the same approach used to derive mean subsidence rates.

3. Results

3.1. Mean subsidence rates

Using the site-based approach (Method 1) to calculate mean subsidence rates within the dataset, we estimated $Subs_m$ to be -4.3 cm yr^{-1} (± 2.0 cm yr^{-1} standard deviation) in *Acacia* plantations (n = 220). Subsidence rates showed a slight negative skew, with a small number of very high subsidence rates but very few subsidence rates < 1 cm yr^{-1} (10th percentile -6.65 cm yr^{-1}, median -4.04 cm yr^{-1}, 90th percentile -2.07 cm yr^{-1}). Mean subsidence rates were lower in the adjacent native forest (-3.4 ± 1.8 cm yr^{-1}, n = 92). The distribution and skew of data were similar to the plantations (10th percentile -6.31 cm yr^{-1}, median -2.81 cm yr^{-1}, 90th percentile -1.56 cm yr^{-1}).

Using the individual measurement-based approach (Method 2) we obtained $Subs_m$ values of -4.4 (± 3.6) cm yr^{-1} and -3.2 (± 3.9)

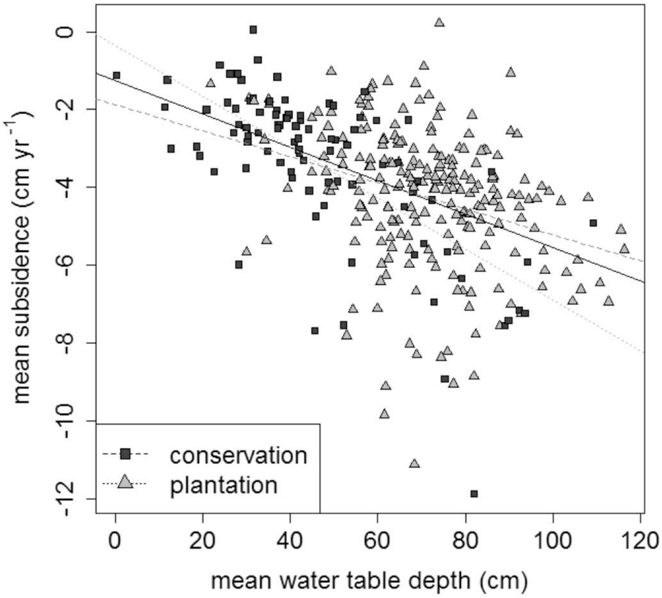

Fig. 2. Mean measured rate of subsidence and water table depth for all individual measurement sites. Solid line shows the best fit line of a linear regression fitted to all data (n = 318), short dashed line show fit to native conservation forest sites (n = 92) and long dashed line fit to *Acacia* plantation sites (n = 220).

cm yr^{-1} for plantation and adjacent forest respectively. The close agreement between subsidence estimates, despite differences in methodology and data inclusion/exclusion criteria, provides some confidence in the results obtained, notably for the plantations where the different methods gave very similar estimates of mean subsidence. The slightly larger difference in forest subsidence estimates may reflect the inclusion of data from recently initiated subsidence monitoring sites in more remote conservation forest areas in the analysis for Method 2, whereas many of these records were still too short (< 3 years) for inclusion in the analysis for Method 1.

3.2. Spatial controls on mean subsidence rates

3.2.1. Water table depth

As expected, there were differences in mean WTD (WTD_m) between plantation and adjacent native forest sites, but both showed evidence of drainage impacts (*WTD* means and standard deviations 70 ± 17 cm and 47 ± 22 cm respectively). Almost all plantation sites had a WTD_m of 40 cm or greater, whereas forest sites spanned a very wide range, from 0 to 109 cm. The relationship between $Subs_m$ and WTD_m across the dataset is shown in Fig. 2. We observed a modest but highly significant correlation between $Subs_m$ and WTD_m across the dataset as a whole, with a weaker (but still significant) relationship for the plantation data only, and a stronger relationship for the native forest sites only:

$$\text{Full dataset:} \quad Subs_m = -0.0431\,\mathrm{WTD_m} - 1.24 \qquad R^2 = 0.25 \qquad p < 0.001 \tag{1}$$

$$\text{Plantation:} \quad Subs_m = -0.0334\,\mathrm{WTD_m} - 1.88 \qquad R^2 = 0.09 \qquad p < 0.001 \tag{2}$$

$$\text{Native forest:} \quad Subs_m = -0.0654\,\mathrm{WTD_m} - 0.35 \qquad R^2 = 0.45 \qquad p < 0.001 \tag{3}$$

Additional information for these regression equations (significant and standard error of intercepts and coefficients, number of data points) are provided in Supplementary Table 2. For the full- and plantation-only datasets, the intercepts on the regressions were significantly different from zero (p < 0.001 based on linear mixed effects modelling) whereas the intercept on the forest-only regression was not. A high degree of scatter in subsidence rates was noted at intermediate mean annual WTDs (60 to 80 cm), and around 50% of the plantation sites fell within this range during the measurement period (reflecting the target WTD_m of 70 cm for plantation operations). Although the gradient of the relationship between $Subs_m$ and WTD_m for forest was steeper than those for the whole-dataset and plantation-only regressions, inspection of the individual data points did not reveal clearly divergent relationships or offsets between subsidence rates in the plantation and forest categories at equivalent WTDs (Fig. 2). Thus we did not find strong evidence that $Subs_m$ at any given WTD_m was substantially different between plantation and adjacent native forest.

3.2.2. Distance to nearest canal or forest/plantation boundary

For areas under *Acacia* plantation, $Subs_m$ was also unrelated to distance to nearest canal ($d_{p\text{-}c}$) (R^2 = 0.00, p = 0.34). Note however that we were unable to include field drains in this analysis, thus true distance to the nearest drainage feature may be less than (and not necessarily related to) distance to the nearest canal. We also found no significant relationship between $Subs_m$ under Acacia and distance to the forest/plantation boundary and rate of subsidence; i.e. plantation blocks at the centre of concession areas were not found to be subsiding faster than those at the periphery. For native forest areas (including forest buffer areas within plantation concessions) in contrast, we observed a significant non-linear relationship between subsidence rate and distance from the nearest forest/plantation boundary:

$$\text{Native forest:} \quad Subs_m = -0.471\ \ln(d_{f-p}) - 5.87 \qquad R^2 = 0.14 \qquad p < 0.001 \tag{4}$$

We also observed a correlation between mean WTD_m and distance from the nearest forest/plantation boundary for the same dataset:

$$\text{Native forest:} \quad WTD_m = 8.05\ \ln(d_{f-p}) - 89.9 \qquad R^2 = 0.32 \qquad p < 0.001 \tag{5}$$

These relationships (Fig. 3) suggest that both WTD_m *and* $Subs_m$ are strongly affected within 300 m of the nearest forest/plantation boundary, which is in most cases also the nearest canal. Neither WTD nor subsidence were found to approach zero within the range of sites available for the analysis (note that more recent data have been collected from locations remote from the plantations; see below). Combining $d_{f\text{-}p}$ and WTD_m did not increase the overall amount of variance in $Subs_m$ that could be explained.

