The distribution and (future) use of Switzerland's organic soils

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SUMMARY

Drained organic soils have high rates of greenhouse gas (GHG) emissions, a problem that can only be mitigated by re-wetting. In agricultural regions, the suitability of re-wetting as a mitigation option depends on the extent and use of organic soils, economic conditions, and the availability of other land to ensure continued agricultural production. Here we analyse the conditions for GHG savings from farmed organic soils for Switzerland, a densely populated country where scarce agricultural land is protected by regional quota systems and where there is a minimum self-sufficiency target for food production. We present a new estimate of the extent of organic soils (32,702 ha). Agriculture dominates the use (61 %) and GHG emissions (89 %) of these soils, and their GHG emissions amount to 25 % of the national 2050 emission reduction goal for agriculture. At national level, only 1.2 % of agriculture (by area) takes place on organic soils, suggesting that losses in agricultural production associated with re-wetting of these soils may be manageable. In some regions, however, organic soils are very important for agriculture, and alternative means to reduce GHG emissions need to be sought for these regions. We explore future paths for such regions, considering economic viability and the availability of land.

KEY WORDS: greenhouse gas emissions, mapping, paludiculture, peatlands

INTRODUCTION

In many European countries, peat-forming ecosystems (mires) have undergone a history of drainage over recent centuries and it has been estimated that > 50 % of mires in Europe are degraded (Joosten & Clarke 2002, UNEP 2022). The drainage of mires is often associated with economic benefits through the gain of land for food, timber or biomass (for energy) production. Since the second half of the 20th century, however, the recognition that near-natural peatlands are undervalued for the variety of ecosystem services they provide has increased (Joosten & Clarke 2002, Verhoeven & Setter 2010, Wijedasa *et al.* 2016). These ecosystem services include habitats for often rare animals and plants adapted to wet environments (Parish *et al.* 2008), water retention, buffering and purification (Joosten & Clarke 2002, Joosten *et al*. 2017), recreation, and carbon (C) sink and storage functions (Byrne *et al.* 2004, Verhoeven & Setter 2010).

Drained peatlands are also associated with a number of negative effects including heightened risk of peat fires, damage to infrastructure (due to instability following soil subsidence), high costs of maintaining the drainage systems required to manage these soils for conventional commodity production, high rates of dissolved organic C, soil subsidence eventually impairing commodity production and thus food security, and large greenhouse gas (GHG) emissions resulting from the rapid oxidative decomposition of drained peat (Liu *et al*. 2019, Page *et al.* 2020, Freeman *et al.* 2022). This last issue has recently received heightened attention from the scientific community and authorities. At the global scale, drained peatlands are not only a significant cause of climate change (UNEP 2022) and from the policy perspective a hurdle in reaching climate goals (Leifeld *et al.* 2019a, Günther *et al.* 2020), but could also be part of the solution through mitigation of their GHG emissions by re-wetting (Leifeld & Menichetti 2018, Strack *et al.* 2022) and their capacity for adaptation to future climates (UNEP 2022). Further benefits include reversing soil degradation and, especially, restoring biodiversity (Tanneberger *et al.* 2021). Knowing the extent of peatland soils and how they are managed is a prerequisite for quantifying their emissions and estimating their potential for climate change mitigation.

The combination of negative effects and opportunity costs resulting from lost ecosystem services means that drained peatlands can pose a high cost to societies (Verhoeven & Setter 2010, Bonn *et al.* 2014). In recognition of this, national strategies dealing with the sustainable management of peatlands are being launched (e.g., within Europe, Nordbeck & Hogl 2023). Raising the water level in drained organic soils is, with a few exceptions (e.g., Ojanen *et al.* 2013), paramount to reducing GHG emissions (Günther *et al.* 2020, Freeman *et al.* 2022). For sites where restoration to mire habitat is not feasible, paludiculture - the production of commodity biomass including food - whilst maintaining high water levels that halt soil subsidence (Tanneberger *et al.* 2022), is becoming a recognised alternative. Co-benefits of paludiculture include the provision of (semi-)wetland habitats and reduced GHG emissions (Verhoeven & Setter 2010, Tanneberger *et al.* 2022) even if C losses through harvests are accounted for (e.g., Günther *et al.* 2015, Daun *et al.* 2023). However, there are also disadvantages. In the agricultural context these include the potential loss of income previously generated by a site, along with a reduction in food production. The latter occurs if crops for non-food products (e.g., building materials or bioenergy) are grown, or even if the land is used to produce feed or food because many uses are often associated with reduced yields (Freeman *et al.* 2022). The extent to which these disadvantages are a problem for a region depends to a large extent on the context (Girkin *et al.* 2023, Willenbockel 2024). This includes how the land is currently used for production, the financial gains of the current land management compared to other options including paludiculture, the importance of organic soils for agricultural production in the region, and how much other land is available for agricultural production. These considerations also strongly influence the social acceptability of changes in land management. In this article we explore these issues for Switzerland, a country with a high proportion of drained peatlands where agricultural land is scarce.

Peatlands in Switzerland have been exploited since the beginning of the 18th century, and most of them are managed for commodity production (Wüst-Galley *et al.* 2019). Although specific measures are lacking, the maintenance of C in organic soils forms part of the national soil strategy (FOEN 2020) and the agricultural climate strategy (FOAG *et al.* 2023). Determining the policy strategies and measures required to reduce C losses from these soils depends on knowing where they are located and how they are managed. In Switzerland, the best land for arable cropping - the 'crop rotation area' - is protected by law (ARE 2023). More specifically, a minimum quota of this surface must be maintained (438,460 ha, or 30 % of the total agricultural land area). This quota is currently exceded by merely 7,220 ha (or 1.6 %; ARE 2023). Additionally, the country aims to maintain a self-sufficiency rate of 50 % for food production (FOAG *et al.* 2023).

Organic soils in this study encompass the soils of mires, as well as C-rich soils that remain in the landscape following peatland drainage (mostly Histosols). Switzerland has no national soil map, and the most recent estimate of its organic soil area used a 'bottom-up' approach involving the collation of spatial data sets indicative of organic soils (Wüst-Galley *et al.* 2015). Since the data sets used in that study were acquired (2012–2013), additional relevant data sets have become available, meaning an updated estimate is now possible and pertinent. In this study, a new estimate of Switzerland's organic soil area is generated (step 1); then this is used to assess the importance of these soils for national GHG emission reduction targets (step 2) and for agricultural production at regional and national levels (step 3). Lastly, information on the current management and marginal gains (profitability) of crops is used to assess the competitiveness of paludiculture under current economic conditions (step 4). Our aim is to provide contextual information on the role of organic soils in Switzerland's agricultural landscape, which is basic information for policy-makers.

