Surface peat and its dynamics following drainage - do they facilitate estimation of carbon losses with the C/ash method?

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SUMMARY

Peatlands are under pressure, and information about the impacts of land use and climate change on their carbon (C) balance is needed. Quick and inexpensive methods that rely on changes in the ash and C contents of surface peat have been suggested, on the basis of consistent changes observed from repeated measurements in peatlands under agricultural use. More general applicability has been claimed for such methods; however, their application requires a thorough understanding of the processes that shape the ash and C contents following drainage. In this article, we review the characteristics and dynamics of the surface peat following drainage in cases where the sites remain vegetated and follow the secondary succession induced by drainage. We conclude that methods which merely examine the ash and C contents of surface peat samples will generally not lead to reliable information about the C balances of the sites but, rather, to arbitrary values incorporating unknown proportions of several non-random errors. The main challenges are to find truly appropriate reference samples, and to ensure that the samples from the site of interest have not been modified by processes other than decomposition. These challenges arise from several physical, chemical and biological processes that shape the surface peat following drainage, in addition to the decomposition process. Even though the theoretical basis of C/ash methods may be sound, it is virtually impossible to find vegetated sites where decomposition-induced changes in the C/ash quotient would not be masked by the outcomes of the other processes. Thus, in most cases, the C/ash method involves not only estimable random errors but also serious non-random errors that cannot be taken into account.

KEY WORDS: C/ash method, carbon balance, drainage, surface peat.

INTRODUCTION

Peatlands cover just 3 % of the Earth's surface, but contain 30 % of all soil carbon (C). Peat soils accumulate under water-saturated, anoxic conditions where anaerobic decomposition is slow. The use of peatlands for agriculture, forestry and different plantations (such as oil palm) has been widespread (e.g., Joosten & Clarke 2002, Čížková *et al.* 2013) and shows no signs of abating as wide areas of previously inaccessible tropical peatlands are increasingly converted to different land uses (Miettinen *et al.* 2016). Most uses require drainage to lower the water-table level so that the roots of plants receive enough oxygen. Climate change threatens to lower water-table levels even in pristine peatlands (Gorham 1991). Lowered water-table levels re-expose already-accumulated organic matter to aerobic decomposition, while warmer temperatures enhance microbial metabolism and thus may increase the rate of decomposition. Consequently, the fate of the huge C reserves of peat is a concern to scientists around the globe (Dise 2009, Hirano *et al.* 2009, Fenner & Freeman 2011).

Evidence has slowly but steadily accumulated as to how C cycling and the C balance of different peatland types under varying climatic conditions respond to changes in land use and climate. Much of this is based on methodologies that rely on the quantification of C gas exchange between the soil and the atmosphere (e.g., Ojanen *et al.* 2013, Petrescu *et al.* 2015). Progress has been relatively slow, largely because we have only gradually learned how to capture all C inputs and outputs. For instance, the first attempts to evaluate C losses from peat soils were made by measuring total carbon dioxide (CO_2) emissions from the soil (Armentano & Menges 1986, Silvola 1986). However, emissions encompass not only the heterotrophic respiration that is derived from the net decomposition of the accumulated peat, but also the heterotrophic respiration from annually replenished fresh plant litter inputs (Straková *et al.* 2012), autotrophic respiration of plant roots and rhizomes (Silvola *et al.* 1996), and the CO_2 that is derived from methane (CH_4) production and oxidation. It follows that total soil respiration tells us nothing about the C balance of the soil. Later, procedures that separate the different components of soil respiration have been developed, based on the isotopic composition of CO_2 (e.g., Crow & Wieder 2005), or as a sequence of measurement plots that successively exclude different components (Minkkinen *et al.* 2007). Currently, the net C exchange, which takes into account CO_2 uptake through photosynthesis (e.g., Lohila *et al.* 2011) is often estimated with the eddy covariance method (e.g., Aubinet *et al.* 2012); however, specific component fluxes, such as litter production or heterotrophic respiration from peat decomposition, and water-borne C losses still need to be estimated separately to obtain a complete picture of the C cycle and balance. In general, reliable results require several years of observations under varying weather conditions (e.g., Lund *et al.* 2012, Peichl *et al.* 2014, Helfter *et al.* 2015), and all widely accepted methods are rather laborious.

A general limitation of the gas exchange methodologies is that they capture only current patterns. This fulfils the needs of the greenhouse gas inventories required by international climate treaties, for example, or of ecosystem models operating at short time steps. However, they do not reveal cumulative impacts and their dynamics over the whole period since water-level drawdown. There have been attempts to capture these by resampling sites, with earlier data facilitating the estimation of soil C content (Minkkinen & Laine 1998, Simola *et al.* 2012); by comparing undrained and drained sides along a boundary ditch (Minkkinen *et al.* 1999, Pitkänen *et al.* 2013); and by using gas exchange methodologies with the chronosequence (time–space substitution) approach (Hargreaves *et al.* 2003). These approaches are laborious, and involve uncertainties relating to the comparability of samples and/or sites.

Other methods have been suggested to reduce the workload and level of expertise required, and to enable more extensive inventories and faster results for quantifying the cumulative post-drainage impacts on the C store. One such method, based on C/ash quotients (henceforward “ash method”), was presented by Grønlund *et al.* (2008) and seemingly successfully applied in soils under continued (and well-controlled) agricultural use. The method was based on the progressive increase in surface peat mineral (ash) concentration due to continuing decomposition-induced C losses following drainage and cultivation, documented through repeated measurements. Theoretically, the method seems plausible, since we know that in decomposition, C is lost from the peat soil, while the majority of the soil ash content may be silica (e.g., Jinming & Xuehui 2009) or other such material that is not immediately used by either microbes or plants as the main nutrients may be, and would thus accumulate in the soil. The authors recognised potential sources of error, e.g., inhomogeneity of the peat. Yet, they recommended the ash method for wider use because it is easy and cheap, and suggested that it could even be used where repeated measurements were lacking, by using a nearby undrained peatland site, or a deeper layer of the drained site itself, as a reference for the pre-drainage peat mineral content. Accordingly, the method, with varying degrees of modification, has since been applied in more complex systems: grasslands (Rogiers *et al*. 2008, Krüger *et al.* 2015), drained bogs (Leifeld *et al.* 2011, methods I, II in Krüger *et al.* 2016, Wüst-Galley *et al.* 2016) and drained fens (Kareksela *et al.* 2015, Krüger *et al.* 2016). Moreover, instead of repeated measurements, one-time samplings of drained sites and independent undrained control sites have been used (Kareksela *et al.* 2015). These were further tested using concentrations of single elements in addition to ash concentration.

When any version of the ash method is applied, one should be well aware of all factors other than decomposition that will affect the C/ash quotient of peat. The peat soil is such a dynamic system that a lack of comprehensive understanding of the processes that take place in the soil matrix following drainage and/or land-use change may lead researchers astray when the method is applied under more complex situations than well-controlled agricultural fields. Our aim here is to review characteristics and processes that affect surface peat in ways that challenge the estimation of the soil C balance after land-use change by just examining the peat itself. These factors include at least the following:

1: variation in ash and element concentrations between peatland sites and site types;

2: spatial variation in ash and element concentrations within sites;

3: vertical distribution of ash and element concentrations;

4: physical processes that shape the surface peat following drainage; and

5: biological processes that shape the surface peat following drainage.

Each of these factors will be discussed, in turn, in the next section. We shall focus our examination on permanently vegetated non-agricultural sites, where the soil profile is not affected by ploughing or any other means of soil preparation.

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FACTORS TO CONSIDER

1: Variation in ash and element concentrations between peatland sites and site types

Information as to how ash or element concentrations vary between sites that represent the same floristically defined site type or broader peatland type is crucial when determining whether or not a particular site can be used as a reference for the evaluation of post-drainage changes in ash or element concentrations. This is best evaluated using data from extensive inventories. Several such datasets exist, and we will use examples from boreal peatlands in Finland throughout this paper.