3.2.3. Other potential explanatory variables

We did not find significant correlations between $Subs_m$ and any of the other candidate explanatory variables considered, individually or in combination. A subset of 256 subsidence poles had accompanying peat depth measurements, which ranged from 2.5 to 16.8 m, as well as initial surface elevation data. For this subset we found no relationship between subsidence and either peat depth (R^2 = 0.01, p = 0.79) or initial peat surface elevation (R^2 = 0.05, p = 0.45). Adding these variables to the significant relationships described above between $Subs_m$ and WTD_m (whole dataset and plantation-only data), or with $d_{f\text{-}p}$ at the native forest sites, actually reduced the overall significance of the regressions, because only a subset of sites had recorded values for these variables.

Analysis of the plantation data by current harvest rotation, based on mixed effects modelling, provides some suggestion of a reduction in $Subs_m$ with successive rotations: whole-period mean subsidence rates were −5.34, −4.12, −4.28 and −3.63 cm yr^{-1} for sites currently under first, second, third and fourth rotations respectively. However, variability within each group was high, and the number of replicates was limited for first and fourth rotation sites in particular (Table 1). Subsidence rates in the second and third rotation subsets, which account for the majority of data points, were similar. Overall, we did not observe significant differences in subsidence rate between rotations (p = 0.25).

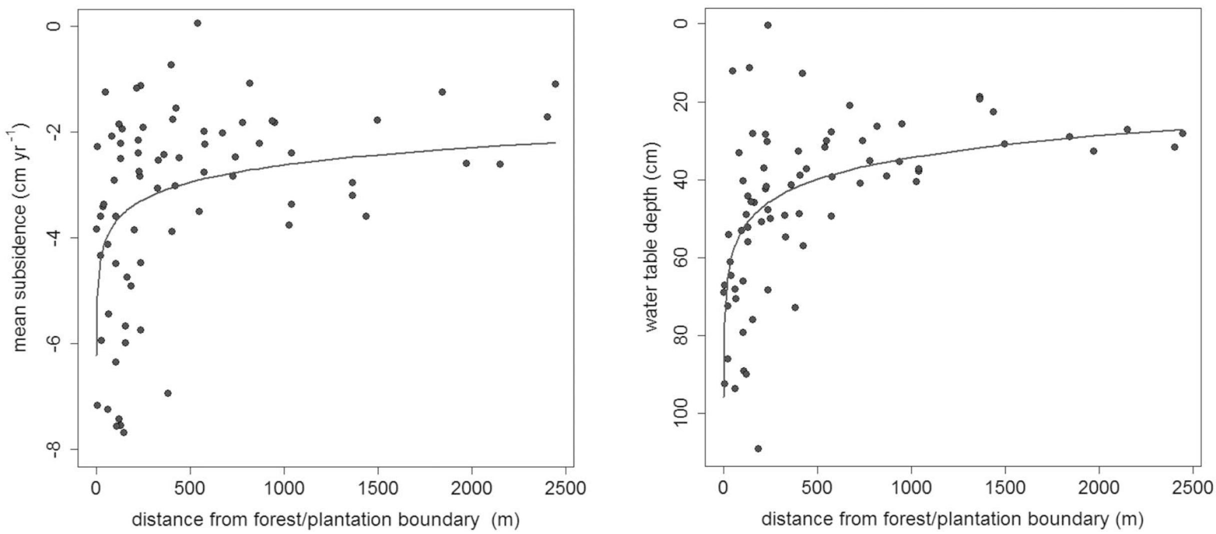

Fig. 3. Relationship between observed mean subsidence rate (a) and water table depth (b) versus distance from the forest/plantation boundary for all measurement sites located within conservation forest (n = 74 in both plots). Lines show log-linear regression equations as shown in text (Eqs. (4)–(5)).

Table 1
Mean and standard deviation of measured subsidence rates (in cm yr^{-1}) in Acacia plantations for individual measurement sites grouped by current plantation rotation number.

Rotation number	Mean subsidence	Standard deviation	Number of poles
1	−5.34	2.56	14
2	−4.12	1.58	87
3	−4.28	1.75	96
4	−3.63	2.04	18

3.2.4. Comparison with previous subsidence estimates

We identified eight studies reporting subsidence data for tropical peatlands that met our selection criteria (Table 2). These included *Acacia* and oil palm plantations, forests with varying levels of drainage, and one study of mixed smallholder agriculture. The only previous study of *Acacia* plantations, by Hooijer et al. (2012) gave a higher mean subsidence rate. This study was based on a subset of our data, and on sites having the same mean *WTD*, but was undertaken when all sites were still in their first or second rotation. Their mean subsidence rate of −5.0 cm yr^{-1} is intermediate between our first and second rotation means (Table 1) and therefore consistent with our results. For studies undertaken in oil palm plantations, the range of subsidence reported was −1.6 to −4.3 cm yr^{-1}, with a mean of −3.4 cm yr^{-1}. Mean *WTD* for these studies was 55 cm. Among forest sites, our mean subsidence rate of −3.2 to −3.4 cm yr^{-1} was higher than previous reported values, however these were all obtained from sites with shallower mean *WTD*s (from 37 cm below the ground surface to 8 cm above the surface), in some cases remote from plantations.

For high-latitude peatlands, we collated data from over 28 studies spanning a climatic range from boreal (Finland, Norway) to sub-tropical (Florida) and incorporating arable, grassland and sites drained for plantation forestry (Table 3). Although the studies varied widely in methodology, number of measurement sites and duration, subsidence rates were consistently lower than those reported for tropical peatlands: treating each study as one data point, mean subsidence rates for arable, grassland and plantation forest were −1.84, −1.27 and −0.92 cm yr^{-1} respectively.

Again treating each row in Tables 2 and 3 as a data point, we observed significant overall relationships between $Subs_m$ and WTD_m (where reported) for both tropical and high-latitude peatlands (Fig. 4), with the following linear relationships (see also Supplementary Table 2):

Table 2
Estimates of subsidence rates in tropical deep peat based on direct measurements of elevation change, and at least two years of data.

Vegetation type	Location	N sites	Duration (years)	Mean WTD (cm)	Subsidence (cm yr^{-1})	Reference
Acacia	Indonesia (Sumatra)	125	2	70	−5.0	Hooijer et al. (2012)
Acacia	**Indonesia (Sumatra)**	**220**	3–10	**70**	**−4.3**	**This study**
Oil palm	Indonesia (Sumatra)	29	3	56	−3.9	Couwenberg and Hooijer (2013)
Oil palm	Indonesia (Sumatra)	42	3	66	−3.7	Couwenberg and Hooijer (2013)
Oil palm	Malaysia (Sarawak)	18	8	45	−4.3	Othman et al. (2011)
Oil palm	Malaysia (Sarawak)	1	2	60	−1.6	Ishikura et al. (2018)
Oil palm	Malaysia (Peninsular)	16	17–21	ND	−2.7	DID and LAWOO (1996)
Oil palm	Malaysia (Peninsular)	10	4	53	−3.8	DID and LAWOO (1996)
Oil palm	Malaysia (Peninsular)	17	21	ND	−3.5	Wösten et al. (1997)
Mixed agricultural	Indonesia (Sumatra)	3	3	54	−2.6	Khasanah and van Noordwijk (2018)
Forest	Thailand	1	23	−8	−0.7	Nagano et al. (2013)
Forest	Thailand	4	23	19	−2.2	Nagano et al. (2013)
Forest	Indonesia (Sumatra)	1	3	37	−1.8	Khasanah and van Noordwijk (2018)
Forest	Indonesia (Sumatra)	51	3	33	−2.4	Hooijer et al. (2012)
Forest	**Indonesia (Sumatra)**	**92**	3–10	**47**	**−3.4**	**This study**

Bold text indicates results obtained from this study.