METHODS

Organic soil area

The general approach used was to collate spatial data sets indicative of organic soils, including updates of four important sources (Figure 1). These data sets were harmonised, combined in a GIS, and a rule-based model was applied to assess the certainty with which the resulting surfaces can be taken to represent organic soils today. Thus, this study represents an update of Wüst-Galley *et al.* (2015).

The study aimed to identify organic soils according to the IPCC (2006) definition. In short, these are soils that formed under waterlogged conditions, with a C-rich layer more than 20 cm thick (or less if C-content >12 % when mixed to 20 cm). The C-rich layer must have a soil C content of >12 % (if soil has no clay) to 18 % (if clay content > 60 %). For some data sets, however, the only relevant information available was the presence of peat. It is possible that some of these surfaces had only thin peat layers and such surfaces were therefore scored as organic soil, but with lower certainty (see next sub-section).

Harmonising data sets

Data sets were harmonised in terms of three aspects. First, individual surfaces were scored according to whether they represent organic soil - at the time of survey - unambiguously [= 'A' with higher certainty; = 'B' with lower certainty] or ambiguously [= 'C']. The scores are indicated in square brackets in the section 'Data sets'. Secondly, data sets were scored as 'historical' or 'modern', with a cut-off date of 1945. Although this date is 80 years ago, it corresponds to the end of one of the major periods of soil drainage and melioration in Switzerland (Maurer 1985, Mühlethaler 1995). We assumed that mires that had not been destroyed before this point in time are likely to have survived as either wetlands or drained peatlands. Additionally, many data sets were collated in the 1950s to 1970s so that using a later cut-off date would probably have led to underestimation of the area of organic soil by a greater amount than the (possible) overestimation caused by using this early cut-off date. Thirdly, the spatial quality of data sets was scored (Table S8 in the Supplement). This incorporated spatial precision, any problems encountered during geo-referencing of data sets, and cases where the outline of peatland was imprecise on the original map.

Data sets

Four data sets were updated for this study (Figure 1). Other information sources from Wüst-Galley *et al.* (2015) that were not updated were also used; these are listed in Tables S4-S7.

The nomenclature of most **soil maps** used in this study follows the national soil type classification system (BGS 2010 and earlier versions), where two soil types, "Halbmoor" and "Moor", are considered to be organic soils [A]. These soils have a peat horizon (> 30 % organic matter) greater than 40 cm thick in the upper 80 cm of soil, with ("Halbmoor") or without ("Moor") mineral sediment layers in between. Soil complexes indicating "organic soil" were also considered to be organic soils [B]. In other soil maps, surfaces classified explicitly as "organic soils" were considered as such [A]. Relevant surfaces from 'indicator maps' were classified as organic soils [B]. Lastly, one map indicated C and clay content; surfaces were considered to be organic soil [A] where the definition of organic soils was met. The soil maps used in this study are listed in Table S1. Surfaces from soil maps represent ~16,300 ha of organic soil. Surfaces with soil types other than those listed above, and other than gleys, were collated separately to form a 'mineral soil' surface.

The **national inventory of raised and transitional bogs** (Grünig *et al.* 1986, with updates in 2003, 2007 and 2017; FOEN 2017) contains all identified bogs > 625 m^2. Surfaces with primary or secondary bog vegetation, as well as peat surfaces denuded of vegetation were considered to be organic soil [A]. Surfaces from this inventory represent ~1,600 ha of organic soil.

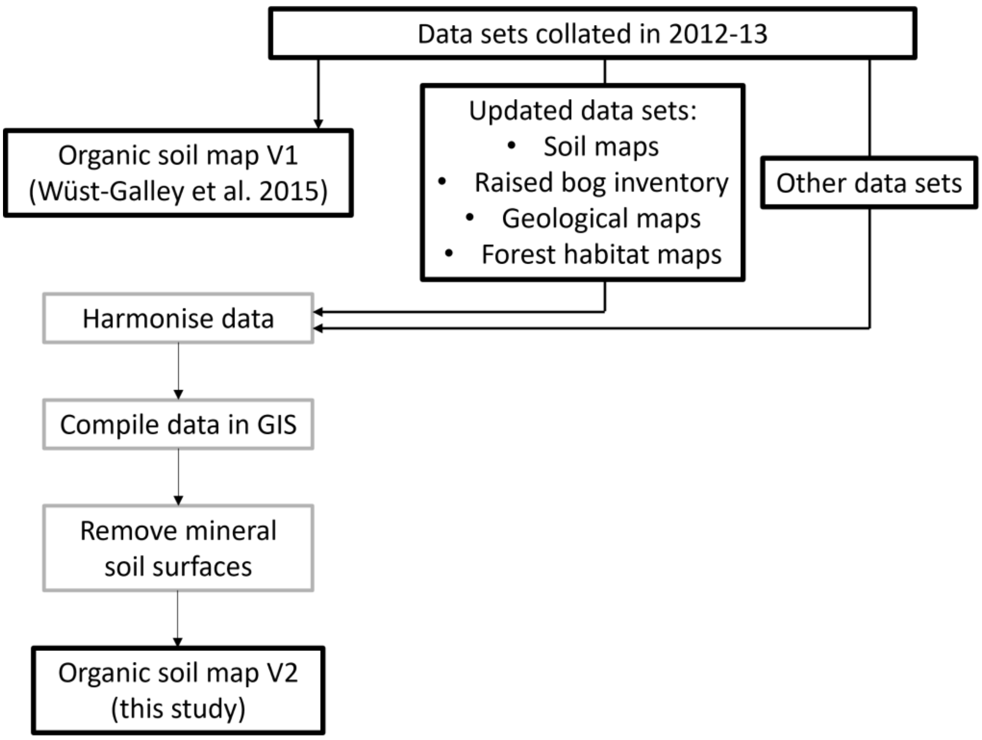

Figure 1. Outline of the methodology for generating the organic soil map. Black boxes denote data sets or products, grey boxes denote processes. Processes and updated data sets are described in the main text; 'other data sets' (i.e., those not updated) are described in the Supplement (separate download).

The GeoCover vector data, published by the Federal Office of Topology (Swisstopo), contains **geological maps** covering the whole country. Map sheets were surveyed by different geologists across a long time period (with a few even older exceptions, circa 110 years, Table S2). Comparison of the GeoCover data with soil maps (which are generally large-scale maps with a precise classification of soils) suggested that not all surfaces indicated by the former are indeed organic soils. Surfaces described as peatlands were therefore considered organic soil but with lower certainty [B]; surfaces described ambiguously as organic soil, e.g., "marsh or peatland" were scored as potentially containing organic soil [C]. Surfaces from GeoCover represent ~16,900 ha of organic soil.