Let us first examine inventory material that originates from undrained/undisturbed and drained sites collected in a climatically uniform region. The material includes sites that were sparsely treed minerotrophic fens and ombrotrophic bogs in their undrained condition. The sites form a drainage age sequence from undrained to sites drained 55 years before measurements, and provide a basis for the evaluation of drainage impacts through time–space substitution. The material has been described in detail by Laiho & Laine (1994, 1995) and Laiho *et al.* (1999). Ash concentrations (% of dry mass) were determined by the incineration of samples at 550 °C for two hours, and C concentrations (% of dry mass) were determined with a CHN-analyser (LECO, Stockport, UK); these results have not previously been published. Ash concentration generally depended on site type/site nutrient regime, being lower in ombrotrophic and transitional than in clearly minerotrophic sites (Figure 1, left). Overall, the variation in ash concentrations within both site groups was relatively high; the Coefficients of Variation ranged from 0.48 to 0.74 in the minerotrophic group and from 0.43 to 0.88 in the ombrotrophic group, depending on sampling depth. C/ash was generally higher in ombrotrophic and transitional sites and, likewise, showed rather high variation (Figure 1, right). The Coefficients of Variation for C/ash ranged from 0.38 to 0.47 in the minerotrophic group and from 0.35 to 0.43 in the ombrotrophic group. Such variation strongly suggests that the use of a single reference site or the comparison of one random sample that represents undrained conditions to another random sample that represents drained conditions involves a high proportion of random error. Furthermore, strong patterns related to the time passed since drainage were not evident; we will return to this later.

Similarly, in an extensive survey on sites drained for forestry across southern Finland (data from Westman & Laiho 2003), ash concentration in the surface peat (0–30 cm) depended on the peatland type (nutrient regime) (Figure 2). This material can be divided into narrower site types because of the larger number of sites *per* type. Ash concentration generally decreased with the site's nutrient status and was usually low - less than 4 % of peat dry mass - in ombrotrophic sites. In minerotrophic sites, the values were higher and more variable. Especially high values may result if the minerogenic waters that enter the peatland deposit mineral matter, e.g., during a flood of a nearby waterway. Fires or volcanic activity (e.g., Zoltai 1989) may result in high ash concentrations over limited depth ranges in all types of peatlands. Furthermore, the thickness of the peat deposit had an impact, with ash concentrations decreasing with increasing thickness at least up to 0.6–0.8 m (Figure 3). In this material as well, the ash concentration was not noticeably affected by the time since drainage within any of the site types, even when the effects of peat thickness and geographic location (latitude) were taken into account (data not shown).

Within any floristically defined peatland type, except perhaps the extremely nutrient-poor bogs, the variation in total concentrations of any individual element is high (Figure 4; see also Westman 1981, Laiho & Laine 1995). Similarly, the concentrations of extractable element concentrations vary widely (Starr & Westman 1978), which would indicate that the processes that retain and release elements may proceed differently in different sites. Following drainage for forestry, the high variability in element concentrations persists (Figure 4; see also Laiho & Laine 1994, 1995) and may be accompanied by trends over time, caused by factors discussed in Sections 3 and 4 below. Consequently, if one compares any chemical attribute of a peat sample from one random drained site to a peat sample from one random undrained site that represents the same peatland type, or even to the average situation in a random group of undrained sites, the difference includes an unknown but potentially high amount of random error. This alone seriously challenges the suggested applicability of the ash method that utilises reference sites instead of repeated measurements.

Another potentially problematic aspect when comparing separate, or even nearby, undrained and drained sites is identifying the original peatland type after the secondary succession initiated by drainage has had time to shape the prevailing vegetation. The wetter and more nutrient-rich the original site, the more rapid and extensive the vegetation succession is likely to be (Laine *et al.* 1995a). This also results in increasing similarity of the vegetation composition in sites where the nutrient regimes range from mesotrophic to oligotrophic (Laine *et al.* 1995a).

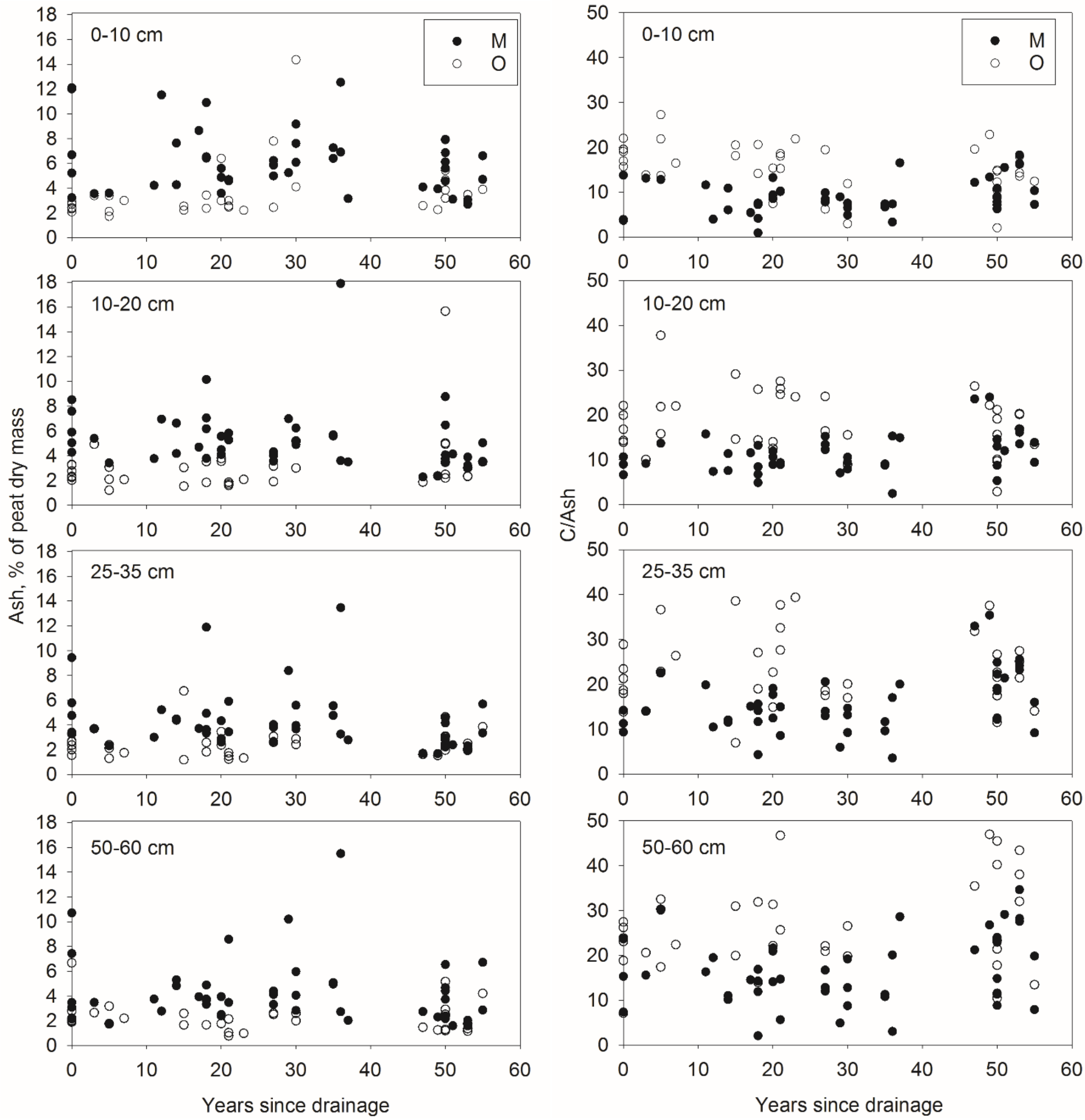

Figure 1. Variation in the ash concentration (% of peat dry mass; left) and the quotient of C and ash concentrations (right), at different sampling depths in two site type groups, namely: clearly minerotrophic sites (M), and ombrotrophic or transient (O) sites. Undrained sites (years since drainage = zero) and sites drained for forestry at different times were included. From the material of Laiho & Laine (1994, 1995); Laine *et al*. (1995a) and Laiho *et al*. (1999) utilised the same material. The M sites (n = 45) were meso-oligotrophic sparsely treed sedge fens with *Carex* peat, and the O sites (n = 37) were ombrotrophic pine bogs and transient sparsely treed sites with oligotrophic *Sphagnum* peat. In the ash panels, the following extreme values were excluded: layer 0–10 cm - a value of 35.3 for an M site drained 18 years before measurements, and a value of 19.6 for an O site drained 50 years before measurements; layer 50–60 cm - a value of 22.4 for an M site drained 18 years before measurements. In the panel for C/ash, layer 50–60 cm, the maximum value of 63.1 for an O site drained 21 years before measurements was excluded.

Since the changes in vegetation are greater in the more nutrient-rich sites, this means in practice that an inexperienced researcher is at high risk of selecting drained sites that are in fact more nutrient- and ash-rich than the undrained reference sites. Should this happen, the C losses will be grossly over-estimated by the ash method. It requires considerable expertise to identify the pre-drainage peatland (site) type (level of nutrient regime) several years after drainage, not to mention if decades have passed (Keltikangas *et al.* 1986, Laine 1989, Laiho & Laine 1994).

As if the inherent between-site differences documented above were not enough, one also has to consider the random error introduced by the spatial variation in soil parameters within the site to be sampled, as discussed in the next section.