Table 3
Comparative subsidence estimates from high-latitude peatlands.

Land-use type	Location	N sites	Duration (years)	Mean WTD (cm)	Subsidence ($cm\ yr^{-1}$)	Reference
Arable	Canada (Ontario)	1	3	102	−3.30	Mirza and Irwin (1964)
Arable	Canada (Quebec)	1	10	ND	−2.50	Mathur et al. (1982)
Arable	Canada (Quebec)	1	38	ND	−2.07	Millette (1976)
Arable	Germany	2	12	98	−2.15	Eggelsmann and Bartels (1975)
Arable	Italy	1	4	50	−0.75	Zanello et al. (2011)
Arable	Switzerland	15	141	110	−1.26	Leifeld et al. (2011)
Arable	UK (England)	7	30	ND	−1.37	Richardson and Smith (1977)
Arable	UK (England)	117	22	ND	−1.48	Dawson et al. (2010)
Arable	UK (England)	1	53	120	−1.56	Hutchinson (1980)
Arable	USA (California)	13	8	90	−1.25	Deverel et al. (2010, 2016)
Arable	USA (Florida)		20	ND	−1.45	Shih et al. (1998)
Arable	USA (Florida)	15	88	ND	−1.82	Aich et al. (2013)
Arable	USA (Florida)	1	76	ND	−1.40	Wright and Snyder (2009)
Arable	USA (Florida)				−3.00	Stephens et al. (1984)
Arable	USA (Indiana)	3	6	75	−2.26	Jongedyk et al. (1950)
Forest	Finland	273	60	ND	−0.37	Minkinnen and Laine (1998)
Forest	Finland	4	30	ND	−0.48	Minkinnen et al. (1999)
Forest	UK (Scotland)	101	29	55	−1.91	Shotbolt et al. (1998)
Grassland	Germany	1	40	80	−0.83	Kluge et al. (2008)
Grassland	Germany	1	66	80	−0.67	Eggelsmann and Bartels (1975)
Grassland	Germany	1	35	ND	−0.50	Eggelsman (1976)
Grassland	Netherlands	8	6	64	−0.53	Schothorst (1977)
Grassland	Netherlands	1	88	15	−0.06	Schothorst (1977)
Grassland	New Zealand	66	80	ND	−2.56	Fitzgerald and McLeod (2004)
Grassland	New Zealand	10	40	ND	−3.40	Schipper and McLeod (2002)
Grassland	New Zealand	119	12	ND	−1.90	Pronger et al. (2014)
Grassland	Norway	11	28	ND	−2.00	Grønlund et al. (2008)
Grassland	Norway	5	31	ND	−1.04	Grønlund et al. (2008)
Grassland	Poland	18	38	53	−0.17	Grzywna (2017)
Grassland	UK	ND	10	ND	−0.62	Brunning (2001)
Grassland	USA (California)	34	28	ND	−2.20	Deverel and Leighton (2010)

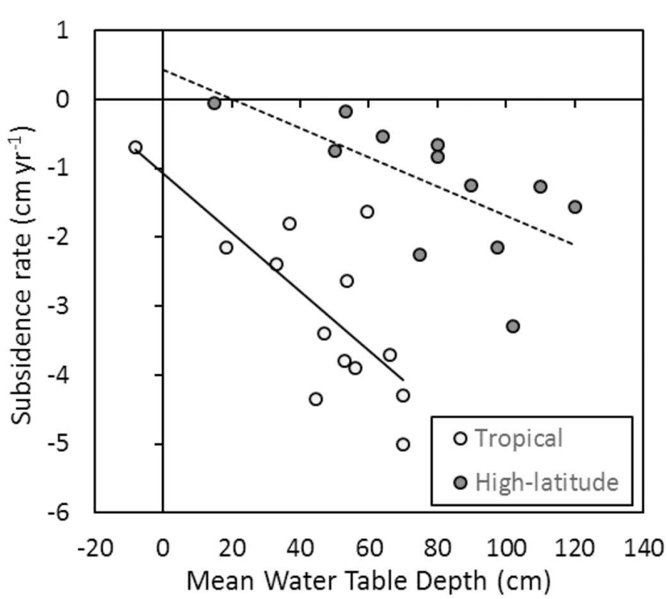

Fig. 4. Relationship between mean subsidence and mean water table depth for those studies reporting both measurements in Tables 2 (tropical peatlands) and 3 (high-latitude peatlands).

Tropics: $Subs_m = -0.0428\,WTD_m - 1.08 \quad R^2 = 0.55 \quad p = 0.003$ (6)

High-latitudes: $Subs_m = -0.0212\,WTD_m + 0.43 \quad R^2 = 0.43 \quad p = 0.02$ (7)

The equations suggest that a 1 cm change in WTD_m will increase subsidence rate by approximately twice the amount in tropical compared to high-latitude peatlands. The negative intercept on the tropical peatland data suggests that some subsidence will occur even at a WTD_m of zero, however this intercept was not significantly different from zero ($p = 0.09$). The positive intercept for high-latitude peatlands was also non-significant ($p = 0.53$).

4. Discussion

4.1. Subsidence rates in tropical peat plantations

Our analyses suggest that drained tropical peatlands under *Acacia* plantation are subsiding at an average rate of -4.2 to $-4.3\ cm\ yr^{-1}$. This is similar to, but lower than, the previous estimate of $-5\ cm\ yr^{-1}$ for *Acacia* reported by Hooijer et al. (2012), which was based on a subset of the sites considered in our study, a shorter measurement period, and data that had (on average) been collected sooner after land conversion. Compared to the larger body of published subsidence estimates for oil palm plantations, our *Acacia* subsidence estimate lies at the upper end of the reported range, and is around $0.5\ cm\ yr^{-1}$ higher than the mean reported value. However, this difference appears to be broadly consistent with the difference in mean *WTDs* (70 cm in *Acacia* versus 55 cm for those oil palm studies reporting values). Applying Eq. (1) (based on all plantation and forest data) gives a predicted $Subs_m$ of $-3.6\ cm\ yr^{-1}$ at a WTD_m of 55 cm, almost identical to the oil palm mean. It remains possible that other factors could influence peat subsidence rates under *Acacia* compared to those under oil palm or other land-cover categories, such as: shorter rotation times (5 years for *Acacia* vs 30 years for oil palm), leading to more frequent ground disturbance and exposure of bare peat to solar radiation (e.g. Jauhiainen et al., 2014); greater shading of the peat surface during the latter part of each rotation; fertiliser application rates (typically highest in oil palm plantations, and absent in forests and unmanaged scrubland); and differences in above and below-ground litter input and distribution (e.g.