Cantonal **forest habitat maps** describe the near-natural forest habitat type which would occur at a site. They are derived using information including bedrock, hydrology, relief and ground flora. Surveys from most cantons follow the classification of Ellenberg & Klötzli (1972; hereafter 'E&K'), or used a classification system that could be 'translated' to the E&K system. Following ground-truthing (Wüst-Galley *et al.* 2015, Wüst-Galley *et al.* 2016; see also Supplementary Information), three E&K habitat types were considered to be associated with organic soil: [A] (*Pino-Betuletum pubescentis*, German: "Föhren-Birkenbruchwald"; *Sphagno-Piceetum*, German: "Moorrand-Fichtenwald"; *Sphagno-Pinetum montanae*, German: "Torfmoos-Bergföhrenwald"); habitat complexes where the main habitat type was one of the three listed above were also considered to be organic soil [A]. Relevant surfaces from the cantons Grisons and Neuchatel, that both use modelled prediction maps, were considered organic soil but with less certainty [B]. Maps used are listed in Table S3. In total, surfaces from the forest habitat maps represent ~ 2,500 ha of organic soil.

Compiling data sets

The harmonised data sets indicative of organic soil were overlaid in a GIS. The information on the certainty with which the data set represents organic soil and indicative age of the data set (see section 'Harmonising data sets') was combined for each resulting polygon. A rule-based model (Table S9) was used to classify polygons into one of four classes: 1) conservative estimate of organic soils; 2) less conservative estimate of organic soils; 3) historical evidence (only) of organic soils; and 4) potentially organic soil with ambiguous evidence only.

In a last step, data sets indicative of mineral soil were used to remove polygons from the above-generated map, as follows. Where the evidence for organic soil was older than that of mineral soil or where the spatial quality of mineral soil data sets was superior to that of organic soil data sets (or the same), the surface was re-classified as mineral soil (1,724 ha) i.e., removed; where the evidence for organic soil was more recent than that of mineral soil and the spatial quality of the data better (or the same), these surfaces were deemed to have truly conflicting data and were also removed from the map (510 ha); where the spatial quality of evidence for organic soil was better than that for mineral soil *and* the evidence of mineral soil was not more recent than that of organic soil, surfaces were deemed to be organic soil (299 ha). The less conservative estimate of organic soils, termed henceforth the 'new estimate of organic soils', was used for all further calculations.

Spatial data for analyses

Spatial information on land use was obtained from three sources: i. The national land use statistics from 2013–2018 (FSO 2019/2020, 100 m × 100 m resolution) using the 'NOAS04' categorisation were used in the analysis for the entire country. Arable land includes field crops, greenhouse cultivation, temporary grasslands (leys) and vegetable production. Agricultural grasslands include lowland (managed year-round) and alpine (summer pastures and meadows) grassland. 'Bushy' and 'stony' grassland were interpreted as 'unproductive' vegetation. ii. The national statistics describing the use of registered agricultural land (KGK 2024) were used to obtain detailed use of farmed organic soils, with definitions of grassland management intensity following Richner *et al.* (2017). iii. Information on the 'crop rotational surface' was obtained from the national dataset "Fruchtfolgefläche" (ARE 2023).

The potential for different crops or grassland types to be managed as paludiculture was evaluated according to Birr *et al.* (2021). In that publication, candidate crops and grassland management (considering grazers as well as vegetation) that are productive when water tables are high enough to reduce or halt peat decomposition were selected. We supplemented this with information on livestock density from Richner *et al.* (2017), to define the intensity of the grassland management regimes - as defined in Switzerland - compatible with a raised water table.

Greenhouse gas emissions

Emission factors from the national inventory document (FOEN 2024), were applied to land use categories from the national land use statistics to calculate GHG emissions (CO_2 and N_2O) from organic soils (Table S10). Emissions of CH_4 from ditches are not included due to a lack of knowledge regarding their coverage. In accordance with FOEN (2024), lowland agricultural grassland was assumed to be deeply-drained nutrient-rich, alpine agricultural grassland to be shallow-drained nutrient-rich, and unproductive vegetation was assumed to be shallow-drained, nutrient status unknown. Although the last of these categories includes many fens and raised bogs, biotope monitoring (Bergamini *et al.* 2019) shows that these biotopes are on average becoming drier and degrading over time, supporting the allocation of this emission factor. As a comparison, emissions were also calculated using emission factors from IPCC (2014).

Data on contribution margins

The contribution margin of a product is its selling price per unit (here, per hectare) minus production costs. Data were obtained from the national 'contribution margin' catalog (Dietiker *et al.* 2021) covering field crops and grasslands. The values represent agriculture on both mineral and organic soils; we used the contribution margins including variable costs (seed, agrichemicals, hail insurance, processing of harvest e.g., drying, and machinery and labour costs) and excluding subsides. Costs associated with drainage or flood protection are not included, nor are externalised societal costs. Wet agriculture is rare in Switzerland and, with the exception of grasslands, no data were found on contribution margins. Therefore, information from Switzerland (paddy rice) and Germany (paludicultures) was taken from Fabian *et al.* (2024).

RESULTS

The new estimates of Switzerland's total area of organic soil surface (Figure 2) are 18,477 ha (more conservative estimate) and 32,702 ha (less conservative estimate). The latter is used for all further calculations. This new estimate is approximately 18 % higher than the corresponding estimate of 27,813 ha from Wüst-Galley *et al.* (2015). The net increase (4,889 ha) excludes 4,699 ha that are no longer in the estimate, and includes 9,588 ha that were newly recognised. Surfaces for which there is only historical documentation of organic soils