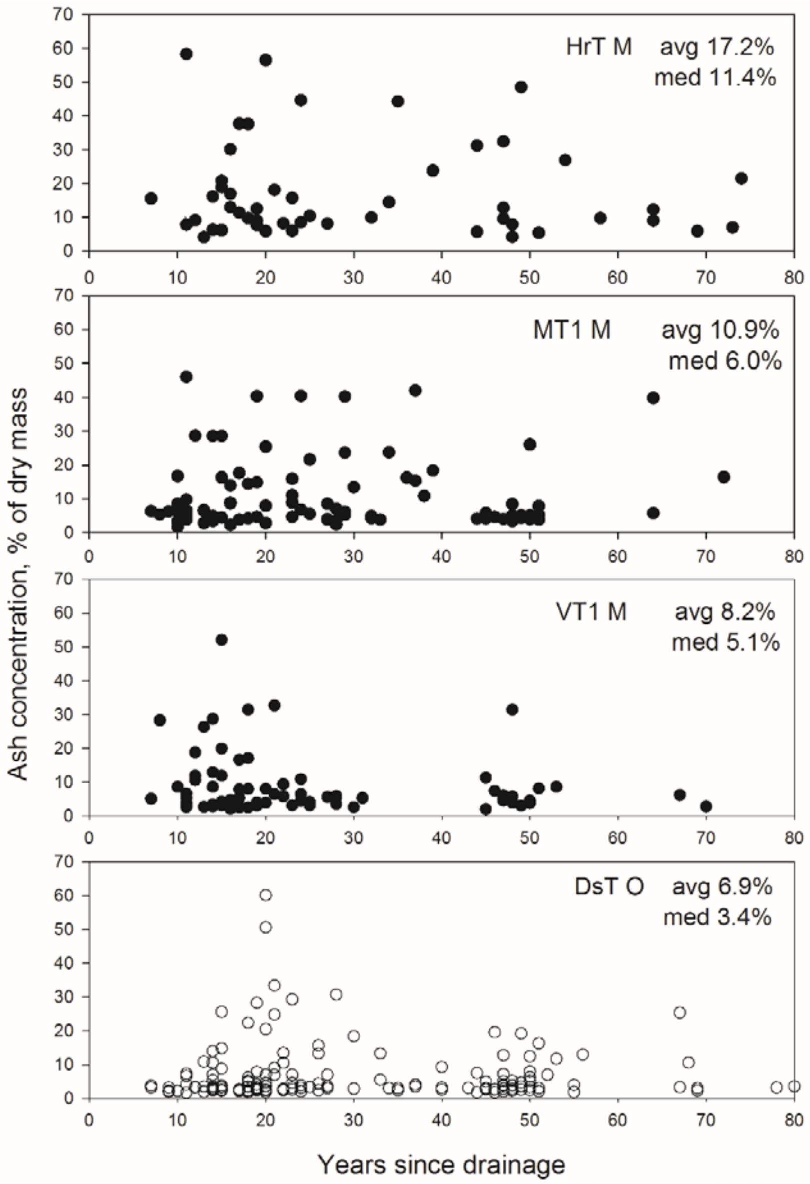

Figure 2. Variation in ash concentration (% of peat dry mass) in a 0–30 cm surface peat layer relative to years passed since drainage within four floristically defined, drained peatland forest site types. Based on the data of Westman & Laiho (2003); ash data previously unpublished. Site types: HrT = herb-rich type (the highest overall nutrient level; see Westman & Laiho 2003) (n = 53), MT = *Vaccinium myrtillus* type (n = 90), VT = *Vaccinium vitis-idaea* type (n = 83), DsT = dwarf-shrub type (the lowest overall nutrient level) (n = 154); M = minerotrophic, O = ombrotrophic. Average (avg) and median (med) concentrations shown. The results shown here for the MT and VT sites concern the originally drier group (1) representing forested sites, which were not covered in the material of Laiho & Laine (1994, 1995); those publications targeted the wetter group (2) sites, which were sparsely treed or treeless before drainage.

2: Spatial variation in peat ash and element concentrations within sites

There is surprisingly little available information on the spatial variation of peat characteristics in general, and element concentrations specifically. Even in studies where several peat samples *per* site have been collected, it has been common to pool the samples before chemical analysis to save costs. The number of samples needed to capture reliable concentration values has, more often than not, been based on guesswork or by limitations set by available resources rather than on information about the extent of variation in the measured parameters.

In their unique study dealing with the spatial variation of element concentrations in peat soils, Laiho *et al.* (2004) examined the within-site variation in two undrained and nine forestry-drained sites belonging to one floristically defined site type, namely tall-sedge pine fen. This is a common minerotrophic fen type in boreal Eurasia, which exhibits a relatively poor (oligotrophic) nutrient regime. The extent of within-site variation depended on the soil attribute examined, being lowest for phosphorus (P) concentration (mg g^{-1} dry mass) and highest for manganese (Mn) concentration. To capture a reliable mean concentration (theoretical maximum deviation of sample mean from true site-level value < 10 %), five subsamples *per* site would be sufficient for P, 10–15 for calcium (Ca) and iron (Fe), and more than 20 for all other elements studied (potassium (K),

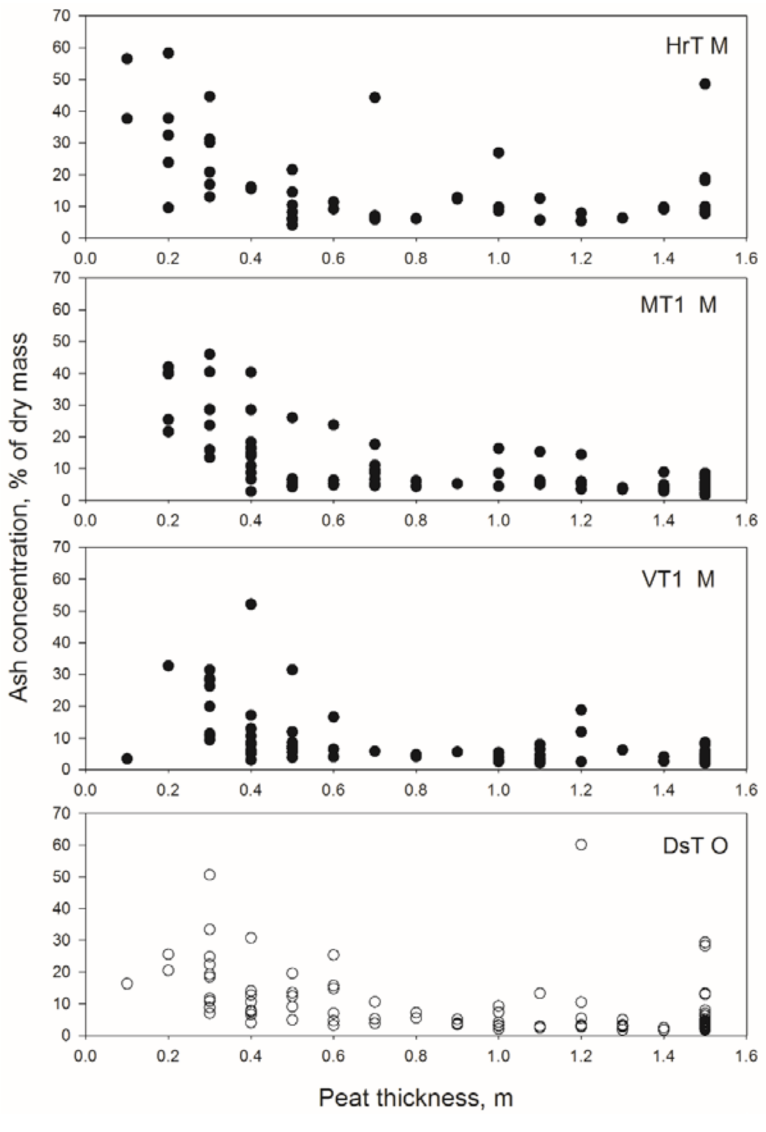

Figure 3. Variation in ash concentration (% of peat dry mass) in a 0–30 cm surface peat layer, relative to thickness of the peat deposit, within four floristically defined peatland site types. The maximum peat thickness measured was 1.5 m; i.e., the 1.5 m thickness point also includes sites with peat thickness > 1.5 m. Based on the data of Westman & Laiho (2003). Site types and sample characteristics as in Figure 2.

magnesium (Mg), Mn, zinc (Zn)), when a composite of 0–30 cm samples was considered. For any 10-cm layer sampled, even higher numbers of subsamples would usually be required. The taking of one to three subsamples per site clearly results in high random error in most of the soil properties measured.