Jauhiainen et al., 2012). Overall, however, we did not find clear evidence that subsidence rates under *Acacia* differ from those under oil palm, or indeed forest land, for an equivalent water table depth.

Both our own dataset and the collated literature data indicate that WTD_m exerts a strong influence over $Subs_m$. The striking similarity between Eqs. (1) and (6), based on independent analyses, provides confidence that the relationship obtained from our dataset may be generalisable to other tropical peatland areas and land-use categories. The collated high-latitude data also show a clear influence of WTD_m on $Subs_m$ (Eq. (7)). The observed divergence of tropical and high-latitude data, and higher slope coefficient in Eq. (6) compared to Eq. (7), support previous conclusions that subsidence rates are higher for any given WTD_m in tropical peatlands than in high-latitude peatlands, primarily due to higher temperatures and associated decomposition rates (e.g. Andriesse, 1988; Stephens et al., 1984). We also compared our results with an empirical model of subsidence as a function of WTD and soil temperature provided by Stephens et al. (1984), based on data from Florida (their Eq. (6)). Based on a mean soil temperature for Acacia plantations of 28.7 °C measured under 1–2 year old plantation (Chandra Deshmukh, unpublished data) and a WTD_m of 70 cm, this equation gives a predicted subsidence rate of $-5.6\ \mathrm{cm\ yr^{-1}}$, around $1.3\ \mathrm{cm\ yr^{-1}}$ faster than our observed rate.

Some caution is required when comparing the high-latitude and tropical datasets collated in our study, because the tropical data were largely obtained from recently-drained systems, whereas many of the high-latitude studies were carried out on sites subject to much longer-term drainage (in some cases over a century). Data from drained high-latitude peatlands clearly show a decrease in subsidence rate over time, from around $6\ \mathrm{cm\ yr^{-1}}$ shortly after drainage to approximately $1\ \mathrm{cm\ yr^{-1}}$ after a century (Pronger et al., 2014). Our results from recently-drained *Acacia* plantations are similar to this initial rate, however other authors (e.g. Couwenberg and Hooijer, 2013) have argued that subsidence rates in tropical peatlands will remain high over longer periods due to differences in peat type, water level management and climatic conditions. Our analysis of subsidence rates in first to fourth rotation *Acacia* showed some evidence of a reduction in subsidence rate over time, but differences were non-significant.

Despite this possible divergence in the long-term trajectory of subsidence in tropical versus high-latitude peatlands, it is clear that the majority of high-latitude data derive from sites that are at an advanced stage on the 'subsidence curve' of Pronger et al. (2014). As a result, long-term mean subsidence rates in these systems may be somewhat under-estimated. Furthermore, the *cumulative* subsidence in historically drained high-latitude peatlands typically exceeds subsidence in tropical peatlands to date. For example, based on the mean values obtained, a high-latitude peatland drained for arable 100 years ago would (even assuming linear subsidence over time) have lost 184 cm of elevation due to secondary subsidence, compared to 86 cm in a tropical *Acacia* plantation drained 20 years ago. Because many of the locations included in Table 3 have now subsided below sea-level, they require continuous pumped drainage, with high associated energy costs. It is also noteworthy that mean WTD_m for the high-latitude studies reporting data (76 cm) exceeded that of the tropical plantation studies (60 cm), with several high-latitude studies reporting $WTD_m > 1$ m (Table 3).

4.2. The influence of plantation management on adjacent native forests

Our results suggest that drainage of tropical peatlands for plantation agriculture can cause subsidence in adjacent areas of native forest. This is consistent with previous analyses of subsidence data, and with broader understanding of the landscape-scale impact of drainage in tropical peatlands (Hooijer et al., 2012; Evans et al., 2014; Baird et al., 2017; Cobb et al., 2017). Our data from forest monitoring sites show clear correspondence between distance to the nearest plantation boundary, WTD_m and $Subs_m$, with the most strongly drained forest sites (such as small forest fragments within plantation concessions) having similar WTD_m and $Subs_m$ to those in the plantations themselves (Fig. 2). The data show clear evidence of plantation drainage impacts within approximately 300 m of the plantation boundary (Fig. 3). This figure contrasts with similar measurements from a drained forest plantation established in blanket bog in Scotland, where subsidence effects were found to extend only 30 m into the surrounding undrained bog (Shotbolt et al., 1998). The difference in drainage impacts is consistent with the greater average thickness and typically higher lateral hydraulic conductivity of tropical compared to high-latitude peatlands, which increases their sensitivity to drainage (Evans et al., 2014; Baird et al., 2017). The observed relationship between $Subs_m$ and WTD_m is also evident in the (albeit small) number of subsidence estimates for tropical forest on peat shown in Table 2. Our forest WTD_m and $Subs_m$ values were both higher than those reported by Hooijer et al. (2012), based on a subset of our sites and a shorter dataset, possibly due to an increase in the proportion of forest subsidence poles close to plantations since the earlier analysis was undertaken. The 2015 El Niño dry season effect on more recent data may also have influenced our estimated mean subsidence rates, particularly in the shorter forest records; Hirano et al. (2012) measured water table drawdown and net CO_2 loss in an undrained forest during two previous (less severe) El Niño years. Brady (1997) also observed localised subsidence in some areas of natural peat swamp forest in Pulau Padang (part of our study area, Fig. 1) during a 3.5 year period following a strong El Niño in 1987, suggesting that peat surface elevation is sensitive to climatic disturbances even in undisturbed areas. On the other hand, Nagano et al. (2013) reported apparent subsidence in an area of intact forest where mean WTD lies above the ground surface.

The precise extent to which plantation drainage effects extend into the forest are difficult to determine. The data shown in Fig. 2 suggest a levelling off of both WTD_m and $Subs_m$ beyond 300 m, but neither approach zero even at the measurement points furthest from the forest-plantation boundary (2500 m). At this stage, we cannot draw robust conclusions regarding the impacts of plantation drainage beyond 300 m. Previous modelling has suggested that entire peat domes may be susceptible to subsidence as a result of drainage (Cobb et al., 2017), and it is possible that the increased topographic gradient resulting from subsidence of surrounding plantations has increased surface runoff in the central forest during the wet season, leading to lower dry-season WTDs. On the other hand, the lack of clear gradients in either WTD_m or $Subs_m$ beyond 300 m may argue against a direct plantation drainage impact. Most of the remote forest subsidence monitoring sites included in our analysis were established relatively recently, and these records may have been disproportionately impacted by the severe 2015 El Niño drought. Monitoring poles established at more remote forest locations within the last three years have not shown evidence of subsidence to date (A. Greer, pers. comm.), supporting this interpretation. Finally, methodological artefacts cannot be discounted; these are discussed below.