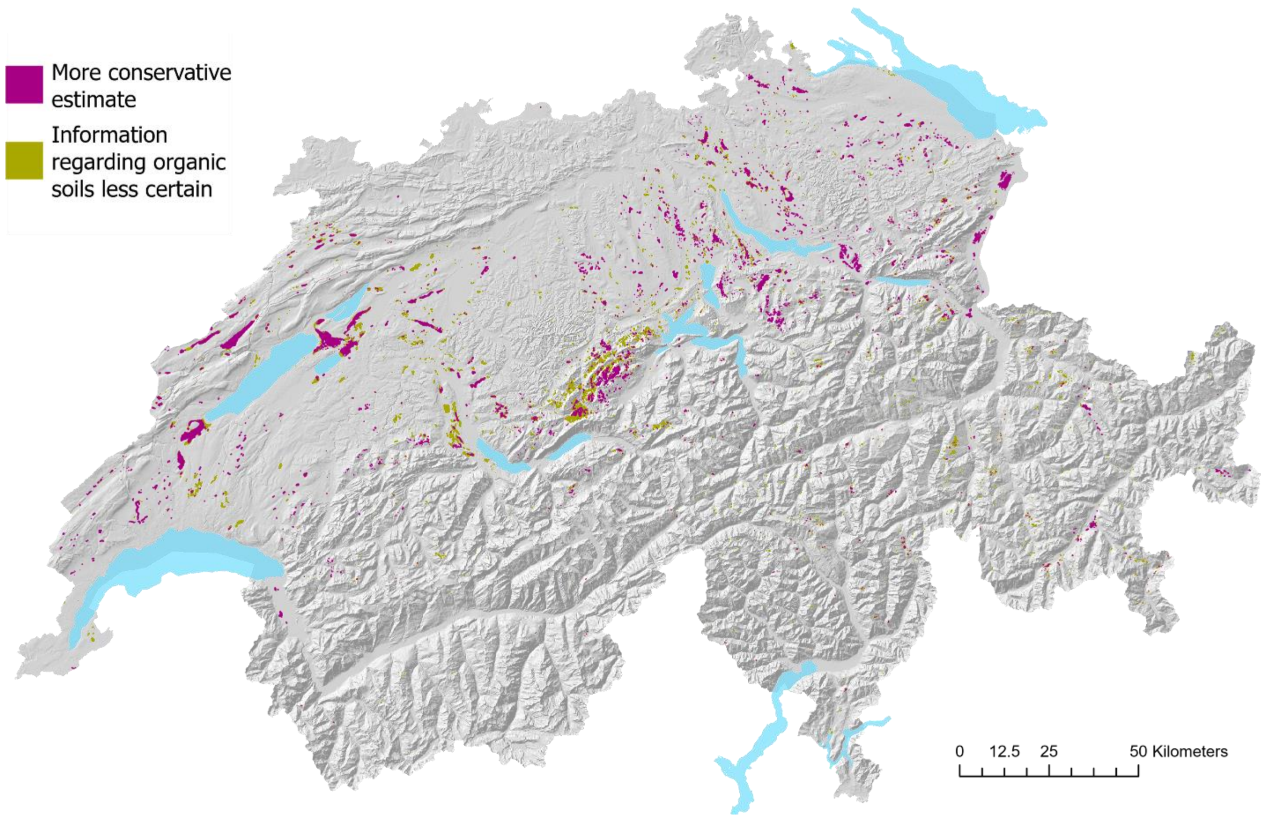

Figure 2. Estimates of the current distribution of organic soils. The more conservative estimate (pink) comprises surfaces for which the information regarding organic soils is more certain; the less conservative estimate (pink and yellow together) additionally contains surfaces from which the information regarding organic soils is less certain; lakes and hillshade © Swisstopo; published under a CC BY-SA 4.0 licence.

(Figure 3) comprise 7,251 ha; these are not surfaces for which modern documentation shows there are nowadays mineral soils (these were excluded from the estimate, see Methods), but surfaces for which conclusive modern documentation is lacking.

The GHG emissions from organic soils calculated using national emission factors (FOEN 2024) are 828 CO_2-eq. in kt, of which 89 % is associated with CO_2 and 11 % with N_2O. Emissions from agriculturally managed organic soil surfaces sum to 740 CO_2-eq. in kt or 89 % of total emissions from organic soils. These emission totals fall within the range of emissions calculated using IPCC emission factors (Table S11).

Of Switzerland's organic soil area, 61 % is managed for agriculture (arable land and agricultural grassland including summer pastures) and 80 % is either used for agriculture or forested (Figure 4). Nationally, 1.5 % of agriculture occurs on organic soils; for arable land (including vegetable production) this value is 2.4 % and for grasslands it is 1.1 %; 7.6 % of vegetable production (including greenhouse production and field vegetables) occurs on organic soils (Table S11). Nationally, 1.9 % of the legally protected 'crop rotation surface' is organic soil. The minimum quota for this surface (see Introduction) would no longer be met if organic soil surfaces were to be excluded (Table 1).

The proportion of agriculture on organic soils at the local level (3 km^2) varies substantially and can be as high as 66 % (Figure 5). Agriculture on organic soils is particularly prominent in four regions: Ponts de Martel in Jura region (A in Figure 5), the Seeland region (B), the Orbe plain (C) and the upland region in the centre of Switzerland (D). Agriculture in the first region is dominated by grassland, in the Seeland and Orbe regions by arable land including vegetable production, and in the last (upland) region by summer pastures.

The 14 most important crops and grassland types, together making up 95 % of the farmed organic soil surface, each cover >300 ha of organic soils (Figure 6). Five of these - less-intensively managed meadows, extensively managed meadows and pastures, litter meadows and summer pastures - could be compatible with a raised water table (Birr *et al.* 2021). The other nine crops and grassland types (potatoes, beets, rape seed, cereals, leys, corn, vegetables, intensively managed meadows, and pastures) are less compatible with water tables high