In a follow-up study, Laiho *et al.* (2008) quantified the proportions of variation in peat element concentrations that derived from differences between geographical regions, peatland basins, sites within the peatland basins, and from within-site heterogeneity. In general, most of the variation in the element concentrations was caused by differences among peatland basins and variation within the floristically determined sites (Figure 5). The respective contributions of these factors may vary between sampling depths, depending on the factor examined (Laiho *et al.* 2004).

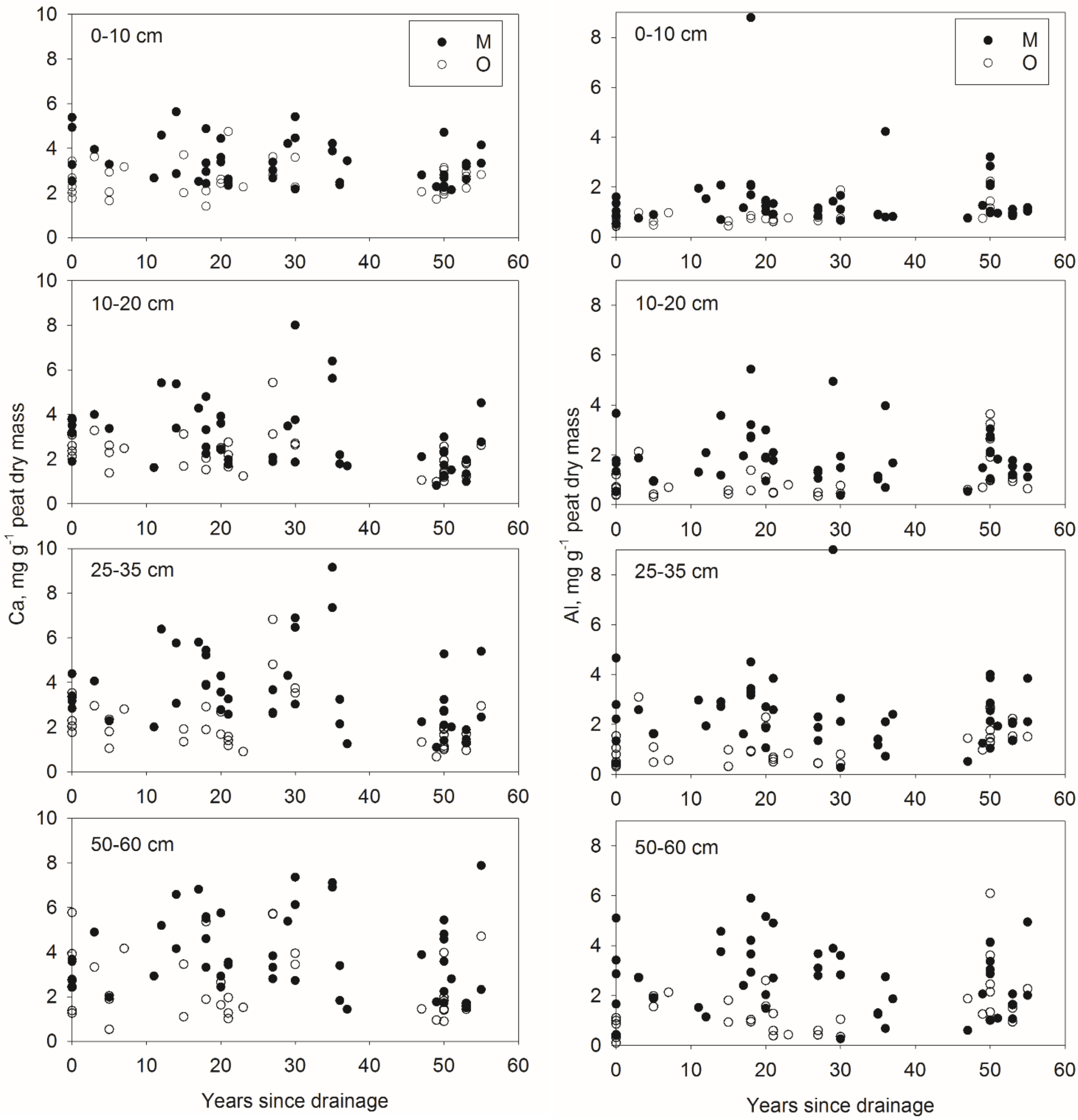

Figure 4. Variation in the concentrations of two example single elements, calcium (Ca, mg g^{-1} peat dry mass) and aluminium (Al, mg g^{-1} peat dry mass), at different depths in two site type groups. Sites as in Figure 1. Data from Laiho & Laine (1995). In the Ca panels, the values from one undrained M site which ranged from 11.7 to 17.1 have been excluded.

Between- and within-site variation in soil characteristics clearly affects the comparability of samples. Another dimension of both between-site and within-site variation is the vertical distribution of soil properties, which we will examine in the next section.

3: Vertical distribution of ash and element concentrations

Do deeper, unaffected layers provide a reasonable ash concentration reference value for a drained site? Extensive vertical distribution descriptions for ash and element concentrations are rather rare, many studies having focused on the surface peat only. However, our understanding of the successional dynamics of peatlands should warn us that the deeper peat layers may in many cases reflect quite different conditions from those at the surface. This is especially true where the surface represents ombrotrophic conditions but the deeper layers are minerotrophic. Not only the inputs of elements, but also the compositions of the plant and microbial communities that processed the material ultimately sequestered as peat, may have differed in such cases. This is likely to be reflected in ash and element concentrations, but not necessarily in directions that we would intuitively expect, since considerable redistribution of elements continuously occurs even in undrained peatlands (e.g., Damman 1986). For instance, Damman (1978) demonstrated variable element concentrations along a four-metre profile of ombrotrophic bog peat. Importantly, he observed that the ash concentration was clearly higher within and above the depth range of water-table fluctuations than deeper in the profile. A somewhat similar pattern, with elevated ash concentrations underneath the permanently oxic surface layer, seems to be common in Central European mires, where it has been explained in terms of mainly anthropic impacts (Sjögren *et al.* 2007). Regardless of whether variations in the depth profile of peat characteristics are autogenous or anthropogenic, it is clear that they need to be known and well-understood before the deeper peat layers can reliably be used as a reference.

However, peat stratigraphic analyses may provide another solution for examining the net impact of disturbance on the peat C balance. In cases where it is possible to locate a truly comparable pair of undisturbed and disturbed sites, e.g., on the opposite sides of a ditch that has split an initially homogeneous peatland, and locate a reference layer of, e.g., charcoal or tephra that is common to the peat profiles of both sites, the net disturbance impact can be gauged by comparing the C content of the whole section of peat above the reference layer at the disturbed site with that at the undisturbed site. This approach has been used by, e.g., Minkkinen *et al.* (1999), Pitkänen *et al.* (2013) and Krüger *et al.*

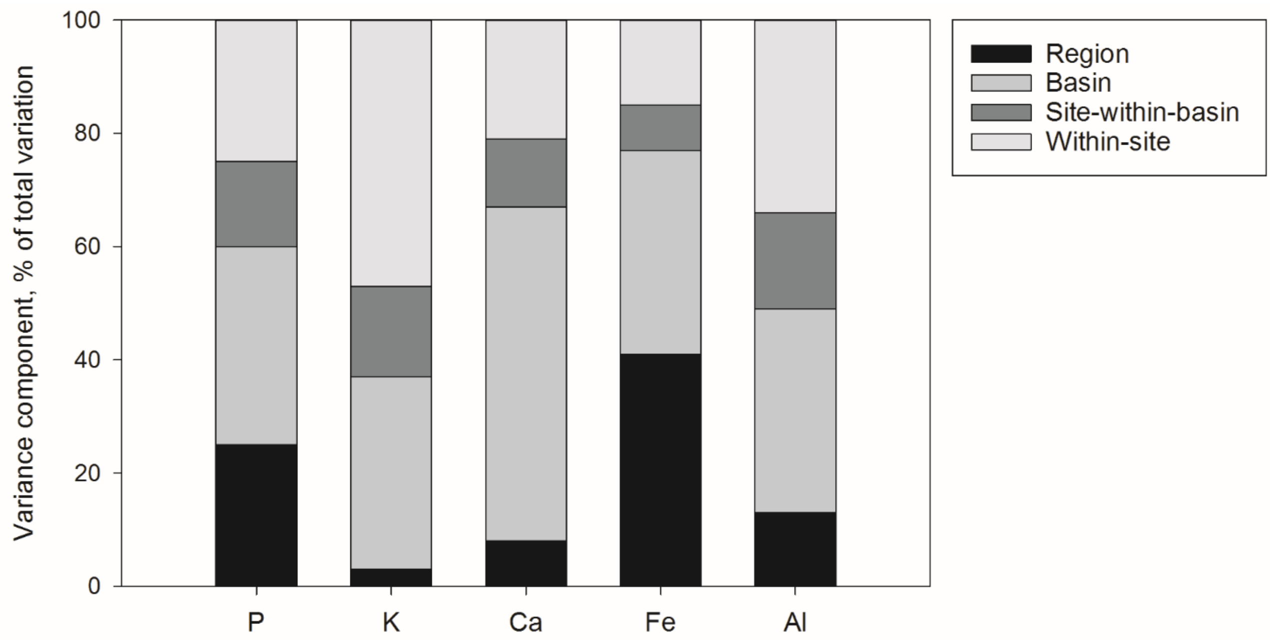

Figure 5. Relative contributions of the four hierarchical levels in a sampling design, namely: region, peatland basin (peatland complex), site within peatland basin, and within-site variation, to the total variation observed in the element concentrations of 878 peat samples collected from 289 sites in 79 peatland basins. From Laiho *et al.* (2008).