Taking account of the uncertainties, our data show a clear effect of plantation canals on water table drawdown and subsidence within native forest areas within 300 m of the plantation boundary, whilst evidence of effects beyond that distance is equivocal. Drying out of peat beneath natural forest close to plantations has been invoked as a contributor to the 2015 El Niño peat fires, but these fires were almost always also associated with human encroachment (e.g. along roads or waterways; Hoscilo et al., 2011; Cattau et al., 2016). A more direct effect of drainage on forest condition can occur via lowering of the peat surface, leading to increased instability and mortality of larger trees in particular (Brady, 1997). One implication of our results is that the impacts of plantation drainage on adjacent forests will depend strongly on contiguous forest area. For example if we conceptualise a circular, plantation-fringed forest with the same total area as the Kampar forest (300,000 ha), and assume a 300 m zone of clear drainage impact around the periphery (subject to the uncertainties in this figure noted above), plantation drainage would affect 2% of the total forest area. If the same forest was split into ten circular 30,000 ha units within a plantation landscape, approximately 6% of the total forest area would be drainage-affected. A 3000 ha area would be 20% affected, and a 300 ha fragment would be 52% affected. Since APRIL's plantations typically incorporate a 300 m buffer area inside the concession boundary, the data suggest that the greatest impacts of plantation management will occur within this native forest buffer zone.

4.3. Methodological issues and uncertainties

The subsidence monitoring dataset analysed for this study is, to our knowledge, the largest available for any peatland area globally. This has provided a unique opportunity to analyse spatial controls on subsidence rate and land-use within a relatively homogenous region, based on consistent measurement and analytical methods. However, as with any large monitoring study, there is the potential for measurement method, recorder error, site disturbance or data analysis method to generate biased or erroneous results. By applying two different analytical approaches, we sought to reduce the risk of methodological biases in the data analysis component. Whilst the source datasets were the same in both cases, the two analyses were carried out independently, by different teams, based on different approaches to data screening, and used different methods to derive mean subsidence estimates. Each method was considered to have advantages and disadvantages.

The site-based Method 1, based on linear regression of elevation versus time for each pole, provides a relatively robust single value for each location using all available data, and can (as in this study) be related directly to other site attributes such as mean WTD, vegetation type or stand rotation. On the other hand it is sensitive to anomalies resulting from recorder errors or datum shifts (e.g. following removal and replacement of a pole) and required careful screening of each individual dataset. We restricted this approach to datasets with a minimum of three years of data and 12 individual measurements in order to obtain robust subsidence rates. This precluded the use of data from recently established poles, which are primarily located in areas of conservation forest. We did not attempt to weight sites in this analysis according to length of record, or to take account of differences in the time periods when data were collected, although longer records would be expected to provide the most reliable subsidence estimates. The use of linear regression factors out any shorter-term effects on subsidence rate linked to management (e.g. harvesting, sequential rotations), seasonal hydrological variations (e.g. Fritz et al., 2008) or inter-year climate variability (e.g. El Niño years), thus providing an estimate of the long-term subsidence rate but no information on temporal variability.

The individual measurement-based Method 2 treated all individual quarterly subsidence measurements equally, and had the advantage that an objective, automated method could be used to exclude outlier values. This method also provides a very large dataset from which to derive mean subsidence rates, and implicitly gives greater weight to longer records, although caution is needed in analysing and interpreting repeated measurements from the same locations. This method may be expected to produce the most accurate overall estimates of mean subsidence rate, but is more limited with regard to the analysis of spatial variation in subsidence relative to other site attributes. On the other hand the method provides insights into drivers of temporal variation in subsidence rates, which will be explored in a subsequent paper. Overall, the similar mean subsidence rates obtained by the two methods for both plantation and forest areas provides some confidence that our estimates are not strongly influenced by the method of analysis.

With regard to the interpretation of spatial variations in subsidence rates, we were constrained by some elements of the explanatory data used. Notably, whilst quarterly subsidence represents a cumulative measure of change over the measurement periods, the accompanying WTD measurement is only a snapshot of conditions at the time of measurement, and may fail to capture average conditions over the preceding three-month period. However, by aggregating all quarterly WTD data for the full period of measurements, and correcting for differences in the number of measurements per quarter, we sought to minimise this issue. In calculating distance from the nearest canal within plantation areas we were not able to take account of proximity to within-compartment field drains, which may have influenced local WTD and subsidence rates. The use of a simple measure of distance to nearest plantation/forest boundary for the forest sites assumed that this was a good proxy for distance to the nearest drainage feature, which was true in most but not all cases. Substituting distance to canal for distance to plantation boundary did not materially change the observed relationships, however (data not shown). The approach used also did not take account of local topography, which could also play a role in determining the distance over which nearby plantation drainage influences forest hydrology. Further work including analysis of Earth Observation data and use of plantation-scale hydrological models should help to further resolve spatial controls on subsidence rate.

A key remaining uncertainty in the dataset relates to the possibility of measurement-related artefacts in the subsidence dataset. Due to their very low bulk density, peatlands are highly sensitive to compaction caused by disturbance. Whilst efforts were made to avoid exerting pressure on the peat surface during pole installation or subsequent quarterly measurements, as described in the methods, some degree of measurement-related compaction cannot be ruled out. By protecting the subsidence poles during five-yearly harvesting and replanting operations, on the other hand, it is possible that local compaction of peat by machinery could have been lower than elsewhere in the plantations. The relatively high scatter and low (albeit significant) R^2 obtained from the plantation-only regression of $Subs_m$ against WTD_m (Eq. (1)), compared to the forest-only and whole-dataset regressions, may in part be due to these methodological issues, although it is also likely that other unmeasured environmental factors contribute to the high level of unexplained variation.

In forest areas, management-related disturbance was not an issue, however disturbance during measurements could have had a proportionally greater effect (especially in areas of undisturbed natural forest) because bulk density is low and prior disturbance negligible. As noted above, this could have been a contributory factor in the non-zero subsidence rates recorded at forest sites distant from plantation areas. The similar (0.7 cm yr^{-1}) rate of subsidence recorded by Nagano et al. (2013) at a site with mean water levels above the peat surface (Table 2) lends some support to this interpretation. However we cannot rule out the possibility either that even remote forest sites have been affected by plantation drainage further down the peat dome, or by historical logging activities (often involving canals), or alternatively that these areas were subject to natural subsidence during the measurement period, for example linked to the 2015 El Niño (when rainfall levels at the nearby Pekanbaru meteorological station were 27% below the long-term mean). The relative importance of these different potential explanatory factors should become clearer as monitoring records for these sites become longer.