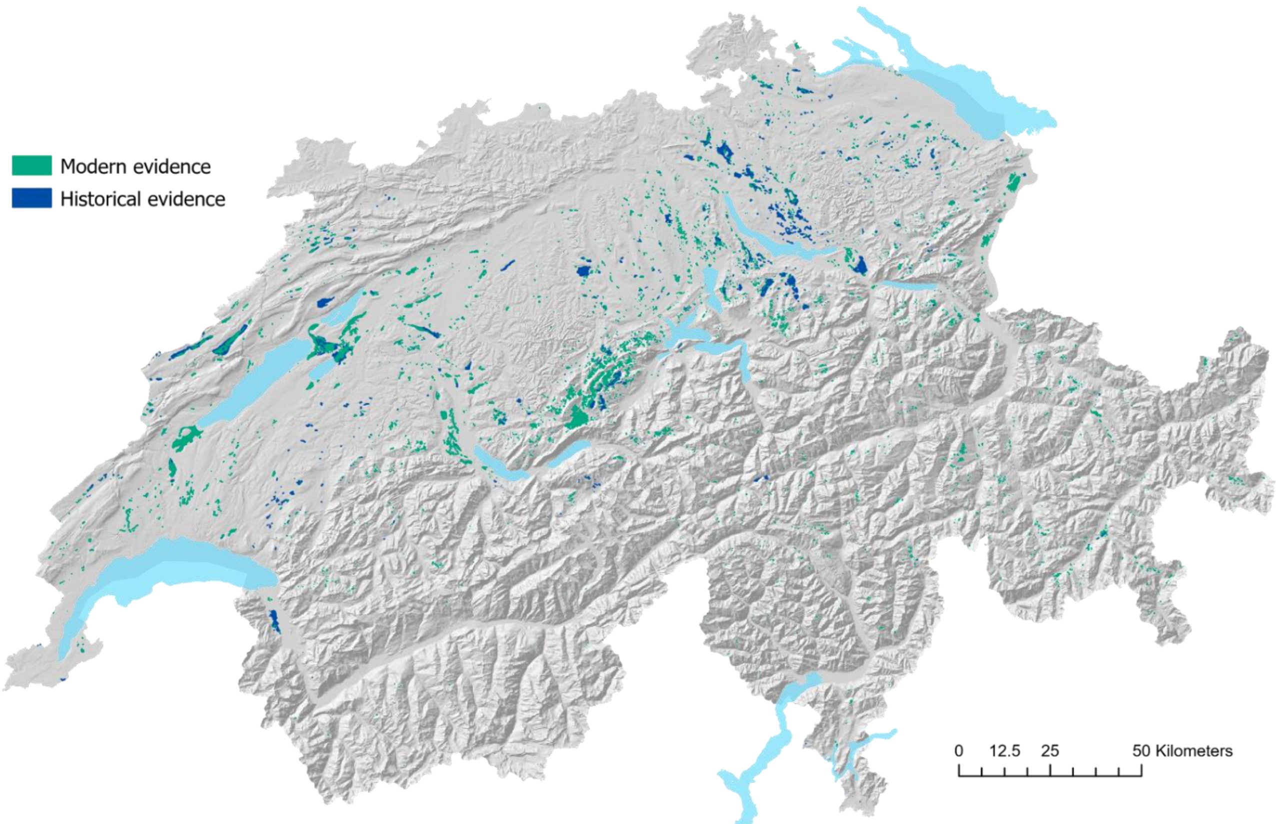

Figure 3. Estimate of the surface of organic soils, for which there is modern evidence (green, i.e., the less conservative estimate from Figure 2) or only historical evidence (dark blue) of organic soil. Lakes and hillshade © Swisstopo; published under a CC BY-SA 4.0 licence.

enough to substantially reduce GHG emissions. Of these, seven are over-proportionally represented on organic soils (Figure 6), up to 5.5 times (for vegetables) more than would occur if they were distributed proportionally on organic and mineral soils. The contribution margins of the nine crops or grassland types not suitable for cultivation with a raised water table range from 385 to 5,779 CHF ha^{-1} (Table 2). With the exception of paddy rice and to a lesser extent *Typha* or less intensively managed meadows, contribution margins of wet crops are typically lower than those of conventional crops.

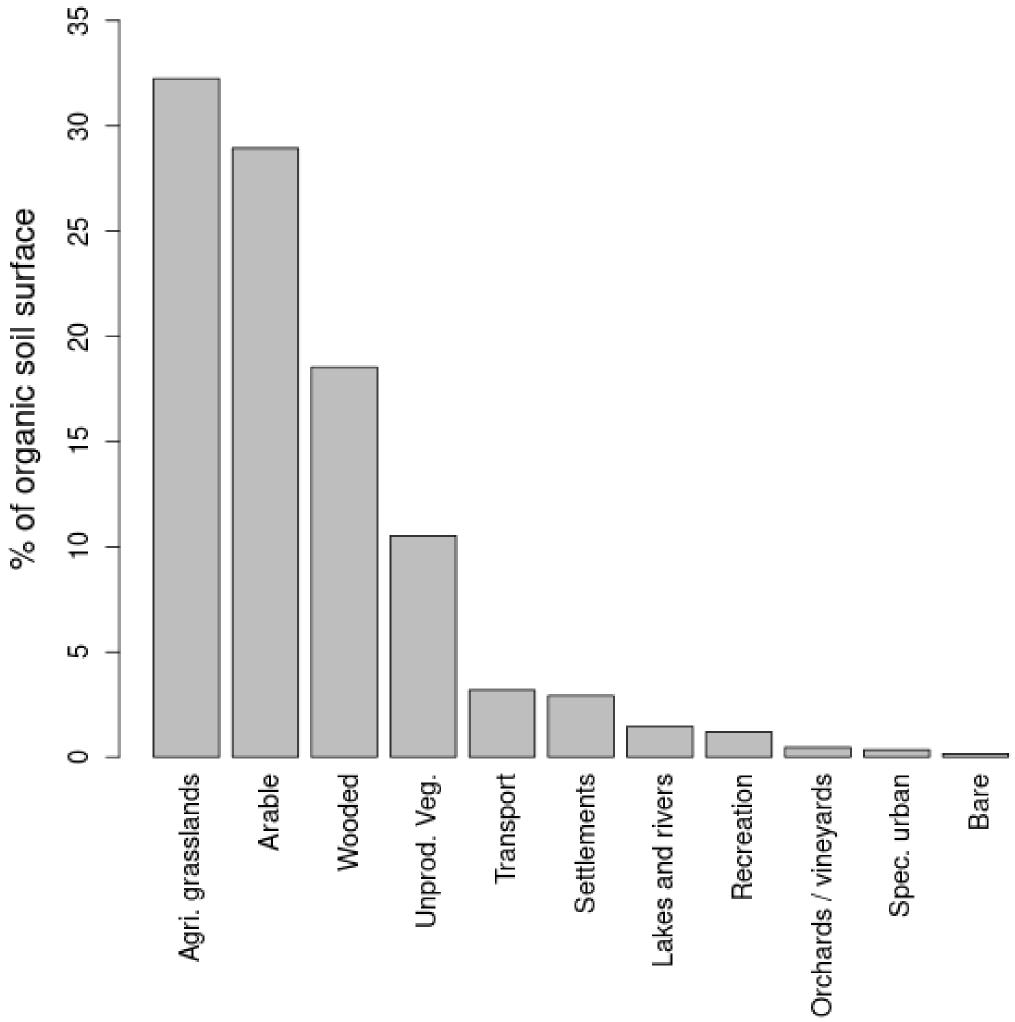

Figure 4. Land use of organic soil surfaces across Switzerland (2014–2019).

Table 1. Overlap of organic soil surface with the crop rotation surface (CRS). The CRS comprises the best surfaces for arable cropping, and there is a legally required minimum extent (quota) of these surfaces.

	Hectares
The CRS surface in 2023	445,680
The CRS minimum quota	438,460
Overlap of organic soils and the CRS	8,414
Deficit in CRS if all organic soils are removed from agricultural management	1,194