(method IV in 2016) and will not be discussed in detail here. One should note, however, that when only the mass of C above the reference layer is measured, the result of the comparison reflects the net impact of disturbance. This does not equal the net loss of C from the peat of the disturbed site in cases where organic matter has continued to accumulate at the surface of the undisturbed site, or at both sites (Straková *et al.* 2010, 2012). To estimate the loss of C from the disturbed site, one should also estimate the difference in C accumulation between the sites after the disturbance, and subtract that from the difference in C content. These methodologies require detailed peat profile analyses and are, consequently, rather laborious. Furthermore, they are applicable only where reliable reference layers can be found.

In cases where reference layers cannot be found, the vertical patterns give rise to another important question that affects application of the ash method. Where is the appropriate zero line when taking soil samples to represent certain depths or depth intervals in undrained peatlands? Is it the top of the *Sphagnum* or other moss shoots? Is it where seemingly dead (Clymo & Duckett 1986) litter peat ends and 'peat proper' begins? What if there is no moss, but only a loose litter layer of sedge leaves? And what is the corresponding layer when sampling a drained site where the surface properties have changed? There may be no single answer to these questions that would represent the absolute truth. However, it is obvious that the questions must be thoroughly evaluated when planning a sampling campaign that includes both undrained and drained sites. Moreover, when a site has been drained for several decades, new organic matter may have accumulated from the litter inputs of the post-drainage vegetation. Straková *et al.* (2010, 2012) showed that, following drainage for forestry, dramatically increased above-ground litter inputs can result in a large accumulation of organic matter despite increased decomposition rates. It should be noted that such accumulations tell us little about the soil C balance, which also depends on the decomposition rate of the pre-drainage peat (Laiho 2006). In some cases it may be relatively easy to identify the surface of the pre-drainage peat layer; however, in many cases it is far from easy, especially where there has been vigorous *Sphagnum* growth and this has decreased only gradually following drainage (Figure 3 in Laiho *et al.* 2003, Laiho *et al.* 2011).

Why does this matter? First of all, when there are clear depth-dependent patterns in the ash or element concentrations, the value captured by any of our samples depends on how we determined the zero line. In the study of Laiho *et al.* (2004), sampling depth accounted for more than 50 % of the total variance for all the studied elements except P and Fe. Sampling depth contributed as much as 90 % of the total variance of Zn and K concentrations. Furthermore, the variance contributions of peatland basin, site, and within-site variation depended on sampling depth. Secondly, when utilising the ash method with reference depths, any major depth-dependent patterns hinder finding a truly comparable reference sample. Finally, when utilising the ash method with reference sites, obtaining peat samples from a drained site that correspond to samples from a certain depth in an undrained site depends on successful zero-line determination. This is a major challenge, since certain physical processes significantly reshape the depth profile following drainage, as discussed in the next section.

4: Physical processes that shape the surface peat following drainage

Peat in pristine sites is mostly water. At saturation, the respective water contents of slightly decomposed *Sphagnum* (bulk density (BD) 0.047 g cm^{-3}) and moderately decomposed *Carex* (BD 0.135 g cm^{-3}) peats are 95 % and 87 % of peat volume, and 2021 % and 644 % of peat dry mass (Päivänen 1973). Peat is also a compressible medium, and changes in the water-table level alter peat volume (Price 2003, Kettridge *et al.* 2013); changes in water storage even affect the position of the peatland surface (Roulet 1991). Changes in peat volume caused by changes in water-table level may be apparent as deep as or even deeper than 70 cm (Kettridge *et al.* 2013). A drop in the water-table level, such as that caused by drainage ditching, increases the weight of the overlying water and peat supported by the soil below. Water is expelled as the pores collapse under this increased weight (Price 2003). Compression of the surface peat and subsidence of the peatland surface following drainage are not reversible as in undrained peatlands, since ditches prevent the build-up of water storage back to the pre-drainage levels. The magnitude of the initial changes caused by purely physical processes depends on the wetness of the site (Lukkala 1949) and the initial structure of the peat (Price *et al.* 2005). The extent of subsidence is generally 20–40 cm, but may be up to 70 cm in boreal peatlands, most of which takes place within five years after ditching (Lukkala 1949). Later, peat compression and subsidence may be enhanced by increased decomposition of the pre-drainage peat, at a rate which depends on peat properties and is generally faster in minerotrophic than ombrotrophic sites (Ojanen *et al.* 2010). Furthermore, compression and subsidence typically vary within the same site, between microforms and depending on the proximity of ditches. Any variation in peat composition, i.e., the dominant plant component, will also influence the degree of compression. Based on the main driving force, a distinction can be made between primary subsidence (caused by purely physical processes) and secondary subsidence (caused by decomposition or other forms of oxidation) (e.g., Ewing & Vepraskas 2006). Primary subsidence leads to increased bulk density in the peat layer affected, while secondary subsidence may or may not have such an effect. However, any mechanism through which peat subsides following water-level drawdown will affect the depth profile of the peat, as outlined below.

Due to compression and subsidence, the pre-drainage depth profile of the surface peat is replaced by one where the layers are in different depth positions than they were before drainage. The post-drainage changes in peat volume extend to relatively large depths and may follow a non-linear as well as a linear pattern relative to depth (Kettridge *et al.* 2013). Consequently, it is not easy to estimate the pre-drainage positions of peat samples taken at any depth from a drained site. This is actually the most critical issue to bear in mind when attempting to evaluate post-drainage dynamics in the soil. For instance, in the theoretical case presented in Figure 6, if one sampled the topmost 20-cm layer at the drained site, the sample would include peat that corresponded to an approximately 42-cm thick layer in the undrained condition. Moreover, if one sampled a one-metre peat layer in both the undrained and the drained situations, the drained sample would include 25 cm of 'extra' peat that would have been below one-metre depth in the undrained situation. To further complicate matters, this would be the case only if there was no surface accumulation of organic matter post drainage. In sites that are permanently vegetated following drainage, like those drained for forestry, fresh organic matter inevitably accumulates (Straková *et al.* 2012), further reshaping the depth positions of the pre-drainage peat layers (right-hand block in Figure 6). Since the extent of subsidence and post-drainage surface accumulation is unknown in most cases, we simply cannot be sure which pre-