4.4. The potential for mitigating subsidence through altered plantation management

Our analysis shows significant ($p < 0.001$) spatial relationships between subsidence rate and mean annual WTD, which could explain varying amounts of observed variation ($R^2 = 0.09$ in the plantation-only data, 0.45 in the forest-only data, and 0.25 in the combined dataset). This suggests that alterations in plantation management that allow average water tables to be maintained closer to the peat surface should generate a commensurate reduction in subsidence rate, on average by 0.04 cm yr^{-1} per 1 cm water table increase. This conclusion is consistent with previous assessments of the relationship between subsidence and water table (Schothorst, 1977; Stephens et al., 1984; Andriesse, 1988; Hooijer et al., 2012; Ishikura et al., 2018) and with broader understanding of the relationship between peat oxidation and aeration depth (e.g. Gorham, 1991). The dataset shows a wide range of mean water tables across the plantation area, which reflects the challenges of maintaining a uniform target drainage depth across a topographically and hydrologically complex landscape. By reducing the size of water management zones within plantations (effectively reducing the elevation difference gradient between each dam) it should be possible to reduce spatial variability in water tables, although local topographic variation, the use of canals for transportation, and seasonal variations in water availability all place limits on the extent to which spatially and temporally uniform water levels can be achieved in practice. On the other hand, the presence of relatively high-WTD sites in the plantation dataset (Fig. 2) suggests that *Acacia* cultivation at mean *WTD*s $<$ 50 cm may be viable. Challenges remain, however, with respect to maintaining stand survival, stability and nutrition under wetter conditions, and (given very high evapotranspiration rates) in sustaining high water levels during extended drought periods. If these challenges could be overcome, our data suggest that reducing mean WTD in the *Acacia* plantations from 70 cm to 40 cm would reduce mean subsidence from $-4.3\ \mathrm{cm\ yr^{-1}}$ to around -3 to $-3.2\ \mathrm{cm\ yr^{-1}}$ (estimates based on whole-dataset and plantation-only regressions, respectively). This 25–30% reduction in subsidence would, assuming it translates into an equivalent reduction in CO_2 emissions, be approximately in line with the Indonesian Government's 'unconditional' emissions reduction target of 29% below business-as-usual by 2030 (Government of Indonesia, 2015). At the same time, our data clearly show a high degree of scatter around the subsidence vs *WTD* relationship, implying that other factors must influence subsidence rate. Whilst we observed lower average subsidence rates in later rotation *Acacia* stands, these differences were non-significant, suggesting that time since drainage alone cannot explain observed spatial variability. We were unable to identify any other site- or management-related variable that appeared to influence subsidence rate. There is therefore an ongoing need to better understand the links between site characteristics, management and subsidence in order to reduce subsidence (thus reducing CO_2 emissions and extending plantation lifetimes) whilst also maintaining economic yields.

One unresolved question for all forms of plantation management on peatlands is whether subsidence would occur even in the absence of drainage. Based on similar relationships to those shown in Fig. 2 of our study, Hooijer et al. (2012) argued that some subsidence was inevitable even if water levels could (in theory) be raised to the peat surface, due to management-related disturbances and reduced vegetation cover. The relatively high degradability of litter produced by plantation species compared to native swamp forest, and removal of harvested biomass that might otherwise have contributed to peat formation, could also contribute to subsidence even at high water tables. According to our analysis of the plantation-only dataset (Eq. (2)), subsidence would still be $1.9\ \mathrm{cm\ yr^{-1}}$ if the water table could theoretically be raised to the surface. If the full plantation-plus-forest dataset is used (Eq. (1)) this value falls to $1.2\ \mathrm{cm\ yr^{-1}}$, and we obtained a similar (albeit non-significant) intercept from our analysis of published tropical peatland subsidence rates (Eq. (6)). If we restrict our analysis to forest sites only, the intercept falls to just $0.4\ \mathrm{cm\ yr^{-1}}$. These findings appear to support the suggestion that plantation-managed peatlands may be subject to some 'baseline' level of subsidence even without drainage, whereas subsidence should be low or zero in saturated peatlands that retain a wetland-adapted native species cover. However, this conclusion is subject to a number of caveats. Firstly, the plantation-only dataset included very few sites with WTD $<$ 40 cm, and had high scatter at intermediate water tables, making the extrapolation of subsidence rates to a zero WTD highly uncertain. Secondly, where plantation and forest sites had similar water tables, we found little evidence of a consistent offset in subsidence rates (Fig. 2). This could suggest that drained forests and plantations are not intrinsically different despite greater disturbance and biomass offtake in the latter, although the high level of observed scatter, especially in the plantation dataset, makes this interpretation rather uncertain. Finally, extrapolating plantation subsidence rates to a WTD of zero has limited meaning at present, because the *Acacia* species currently used for pulp production are not grown in permanently saturated, anoxic peat. Work is ongoing to develop management systems and/or alternative species to enable fibre production on wetter peat, however it is worth noting that some of the factors contributing to subsidence under *Acacia* plantation (i.e. disturbance, biomass offtake) would likely apply to any fibre or food crop grown on peat, including high-water table adapted native tree species or 'paludiculture' crops. Other factors, notably the amount of mechanical disturbance associated with harvesting and replanting on a five-year rotation, might be reduced, but in this case biomass yields and thus economic returns would also likely be lower.

5. Conclusions

We believe that the results of this study, comprising direct subsidence measurements from 312 locations, and over 2000 site years of data (considerably more than the sum of all previous subsidence assessments on tropical peat as summarised in Table 2) provide the most robust current basis for evaluating the impact of drainage on tropical peat subsidence. Mean subsidence rates under *Acacia* of $4.2–4.3\ \mathrm{cm\ yr^{-1}}$ are somewhat lower than previously reported, with implications for plantation longevity and carbon emissions, but nevertheless substantial. Drainage effects were most evident within 300 m of the nearest plantation boundary, and thus had the greatest impact on native forest buffer zones within concession areas. Whilst some larger-scale impacts cannot be ruled out, these 'intensive' drainage-related impacts on larger forest areas are likely to be limited to a few percent of the total area, whilst impacts in smaller forest fragments will be greater. Our data suggest that regulating mean water tables in plantations at a mean of 40 cm could – provided that an economically and practically viable system of doing so can be developed – reduce current mean subsidence rates by 25–30%, helping to meet national emission reduction targets whilst also extending plantation lifetimes and reducing ecological impacts on neighbouring forests. At the same time, we acknowledge the importance of balancing the mitigation of environmental impacts with the need to sustain economic yields in order to support the large and growing populations who currently rely on various forms of peatland cultivation for their livelihoods. If new plantation management techniques or tree species can sustain biomass yields at higher water tables, and water management methods refined to sustain higher water tables during dry periods, this would deliver benefits both to the environment and to the longer-term economy of tropical peatland regions. Finally, we note that subsidence is not only a challenge for tropical peatlands or developing countries, because measured subsidence rates were $1–2\ \mathrm{cm\ yr^{-1}}$ even in high-latitude peatlands that have been drained for over a century. Therefore, as highlighted by Leifeld and Menichetti (2018), the need to develop land-management strategies to mitigate subsidence and CO_2 emissions from drained peatlands represents a global as well as a regional challenge.

Acknowledgements

Work by CE, SP and AL was undertaken as part of their contribution to APRIL's Independent Peat Expert Working Group (IPEWG), with support for data analysis from JW, who undertook the 'Method 1' analysis with CE. DI and FK were commissioned separately by APRIL to carry out an independent analysis of the data ('Method 2'). YS and MFH provided measurement data and undertook the spatial data analysis. The contributions of CE and JW were also supported by the NaUNRISE programme. We are grateful to APRIL's staff for support with the analysis, notably Antony Greer and Tim Fenton, and to the many people who have helped to install and maintain subsidence monitoring since 2007.

Declaration of interest

CE, SP and AL undertook the analysis described in this paper as part of their contribution to the Independent Peat Expert Working Group (IPEWG), which was set up and funded by Asia Pacific Resources International Ltd. (APRIL) to provide objective science-based advice on peatland management. JW's contribution was also supported by APRIL via the IPEWG. YS and MFH are employees of APRIL, and FK and DI were separately commissioned by APRIL to provide an independent analysis of the subsidence dataset. All views expressed are the authors' own.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.geoderma.2018.12.028.