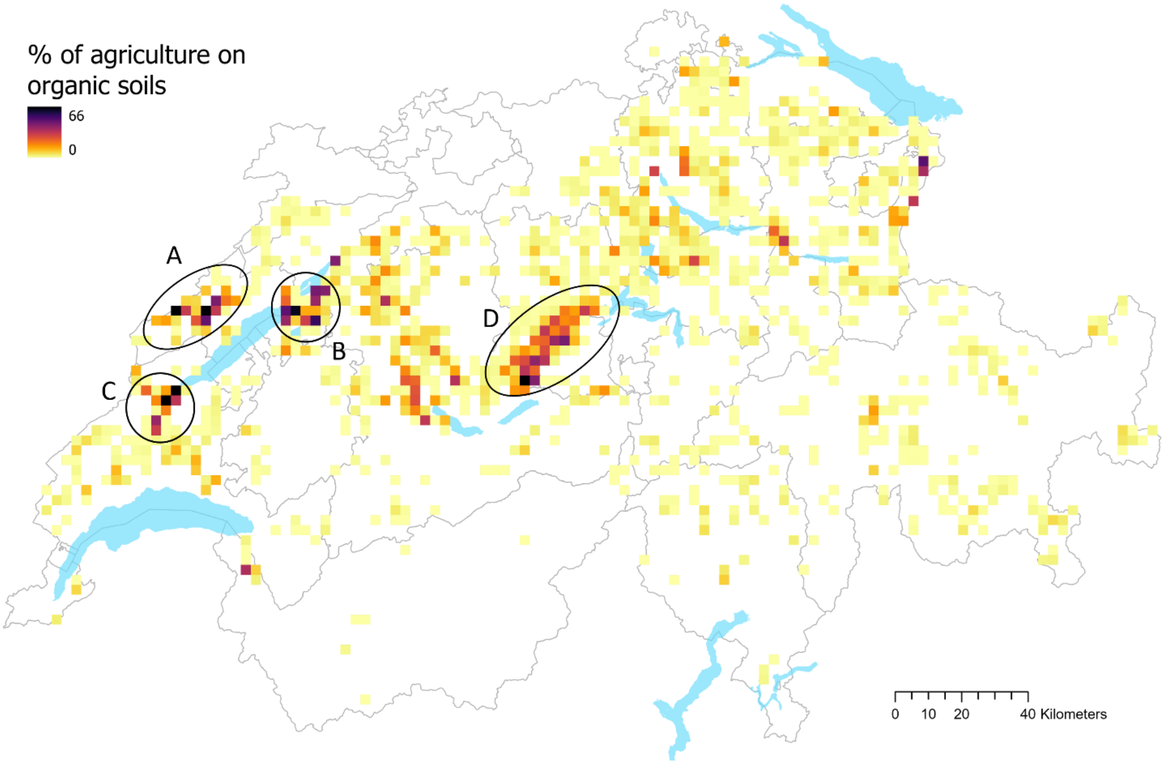

Figure 5. The proportion of agricultural land in a 3 km^2 grid square that lies on organic soil; the regions A, B, C and D are referred to in the main text. Lakes and canton boundaries © Swisstopo.

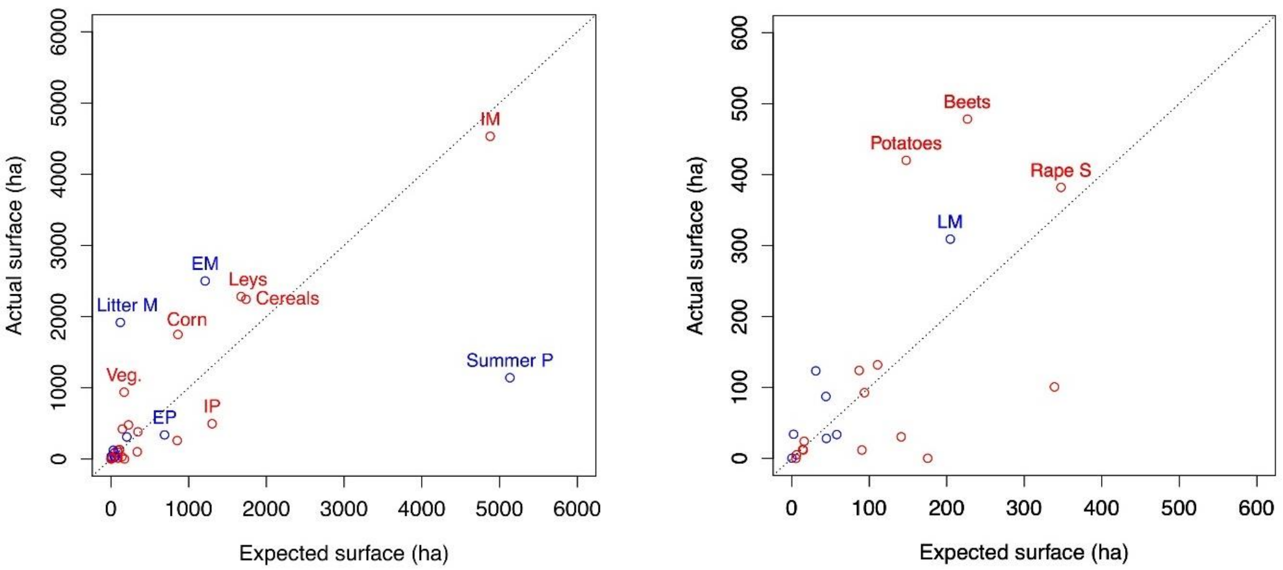

Figure 6. The expected (assuming a proportional distribution on mineral and organic soils) and actual surfaces of organic soils under different agricultural management; left-hand panel shows names of crops / grassland types > 600 ha; right-hand panel shows names of crops / grassland types 300–600 ha (names of crops / grassland types with actual surface < 300 ha not shown); blue = management compatible with raised water level; red = management less compatible with raised water levels (see text for details); abbreviations as follows: M = meadow, P = pasture, I = intensively (int.) managed (for pastures, > 3 livestock units [LU] ha^{-1} grazing period^{-1}; for meadows 143–170 fertiliser-N in kg ha^{-1} yr^{-1}; both assuming low elevation), L = less int. managed (meadows only, 26–38 fertiliser-N in kg ha^{-1} yr^{-1} at low elevation), E = extensively managed (for pastures, < 1 LU ha^{-1} grazing period^{-1}; for meadows, no fertiliser), Rape S. = rape seed, Corn = grain and silo corn, Veg. = vegetables.

Table 2. Marginal contributions of the crops whose management is less, or more, compatible with a raised water table (see main text for details). Ranges of values (where given) indicate either range of management intensity (for conventional crops) or unfavourable or favourable scenarios (for wet crops).

	Lower	Upper	Source
	(CHF ha^{-1} yr^{-1})		
Management less compatible with raised water table			
Intensively managed meadow	385	504	Dietiker *et al.* 2021
Intensively managed pasture			No information found
Ley	815	859	Dietiker *et al.* 2021
Cereals	1,563	1,844	Dietiker *et al.* 2021
Corn	1,311		Dietiker *et al.* 2021
Vegetables			No information found
Beets (sugar and fodder beets)	3,041		Dietiker *et al.* 2021
Potatoes	5,779		Dietiker *et al.* 2021
Rape seed	1,446	1,995	Dietiker *et al.* 2021
Management more compatible with raised water table			
Typha latifolia	-2,224	2,740	Fabian *et al.* 2024
Phragmites australis	-287	677	Fabian *et al.* 2024
Sphagnum spp.	-440	665	Fabian *et al.* 2024
Salix spp.	-40	58	Fabian *et al.* 2024
Oryza sativa	10,513		Fabian *et al.* 2024
Extensively managed meadow	249		Dietiker *et al.* 2021
Less intensively managed meadow	552		Dietiker *et al.* 2021