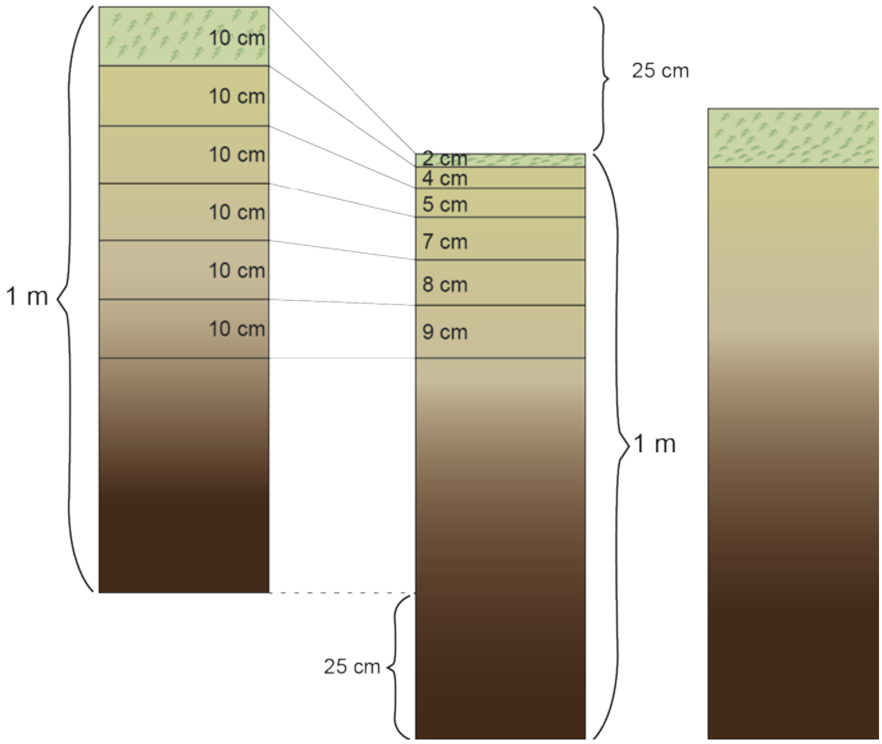

Figure 6. A theoretical example of how peat subsidence following drainage may affect the vertical structure of the peat and the pre-drainage layer sampled. The left-hand block represents the top metre of the peat column in its pre-drainage condition, and the middle block represents the top metre of peat at the same location after drainage. Subsidence is 25 cm and it is assumed that compaction is greatest in the topmost peat and decreases downwards, ending at 60 cm (pre-drainage). Assumed subsidence *per* 10-cm layer is 8 cm 10-cm^{-1} in the topmost layer, then 6, 5, 3, 2 and 1 cm 10-cm^{-1}. In this case, 60 cm of pre-drainage peat would correspond to only 35 cm of post-drainage peat. The right-hand block represents the drained peat column after subsequent accumulation of a litter layer as would occur, for example, if the peatland was drained for forestry.

drainage layer we are actually penetrating when sampling from a specified depth at a drained site, or how much pre-drainage peat we are actually collecting.

This feature is specific to easily compressible peat soils, and it is the main factor that hampers sampling truly corresponding peat layers for undrained-drained comparisons as in, e.g., Kareksela *et al.* (2015). Typically, we are unaware of how subsidence has shaped the depth profile, unless detailed measurements of the extent and depth patterns of subsidence have been made over the period that has elapsed since drainage. The processes have long been recognised; however, their influence on the depth profile have not been thoroughly considered. One should bear in mind when examining the volume changes presented by, e.g., Ewing & Vepraskas (2006), that any such changes in any piece of peat within a profile do not affect just the piece examined, but will alter the depth positions of all pieces of peat above or below the examined piece, as we try to illustrate in Figure 6. Considerations of “equivalent soil mass” (e.g., Ellert & Bettany 1995) that may be applied in mineral soils cannot remedy the situation in the case of peat soils, unless we know exactly how much soil mass we have lost since disturbance; and this we do not know since it is what we are trying to find out. Furthermore, profile comparisons are still hampered by various other biological processes that shape the properties of surface peat following drainage, in addition to decomposition, which should not be assumed to be the only process in action.

5: Biological processes that shape the surface peat following drainage

Different elements fulfil different roles in the structure and function of living organisms. Some are tightly bound into structural tissues and others are concentrated in the vacuolar fluid. During decomposition, are all elements released at the same rate as C? Certainly not (e.g., McGill & Cole 1981). Decomposers target energy-rich and easily decomposable C compounds in order to harvest energy and acquire the C for biomass production. Simultaneously, they require nutrients for both their metabolism and biomass production. These nutrients are taken up either from the decomposing material or, in the case of cord-forming fungi, from other more nutrient-rich sources (Wells & Boddy 1995a, 1995b; Lindahl *et al.* 2001a, 2001b; Fricker *et al.* 2008).

In contrast to the still common paradigm of decomposers ‘releasing’ nutrients, decomposers in peat may be highly selective in retaining most of the limiting nutrients present in the decomposing material and transferring nutrients from one location to another (Boddy & Watkinson 1995, Boddy 1999, Lindahl *et al.* 2002, Boberg *et al.* 2014). The elements actually released are largely those present in easily decomposable compounds that exceed the needs of the decomposers. Boberg *et al.* (2014) recently demonstrated that saprophytic fungi, which are the primary decomposers in boreal forest soils, can increase their C utilisation efficiency (i.e., allocation of assimilated C to mycelial biomass rather than respiration) through the redistribution of nitrogen (N) from older to fresher material. In doing so, fungal biomass is increased without increasing the rate of litter decomposition. Their preference for fresh food over (still edible) leftovers may actually contribute to organic matter accumulation, as N is translocated from older more-decomposed material to newer high-quality C sources, consequently reducing decomposition of the old material (Lindahl *et al.* 2002, Boberg *et al.* 2014).

Soil chemistry is also modified by the growing tree stand, through several mechanisms that are not connected to decomposition. Mycorrhizal roots are able to harvest N and P directly from the soil without corresponding losses of C. Uptake of soluble nutrients such as potassium by the tree stand may reduce soil nutrient concentrations. Expansion of the forest canopy, particularly in typical peatland forests with unevenly spaced trees of mixed heights (Sarkkola *et al.* 2005), increases the influx of elements to the site from dry deposition (Schauffler *et al.* 1996). Tree roots introduce new organic matter, which may be chemically different from the older peat material, directly into the soil. Furthermore, ash and element concentrations in plants vary according to species, organ and soil composition, as well as the amount of water transpired (Goss 1973, Larcher 2003); thus, changes in vegetation induced by drainage can alter the peat ash content. Other mechanisms for reducing soil element concentrations without decomposition and C loss also exist. For instance, the increase in soil acidity observed following drainage results in increased leaching of base cations that are replaced in the cation exchange sites by protons (Laine *et al.* 1995b, 2004).

Overall, the consequences of these processes may be smaller than for processes outlined previously; however, thus far we have few repeated measurements to verify this supposition. It should also be noted that an unquantified fraction of the elements measured in ‘peat’ are actually located in fine roots and other soil biota. In other words, so many different processes drive *in situ* soil element concentrations in multiple directions simultaneously that their interpretation in terms of decomposition is highly challenging.

DEMONSTRATION

Misapplication of the ash method produces seemingly realistic but misleading results

The material of Laiho & Laine (1994, 1995) was designed and carefully selected to be applied as a time–space substitution for an examination of the effects of drainage on peat soils, paying special attention to the comparability of sites, i.e., their similarity in undrained conditions (see also Laine *et al.* 1995a). As mentioned earlier, C concentrations were also determined for this material, thereby allowing us to demonstrate what the ash method in its most simplified form (Kareksela *et al.* 2015) would indicate in terms of the C balances for our sites. Indeed, our material seems ideal for such an evaluation. One should note that the C concentration of peat, which is generally assumed to be 50 % of dry mass and not measured, actually varies as well (see also Minkkinen & Laine 1998).

Let us examine, first, the basic assumption that the ash concentration in surface peat increases following drainage. The analysis of ash data shown in Figure 1 indicated that the main effect of time since drainage was not significant; however, there were significant interactions between time, depth and site group (Appendix 1). Further examination revealed that ash concentration actually decreased with time since drainage in the minerotrophic site group, whereas it increased in the topmost layers of the ombrotrophic and transient group (Figures A1, A2 in Appendix).

Next, let us calculate the ash and C contents (kg m^{-2}) for the 0–20 cm layer, where the post-drainage changes are assumed to be clearest. In the minerotrophic site group, the ash contents of practically all drained sites were within the same range as those of the undrained sites (Figure 7). In the ombrotrophic and transient group, the variation was higher in drained than in undrained sites, and ash contents were generally higher in the older drained