References

Aich, S., McVoy, C.W., Dreschel, T.W., Santamaria, F., 2013. Estimating soil subsidence and carbon loss in the Everglades Agricultural Area, Florida using geospatial techniques. Agric. Ecosyst. Environ. 171, 124–133.

Andriesse, J.P., 1988. Nature and management of tropical peat soils. FAO Soils Bull. 59.

Aswandi, Susanto R.H., Saleh, E., Abdillah, M.R., Iskandar, I., 2017. Simulation of CO_2 emission and land subsidence in reclaimed tidal peat swamp in Berbak Delta, Jambi-Indonesia. Int. J. GEOMATE 12, 186–194.

Baird, A.J., Low, R., Young, D., Swindles, G.T., Lopez, O.R., Page, S.E., 2017. High permeability explains the vulnerability of the carbon store in drained tropical peatlands. Geophys. Res. Lett. 44, 1333–1339.

Billor, N., Hadi, A.S., Velleman, P.F., 2000. BACON: blocked adaptive computationally efficient outlier nominators. Comput. Stat. Data Anal. 34, 279–298.

Brady, M.A., 1997. Organic Matter Dynamics of Coastal Peat Deposits in Sumatra, Indonesia (PhD thesis). University of British Colombia (258 pp.).

Brunning, R., 2001. Archaeology and peat wastage on the Somerset Moors. In: Report to the Environment Agency. Somerset County Council, Taunton.

Cattau, M.E., Harrison, M.E., Shinyo, I., Tungau, S., Uriarte, M., DeFreis, R., 2016. Sources of anthropogenic fire ignitions on the peat-swamp landscape in Kalimantan, Indonesia. Glob. Environ. Chang. 39, 205–219.

Ciais, P., Sabine, C., Bala, G., Bopp, L., Brovkin, V., Canadell, J., Chhabra, A., DeFries, R., Galloway, J., Heimann, M., Jones, C., Le Quéré, C., Myneni, R.B., Piao, S., Thornton, P., 2013. Carbon and Other Biogeochemical Cycles. In: Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. In: Stocker, T.F., Qin, D., Plattner, G.-K., Tignor, M., Allen, S.K., Boschung, J., Nauels, A., Xia, Y., Bex, V., Midgley, P.M. (Eds.), Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Cobb, A.R., Hoyt, A.M., Gandois, L., Jangarun, E., Dommain, R., Salim, K.A., Kai, F.M., Su'ut, N.S.H., Harvey, C.F., 2017. How temporal patterns in rainfall determine the geomorphology and carbon fluxes of tropical peatlands. Proc. Natl. Acad. Sci. 114, E5187–E5196.

Couwenberg, J., Hooijer, A., 2013. Towards robust subsidence-based soil carbon emission factors for peat soils in south-east Asia, with special reference to oil palm plantations. Mires Peat 12 (01).

Dargie, G.C., Lewis, S.L., Lawson, I.T., Mitchard, E.T.A., Page, S.E., Bocko, Y.E., Ifo, S.A., 2017. Age, extent and carbon storage of the central Congo Basin peatland complex. Nature 542, 86–90.

Dawson, Q., Kechavarzi, C., Leeds-Harrison, P.B., Burton, R.G.O., 2010. Subsidence and degradation of agricultural peatlands in the Fenlands of Norfolk, UK. Geoderma 154, 181–187.

De Maesschalk, R., Jouan-Rimbaud, D., Massart, D.L., 2000. The Mahalanobis distance. Chemom. Intell. Lab. Syst. 50, 1–18.

Deverel, S.J., Leighton, D.A., 2010. Historic, recent, and future subsidence, Sacramento-San Joaquin Delta, California, USA. San Francisco Estuary Watershed Sci. 8, 1–23.

Deverel, S.J., Ingrum, T., Leighton, D., 2016. Present-day oxidative subsidence of organic soils and mitigation in the Sacramento-San Joaquin Delta, California, USA. Hydrogeol. J. 24, 569–586.

DID & LAWOO, 1996. Western Jahore integrated agricultural development project. In: Peat Soil Management Study. Department of Irrigation and Drainage (DID), Kuala Lumpur, Malaysia and Land and Water Research Group (LAWOO), Wageningen.

Dommain, R., Couwenberg, J., Glaser, P.H., Joosten, H., Suryadiputra, I.N.N., 2014. Carbon storage and release in Indonesian peatlands since the last deglaciation. Quat. Sci. Rev. 97, 1–32.

Eggelsman, R., 1976. Peat consumption under influence of climate, soil condition and utilization. In: Proceedings of the Fifth International Peat Congress, Poznań, Poland. 1. pp. 233–247.

Eggelsmann, R., Bartels, R., 1975. Oxidativer torfverzehr im niedermoor in abhängigkeit von entwässerung, nutzung und dünung. Mitt. Dtsch. Bodenkundlichen Ges. 22, 215–221.

Evans, C.D., Page, S.E., Jones, T., Moore, S., Gauci, V., Laiho, R., Hruška, J., Allott, T.E.H., Billett, M.F., Tipping, E., Freeman, C., Garnett, M.H., 2014. Contrasting vulnerability of drained tropical and high-latitude peatlands to fluvial loss of stored carbon. Glob. Biogeochem. Cycles 28, 1215–1234.

Fitzgerald, N., McLeod, M., 2004. Subsidence rates of peat since 1924 in the Rukuhia Swamp. In: Technical report 2004/20. Environment Waikato, Waikato, New Zealand.

Fritz, C., Campbell, D.I., Schipper, L.A., 2008. Oscillating peat surface levels in a restiad peatland, New Zealand—magnitude and spatiotemporal variability. Hydrol. Process. 22, 3264–3274.

Gorham, E., 1991. Northern peatlands: role in the carbon cycle and probable responses to climatic warming. Ecol. Appl. 1, 182–195.

Government of Indonesia, 2015. Intended nationally determined contribution, Republic of Indonesia. In: Submission to the United Nations Framework Convention on Climate Change, . http://www4.unfccc.int/submissions/INDC/Published%20Documents/Indonesia/1/INDC_REPUBLIC%20OF%20INDONESIA.pdf.

Grønlund, A., Hauge, A., Hovde, A., Rasse, D.P., 2008. Carbon loss estimates from cultivated peat soils in Norway: a comparison of three methods. Nutr. Cycl. Agroecosyst. 81, 157–167.

Grzywna, A., 2017. The degree of peatland subsidence resulting from drainage of land. Environ. Earth Sci. 76, 559.

Hirano, T., Segah, H., Kusin, K., Limin, S., Takahashi, H., Osaki, M., 2012. Effects of disturbances on the carbon balance of tropical peat swamp forests. Glob. Chang. Biol. 18, 3410–3422.

Hooijer, A., Page, S., Jauhiainen, J., Lee, W.A., Lu, X.X., Idris, A., Anshari, G., 2012. Subsidence and carbon loss in drained tropical peatlands. Biogeosciences 9, 1053–1071.

Hoscilo, A., Page, S.E., Tansey, K.J., Rieley, J.O., 2011. Effect of repeated fires on land-cover change on peatland in southern Central Kalimantan, Indonesia, from 1973 to 2005. Int. J. Wildland Fire 20, 578–588.