DISCUSSION

Methodological approach

Mapping peatland surfaces at the national or global level can be carried out using a top-down (e.g., global-scale modelling) or a bottom-up (i.e., the consolidation of local or regional data) approach. Because peat cannot be mapped remotely, top-down approaches model proxies including hydrology, physiognomic features or vegetation (Minasny *et al.* 2019, Minasny *et al.* 2024). The corollary of this is that top-down approaches are most suitable for regions whose peatlands are in a near-natural state, because proxies become unreliable when the hydrology or vegetation are strongly altered (Minasny *et al.* 2024). In Switzerland, < 10 % of organic soils are near-natural, implying that top-down approaches are suitable for only a small fraction of organic soils. The long history of drainage in Switzerland also precludes the use of historical remotely sensed data that could be used if peatland degradation had occurred in the last few decades (FAO 2020). An additional problem is the fact that in the past peat was extracted from many peatlands, sometimes until no peat remained at a site (Früh & Schröter 1904, Lüdi 1973). This, alongside the long history of drainage, makes the use of predictive modelling challenging; a site that was once a peatland might be predicted as such on the basis of environmental factors, but the peat might no longer be there, having either been removed (peat extraction) or decomposed. However, a combination of digital mapping with satellite data (supplying information on the status quo) and targeted sampling from sites which were peatlands in the past might be a promising alternative approach to mapping of these soils for Switzerland in the future. A further information source might be radiometric data, which measures naturally occurring bedrock radiation and has been successfully used to map peatlands in Ireland (Minasny *et al.* 2019, Beamish & White 2024).

Bottom-up approaches have been used at several scales including continental (e.g., Tanneberger *et al.* 2017) and global (UNEP 2022) scales. Although Swiss organic soils contribute only a very small amount to the European and global total areas, the methodological challenges and solutions relating to the bottom-up method encountered in this study are also relevant to these smaller scales. First, the various maps used might have different classification systems, potentially jeopardising consistency across the map (Minasny *et al.* 2024). Consistency can however be upheld, either by using data from national surveys (in this study e.g., national mire inventories) or surveys that have standard classification systems (in this study e.g., forest habitat maps). For other data sets where inconsistency might occur (in this study e.g., geological maps), this can be minimised by harmonising the data sets (as in this study, see Methods). Second, the bottom-up approach relies on published data sets, meaning it is vulnerable to data gaps. This is indeed the case in this study; e.g., one canton contains 11 % of Swiss forests but has no forest habitat map, and only 19 % of agricultural surfaces are covered by soil maps of adequate quality (Rehbein *et al.* 2019). Third, properties of the final map, including scale and age, vary across space. For this reason, we recommend the map generated in this study is suitable for use at cantonal and smaller scales only.

The map has not been validated, but could be validated in the future, using information from new soil samples as they become available. However, the nature of the map makes a nationwide validation difficult. The map is derived from hundreds of information sources, meaning that individual surfaces are derived from different input data. The corollary of this is that a large number of surfaces would need to be checked in order to validate the map, since showing the map to be accurate (or not) in one region would not mean that it is accurate (or not) in another region. A more meaningful approach is, therefore, to validate individual data sets. Wüst-Galley *et al.* (2015) and Wüst-Galley *et al.* (2016) ground-truthed two important data sets (the Caricion fuscae vegetation type of the fen inventory and three relevant categories of the forest habitat maps), confirming they were reliable indicators of organic soils. Additionally, the reliability of other information sources was checked by comparing them to more reliable information sources, namely soil maps, where an overlap occurs.

Surface

The new estimate of the organic soil surface includes a gross gain of 9,588 ha (roughly one-third of the previous surface), which was obtained by including merely 9–10 years' worth of newly available data. This implies that the current (new) estimate is also an underestimate and can be improved in the future by incorporating more data, most importantly forthcoming geological and soil maps. On the other hand, some surfaces might have been erroneously classified as organic soil. This is due to a combination of continual C losses from drained surfaces and the use of maps up to 80 years old as direct evidence of organic soil; it is possible that a drained peatland which had a 1 m thick peat layer 80 years ago no longer contains enough C to meet the definition of an organic soil. This would lead to an overestimation of the organic soil surface, although the implication for emissions is less clear (see next section).

The original extent of organic soils in Switzerland is estimated to have been between 97,000 ha and 149,000 ha (Wüst-Galley *et al.* 2019), implying a loss of 65–77 % of organic soils when compared to the estimate made in this study. The corresponding mean annual loss (0.34–0.48 %) corresponds well to the long-term rate of wetland change for Europe (Davidson 2014).

Emissions

GHG emissions from organic soils make up to 2.0 % of Switzerland's total emissions. The emission estimate from organic soils is however associated with uncertainty. Following FOEN (2024), we assumed all agriculturally-used grasslands outside of the summer pasture region to be deeply-drained, but it is possible some of these might have a higher water table than the assumed -30 cm. This might be very important, as a sigmoidal relationship between water level and CO_2 emissions has been demonstrated (Tiemeyer *et al.* 2020, Koch *et al.* 2023), meaning a slightly higher water level leads to much lower emissions. At the national level, there are no water level monitoring programmes in organic soils in Switzerland. To reduce the emission uncertainty, one strategy would be to implement a water level monitoring or modelling system, such as that in Denmark (Koch *et al.* 2023), and to develop water level dependent emission factors. This will become more important in the future as the number of GHG emission mitigation projects, such as the re-wetting of organic soils, increases.

There are several reasons why we might have underestimated emissions from organic soils. First, as stated in the previous section, the estimate of the organic soil surface itself is likely to be an underestimate. Second, we did not explicitly include those C-rich soils that may have enough organic C to qualify as organic soils according to IPCC (2006) but are not organic soils according to national soil mapping guidelines. These include former organic soils that have lost a lot of C, as well as those that have been covered with a mineral soil layer to facilitate their agricultural management (Leifeld *et al.* 2019b). A lack of information means their extent and distribution, and thus their emissions, cannot currently be estimated. We also did not include C-rich soils of wet origin that *do not* qualify as organic soils according to IPCC (2006). Both groups of soils are however relevant for emission estimates, as field measurements (Leiber-Sauheitl *et al.* 2014, Eickenscheidt *et al.* 2015, Tiemeyer *et al.* 2016) and incubation experiments (Liang *et al.* 2024) show their emissions can be just as high as those of 'typical' organic soils. While it is likely that some of these soils were implicitly included in the current study (e.g., surfaces surveyed decades ago as 'peat' but that have lost so much C that the organic C content is now below 12 %), we have most likely underestimated their extent because these soils are not recognised as 'peat' or 'organic soils' in the national soil mapping guidelines. Underestimating GHG emissions by disregarding C-rich soils has been highlighted as problematic for other regions (Eickenscheidt *et al.* 2015, Liang *et al.* 2024). In summary, we suggest that the estimate of GHG emissions from C-rich soils in Switzerland is an underestimate. Moving away from a category-based approached (mineral vs. organic soils) and, rather, incorporating information on soil C density alongside water level estimates, would improve the estimate of emissions from these soils.