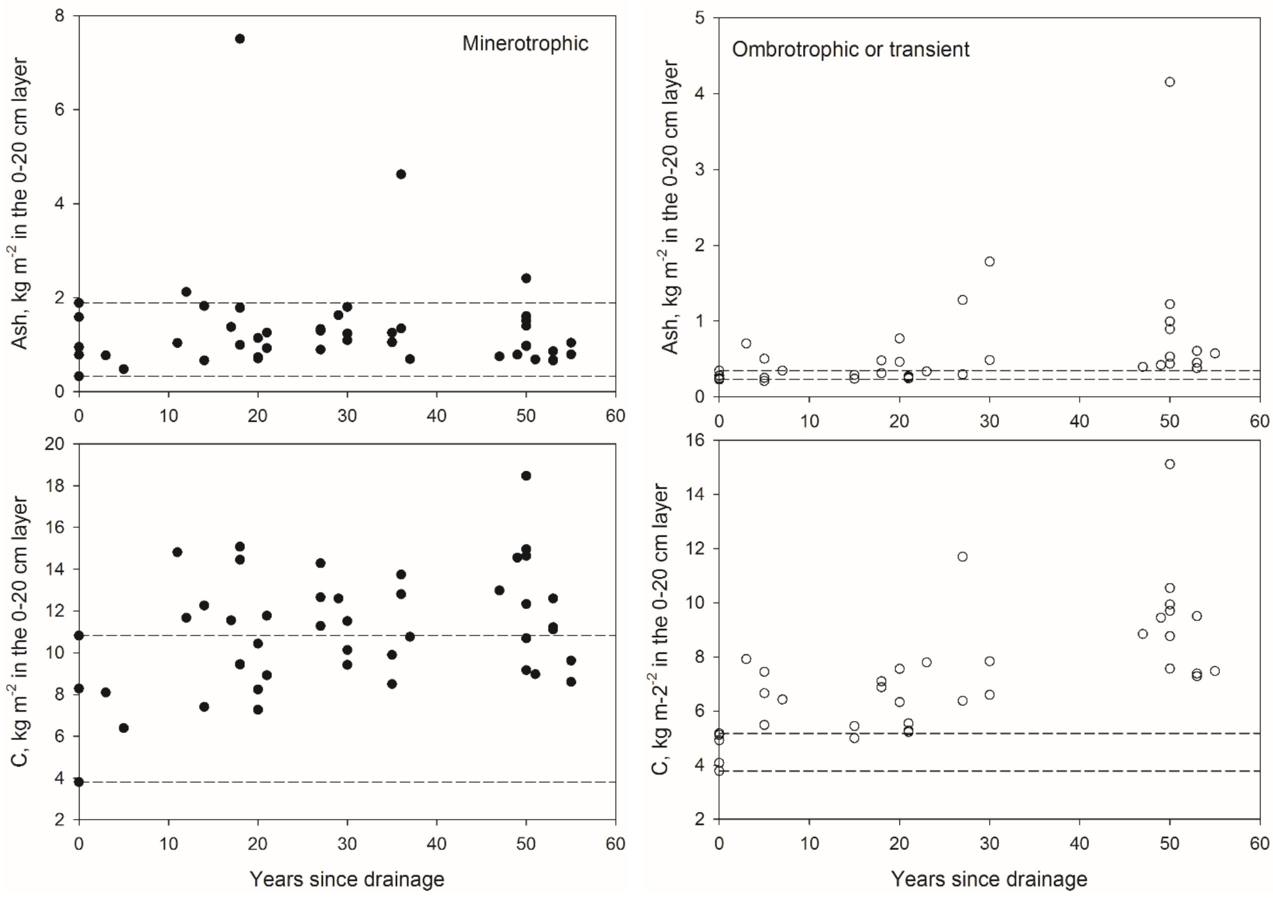

Figure 7. Ash (upper panels) and carbon (C) (lower panels) contents (kg m^{-2}) in the topmost 0–20 cm peat layer, in ‘minerotrophic’ (left) and ‘ombrotrophic and transient’ (right) sites sampled by Laiho & Laine (1994, 1995). Sites as in Figure 1. Note that the y-axis scales differ between the site type groups. Dashed lines indicate the range of values in the undrained sites.

sites. Quite strikingly, however, C content clearly increased following drainage. We attribute the increase largely to subsidence and compaction of the surface peat, which is evidenced by clear changes in the bulk density profiles following drainage (Laiho *et al.* 1999). In older drained sites in particular, all of the other processes described above were also active.

Finally, we calculate the 'C loss' for each site based on the C and ash contents measured for each drained site, using the mean C/ash measured for the undrained sites as a proxy for pre-drainage conditions as in Kareksela *et al.* (2015), according to the equations:

$$C_{\mathrm{e}} = \frac{C_{\mathrm{p}}}{ash_{\mathrm{p}}} \times ash_{\mathrm{d}} \qquad [1]$$

$$\Delta C = C_{\mathrm{e}} - C_{\mathrm{d}} \qquad [2]$$

where C_{e} is the expected C content (kg m^{-2}) of the 0–20 cm peat layer, $C_{\mathrm{p}}/ash_{\mathrm{p}}$ is C/ash pre-drainage, ash_{d} is the ash content measured post-drainage, ΔC is the change (loss or gain) in C content and C_{d} is the C content measured post-drainage.

These calculations mostly indicate C gains for the minerotrophic sites, and mostly C losses for the ombrotrophic and transient sites (Figure 8). However, this demonstration has not actually identified any real patterns of advancing decomposition and C loss; merely the net effect of inherent variation and the different processes that shape the surface peat. In this case, the values also appear dubious, since:

1) based on Ojanen *et al.* (2013, 2014) we know that a net loss of C must currently be taking place in many of the minerotrophic sites, while net gain may be experienced by at least some of the ombrotrophic and transient sites; and
2) the minerotrophic sites in particular are likely to have experienced a net loss of soil C during the first years following drainage (Laiho 2006) when litter inputs from the disturbed vegetation were relatively low (Straková *et al.* 2010, 2012), and the 'decomposition potential' of the newly oxic peat layers was at its highest (Jaatinen *et al.* 2008).

We wish to emphasise, however, that the results presented above do not and cannot disprove the concept of increasing ash concentration in a single block of peat with advancing decomposition. Rather, we suggest that other processes that were dealt with in the previous sections thoroughly mask such a process in field samples, especially in the more nutrient-rich sites.

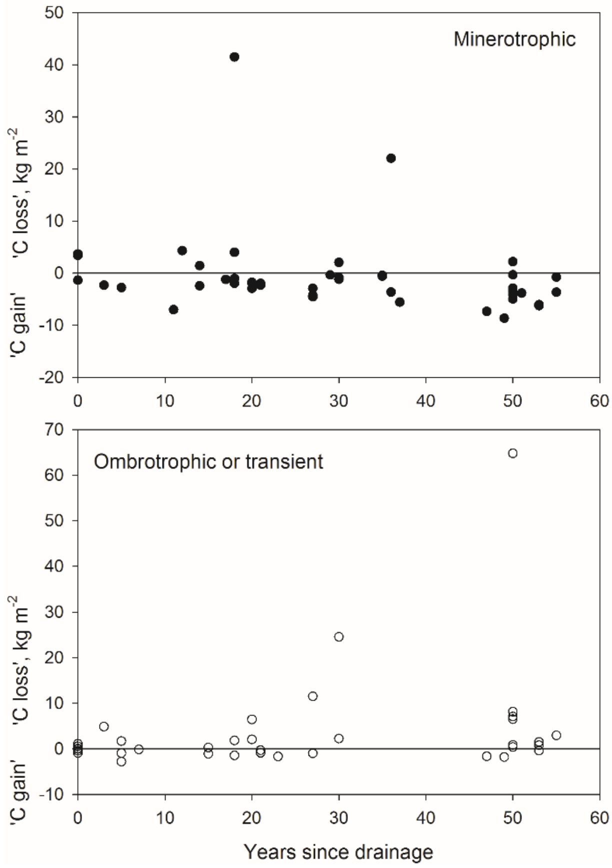

Figure 8. Carbon (C) loss or gain (kg m^{-2}) in the 0–20 cm peat layer estimated by subtracting the measured C content from the expected C content, estimated for each site from its peat ash content and the average value of C/ash for the undrained sites (Equations 1 and 2), as is done when applying the ash method using reference sites (e.g., Kareksela *et al.* 2015). Sites as in Figure 1.

CONCLUSIONS

Peat soils are incredibly dynamic. They consist almost entirely of organic matter. Decomposition may be relatively fast in the oxic surface layers (Straková *et al.* 2012), but vegetation cover guarantees continuous litter inputs; we could say that peat soils change almost from day to day. It is critical to keep this in mind when making any comparative studies or indeed any studies at all. The C/ash quotient may well be useful, in principle, for the estimation of soil C losses with repeated measurements in simple systems where the inputs and outputs of elements present in ash are known or can be reliably estimated. Such simple systems are extremely rare, however. The closest may be unvegetated cutover sites without major groundwater inputs. In complex systems, it is highly challenging to find a truly comparative reference sample when attempting to apply the ash method with either undisturbed reference sites or reference layers, due to the multitude of processes that shape the surface peat following drainage. If it is applied on the basis of single measurements, especially in initially minerotrophic peatlands, the presence of numerous sources of independent non-random error will most likely spawn biased outcomes. Since the range of outcomes is in any case dictated by the magnitude of the ash and C concentrations, the results may seem realistic; especially if 'outliers' are eliminated. This makes careless application of the ash method especially misleading. Unfortunately, however, there is no simple method for estimating the C balance of peat soils in most cases.

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Appendix

Analysis of the impacts of site group (typegrp1 = minerotrophic sites, typegrp2 = ombrotrophic and transient sites), sampling depth (Layer 1: 0–10 cm, Layer 2: 10–20 cm, Layer 3: 25–35 cm, Layer 4: 50–60 cm), and time passed since drainage (age; zero = undrained) on the ash concentration (% of peat dry mass) in the material of Laiho & Laine (1994, 1995).