Hutchinson, J.N., 1980. Record of peat wastage in the East-Anglian fenlands at Holme Post, 1848–1978 AD. J. Ecol. 68, 229–249.

Ishikura, K., Hirano, T., Okimoto, Y., Hirata, R., Kiew, F., Melling, L., Aeries, E.B., Lo, K.S., Musin, K.K., Waili, J.W., Wong, G.X., Ishii, Y., 2018. Soil carbon dioxide emissions due to oxidative peat decomposition in an oil palm plantation on tropical peat. Agric. Ecosyst. Environ. 254, 202–212.

Jauhiainen, J., Hooijer, A., Page, S.E., 2012. Carbon dioxide emissions from an Acacia plantation on peatland in Sumatra, Indonesia. Biogeosciences 9, 617–630.

Jauhiainen, J., Kerojoki, O., Silvennoinen, H., Limin, S., Vasander, H., 2014. Heterotrophic respiration in drained tropical peat is greatly affected by temperature – a passive ecosystem cooling experiment. Environ. Res. Lett. 9, 105013.

Jongedyk, H.A., Hickock, R.B., Mayer, I.D., Ellis, N.K., 1950. Subsidence of muck soils in northern Indiana. Purdue University Agriculture Experimental Station Special Circular, pp. 366.

Khasanah, N., van Noordwijk, M., 2018. Subsidence and carbon dioxide emissions in a smallholder peatland mosaic in Sumatra, Indonesia. Mitig. Adapt. Strateg. Glob. Chang. https://doi.org/10.1007/s11027-018-9803-2.

Kluge, B., Wessolek, G., Facklam, M., Lorenz, M., Schwarzel, K., 2008. Long-term carbon loss and CO_2-C release of drained peatland soils in northeast Germany. Eur. J. Soil Sci. 59, 1076–1086.

Kool, D.M., Buurman, P., Hoekman, D.H., 2006. Oxidation and compaction of a collapsed peat dome in Central Kalimantan. Geoderma 137, 217–225.

Leifeld, J., Menichetti, L., 2018. The underappreciated potential of peatlands in global climate change mitigation strategies. Nat. Commun. 9, 1071.

Leifeld, J., Müller, M., Fuhrer, J., 2011. Peatland subsidence and carbon loss from drained temperate fens. Soil Use Manag. 27, 170–177.

Mathur, S.P., Levesque, M.P., Richard, P.J.H., 1982. The establishment of synchrony between subsurface layers and estimation of overall subsidence of cultivated organic soils by a palynological method. Can. J. Soil Sci. 62, 427–431.

Miettinen, J., Shi, C., Liew, S.C., 2016. Land cover distribution in the peatlands of Peninsular Malaysia, Sumatra and Borneo in 2015 with changes since 1990. Glob. Ecol. Conserv. 6, 67–78.

Miettinen, J., Hooijer, A., Vernimmen, R., Liew, S.C., Page, S.E., 2017. From carbon sink to carbon source: extensive peat oxidation in insular Southeast Asia since 1990. Environ. Res. Lett. 12, 024014.

Millette, J.A., 1976. Subsidence of an organic soil in southwestern Québec. Can. J. Soil Sci. 56, 499–500.

Minkinnen, K., Laine, K., 1998. Long-term effect of forest drainage on the peat carbon stores of pine mires in Finland. Can. J. For. Res. 28, 1267–1275.

Minkinnen, K., Vasander, H., Jauhianen, S., Karsisto, M., Laine, J., 1999. Post-drainage changes in vegetation composition and carbon balance in Lakkasuo mire, Central Finland. Plant Soil 207, 107–120.

Mirza, C., Irwin, R.W., 1964. Determination of subsidence of an organic soil in southern Ontario. Can. J. Soil Sci. 44, 248–253.

Nagano, T., Osawa, K., Ishida, T., Sakai, K., Vijarnsorn, P., Jongskul, A., Phetsuk, S., Waijaroen, S., Yamanoshita, T., Norisada, M., Kojima, K., 2013. Subsidence and soil CO_2 efflux in tropical peatland in southern Thailand under various water table and management conditions. Mires Peat 11, 06.

Othman, H., Mohammed, A.T., Darus, F.M., Harun, M.H., Zambri, M.P., 2011. Best management practices for oil palm cultivation on peat: ground water-table maintenance in relation to peat subsidence and estimation of CO_2 emissions at Sessang, Sarawak. J. Oil Palm Res. 23, 1078–1086.

Page, S.E., Rieley, J.O., Banks, C.J., 2011. Global and regional importance of the tropical peatland carbon pool. Glob. Chang. Biol. 17, 798–818.

Pinheiro, J., Bates, D., DebRoy, S., Sarkar, D., Core Team, R., 2016. nlme: Linear and Nonlinear Mixed Effects Models. http://CRAN.R-project.org/package=nlme.

Pronger, J., Schipper, L.A., Hill, R.B., Campbell, D.I., McLeod, M., 2014. Subsidence rates of drained agricultural peatlands in New Zealand and the relationship with time since drainage. J. Environ. Qual. 43, 1442–1449.

R core team, 2015. R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria.
Richardson, S.J., Smith, J., 1977. Peat wastage in East Anglian fens. J. Soil Sci. 28, 485–489.
Schipper, L.A., McLeod, M., 2002. Subsidence rates and carbon loss in peat soils following conversion to pasture in the Waikato Region, New Zealand. Soil Use Manag. 18, 91–93.
Schothorst, C.J., 1977. Subsidence of low moor peat soils in western Netherlands. Geoderma 17, 265–291.
Shih, S.F., Glaz, B., Barnes, R.E., 1998. Subsidence of organic soils in the Everglades agricultural area during the past 19 years. Soil and Crop Sci. Soc. Florida Proc. 57, 20–29.
Shotbolt, L., Anderson, A.R., Townend, J., 1998. Changes to blanket bog adjoining forest plots at Bad a' Cheo, Rumster Forest, Caithness. Forestry 71, 311–324.
Stephens, J., Allan, L., Chen, E., 1984. Organic soil subsidence. Rev. Eng. Geol. 6, 107–122.
Wakhid, N., Hirano, T., Okimoto, Y., Nurzakiah, S., Nursyamsi, D., 2017. Soil carbon dioxide emissions from a rubber plantation on tropical peat. Sci. Total Environ. 581-582, 857–865.
Weber, S., 2010. bacon: an effective way to detect outliers in multivariate data using Stata (and Mata). Stata J. 10, 331–338.
Wijedasa, L.S., Jauhianen, J., Könönen, M., et al., 2016. Denial of long-term issues with agriculture on tropical peatlands will have devastating consequences. Glob. Chang. Biol. 23, 977–982.
Wösten, J.H.M., Ismail, A.B., Vanwijk, A.L.M., 1997. Peat subsidence and its practical implications: a case study in Malaysia. Geoderma 78, 25–36.
Wright, A.L., Snyder, G.H., 2009. Soil Subsidence in the Everglades Agricultural Area. Everglades Research and Education Center, Belle Glade, FL.
Zanello, F., Teatini, P., Putti, M., Gambolati, G., 2011. Long term peatland subsidence: experimental study and modeling scenarios in the Venice coastland. J. Geophys. Res. 116, F04002.