A mixed linear model was fitted to log-transformed ash concentrations (Figures A1, A2) including all interactions of age, layer, and type group as explanatory variables and plot as random effect. In other words, separate linear regressions of log-ash *versus* age were fitted to each combination of layer and type group, but with common plot effects and residual variance. Layer 4 and type group 2 were the baselines, so that the intercept and slope (parameter value for AGE) determine the regression for that combination and the values for other levels are the differences of the corresponding regression parameter estimates from the baseline (Table A1).

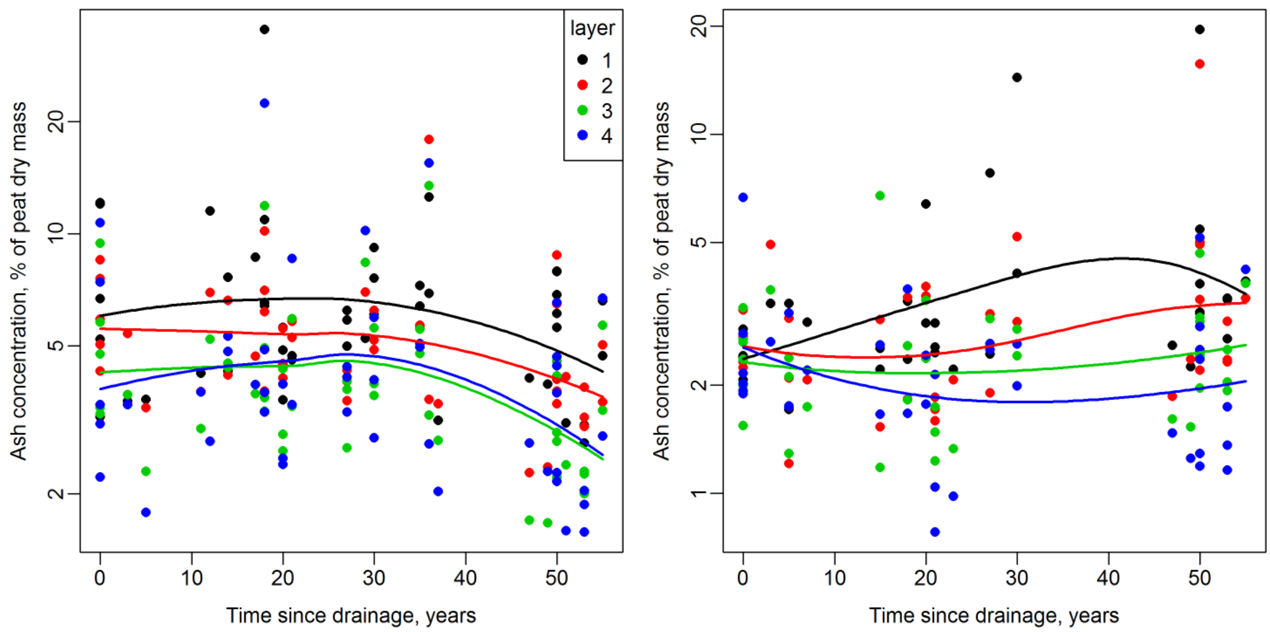

Figure A1. Ash concentrations (% of peat dry mass) as a function of time since drainage (years), in type groups 1 (left) and 2 (right); note the logarithmic scale of the y-axis. The lines show local polynomial regressions (Cleveland *et al.* 1992) fitted separately to the log-transformed ash concentrations of each layer. Fitting was carried out in the R environment (R Core Team 2015) using the loess function with smoothing parameter span = 1.2.

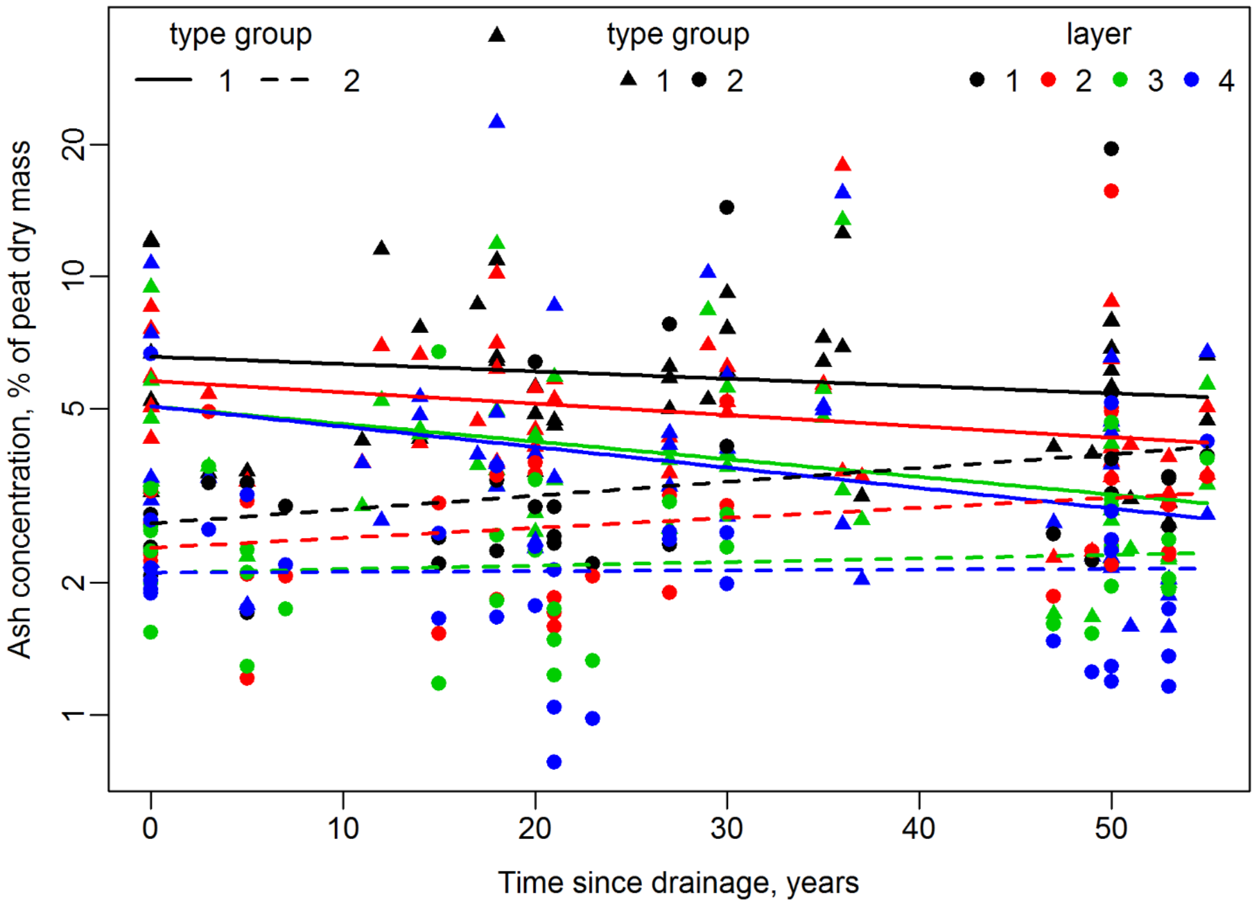

Figure A2. Fixed effects of the mixed linear model fitted into the log-transformed ash concentrations (lines). Triangles and dots show the original observations as in Figure A1.

Table A1. The model parameters.

Value	Std.Error	DF	t-value		p-value
(Intercept)	0.8068	0.1219	240	6.617	0.000
LAYER1	0.1340	0.0981	240	1.366	0.173
LAYER2	0.0391	0.0981	240	0.399	0.690
LAYER3	-0.0287	0.0981	240	-0.293	0.770
TYPEGRP1	0.7445	0.1752	80	4.250	0.000
AGE	-0.0036	0.0038	80	-0.947	0.346
LAYER1:TYPEGRP1	0.2662	0.1409	240	1.889	0.060
LAYER2:TYPEGRP1	0.1952	0.1409	240	1.385	0.167
LAYER3:TYPEGRP1	0.0654	0.1409	240	0.464	0.643
LAYER1:AGE	0.0135	0.0031	240	4.424	0.000
LAYER2:AGE	0.0101	0.0031	240	3.294	0.001
LAYER3:AGE	0.0057	0.0031	240	1.852	0.065
TYPEGRP1:AGE	-0.0036	0.0053	80	-0.673	0.503
LAYER1:TYPEGRP1:AGE	-0.0127	0.0043	240	-2.965	0.003
LAYER2:TYPEGRP1:AGE	-0.0099	0.0043	240	-2.321	0.021
LAYER3:TYPEGRP1:AGE	-0.0076	0.0043	240	-1.778	0.077