Despite these shortcomings, the GHG emissions presented here can elucidate the role of these soils at a coarse level. Switzerland is aiming for net zero GHG emissions by 2050 (Federal Council 2021). Emissions from organic soils (all land uses) comprise ~2.4 % of the GHG emission savings that need to be made to reach this goal. Annual emissions from organic soils managed for agriculture amount to 10 % of national agricultural emissions (including energy and land use, FOEN 2024). Furthermore, they comprise ~25 % of the emissions that need to be eliminated between 2020 and 2050 according to the climate strategy for agriculture and nutrition (FOAG *et al.* 2023). Because, on average, rewetted organic soils still emit GHGs (Wilson *et al.* 2016), this does not mean that rewetting all of Switzerland's organic soils would result in a quarter of the agricultural emission reduction target being met, but it does indicate a relevant mitigation potential. In a drained state, these soils are highly problematic for reaching emission targets. Whilst this has been shown at the global scale (Leifeld *et al.* 2019a, Günther *et al.* 2020), we show it is relevant also for a country with a relatively small proportion (< 1 %) of organic soils.

Strategies and future paths

At the national level, Switzerland's organic soils are predominantly (~80 % in total) under agricultural and forestry management, and in the valley region including the large lowland central plateau north of the Alps, 54 % of organic soils are arable land. At the national level, organic soils are not particularly important for agriculture in terms of area, as only 1.2 % of agriculture occurs on them. Re-wetting these surfaces either for restoration or for paludiculture would thus result in a relatively small loss of (conventional) agricultural surface at the national scale. At the regional scale, however, the situation can be quite different; in some regions organic soils are not relevant for agriculture, yet in other regions up to 66 % of agriculture (on the Orbe plain) occurs on them (Figure 5). This variation means that a variety of strategies and measures will be necessary to deal with drained organic soils in different regions; a single strategy or set of measures for the whole country will not suffice (Girkin *et al.* 2023).

Following the framework of Girkin *et al.* (2023), restoring wetland habitats should be prioritised in those regions where organic soils are of less importance to agriculture. In such regions, farming income losses could most easily be compensated and the loss of production would be minimal; the substantial reduction in GHG emissions that would result from the change in land use needs to happen somewhere. Peatland restoration could also be prioritised for organic soil surfaces covered with commercially unproductive vegetation, irrelevant of the geographical region. Although these surfaces contribute only a small proportion (~8.5 %) of emissions from organic soils, it can be expected that their relatively low economic importance means habitat restoration should have high social acceptance. Furthermore, these sites include degraded raised bogs and fens whose quality is decreasing primarily because they are drying out (Bergamini *et al*. 2019); reinstating their original water regimes should therefore lead to improvements in habitat quality as well as reduced GHG emissions. In regions where agriculture is most dependent on organic soils, continued agriculture with changes in management could be prioritised (see next section). At a more local scale, one further aspect relevant to decision making is the C stock of an organic soil, which is a function of the area and depth of the C-rich layer, its C content and bulk density. In the long term, soils with higher C stocks have the potential to release more CO_2, meaning these should be prioritised for rewetting.

One measure that is claimed by practitioners to reduce GHG emissions while maintaining commodity production is the addition of a mineral soil layer on top of organic soils. This is being carried out increasingly in Switzerland to counteract soil subsidence (Ferré *et al.* 2019) and, if the resulting surface fulfills the requirement of high-quality arable soil, this measure also enables the legally-required minimum quota of the 'crop rotation surface' to be maintained. However, the few studies that measured GHG emissions from fields with mineral coverage reported emissions typical of drained organic soils (Höper 2015, Tiemeyer *et al.* 2016), indicating this measure does not reduce emissions. The only study specifically testing the role of a mineral soil layer in reducing GHG emissions reported reduced N_2O emissions only, hypothesising that a mineral soil cover can minimise CO_2 emissions only if it is accompanied by a raised water table that ensures the peat layer is completely water-saturated (Paul *et al.* 2024). Though it is viable under current economic conditions, the potential of this measure to mitigate GHG emissions from farmed organic soils still needs to be tested.

One alternative to conventional (drained) cultivation is paludiculture. It is barely practised in Switzerland, but 30 % of farmed organic soils are managed in a way that is compatible with paludiculture (Figure 6). Although the costs and technical feasibility of raising water tables on these surfaces need to be considered, this implies that GHG emissions for roughly a third of the farmed organic soil surface could be reduced without changing the crop or grassland type cultivated. For the remaining surfaces, a change in crop or grassland type will be needed to substantially reduce emissions.

Such changes in management are typically associated with a reduced income (Freeman *et al.* 2022); we show that the marginal contributions of conventional crops are generally much higher than those of paludicultures (Table 2). However, it is possible that the widely applied approach of using marginal contributions of conventional crops on drained peatlands as an economic indicator falls short of representing the actual costs of peatland agriculture, which also include drainage systems (covered to a large extent by the government in Switzerland) as well as societal costs of air and water pollution, and soil loss. On the other hand, it is likely that the marginal contributions of certain conventional crops presented here are under-estimates for cultivation on *organic* soils. This is because ten of the most commonly grown crops are over-proportionally represented on organic soils (Figure 6), suggesting there are further benefits for farmers associated with growing these crops on organic soils, at least in the short term. Although we found no quantification of this, the agricultural benefits of cultivating organic soils are mentioned for other regions (e.g., in the tropics, Andriesse 1988; in the UK, Page *et al.* 2020). In summary, the high marginal contributions of conventional crops compared to paludiculture indicate that the latter is generally not economically viable under current economic conditions. One exception to this is paddy rice. Although paddy rice is currently not grown as a paludiculture in Switzerland (the water table is low over winter months), the fields are flooded in summer, providing opportunities for GHG savings (Wüst-Galley *et al.* 2023). Through direct marketing, paddy rice is a profit-making crop under current economic conditions in Switzerland (Table 2; Gramlich *et al.* 2021).

One further aspect of rewetting farmed organic soils is the high-quality 'crop rotation surface', of which a minimum amount is *legally* required for food security and which might be at risk if the water table is raised. Excluding areas of organic soil would eliminate the current surplus of crop rotation surface, even resulting in a small deficit (Table 1). Rewetting these soils would therefore require either legislative changes or compensation for the lost agricultural surface elsewhere. However, it could be argued that (drained) organic soils should not form part of the 'crop rotation surface' because drainage accelerates decomposition, meaning the contribution of these soils to food security might be impaired in the long term. This becomes even more relevant if the flood risk from surface water run-off is considered; the low hydraulic conductivity of degraded organic soils (Lennartz & Liu 2019), in conjunction with their low elevation, makes them prone to flooding after heavy rainfall (Ferré *et al*. 2019).

CONCLUSION

From the perspective of GHG emissions, it is widely recognised that the agricultural use of organic soil has to change. How best to deal with these problematic soils depends on the context in which they are found, and this article presents some basic contextual information regarding this issue for Switzerland. Although the organic soil surface in Switzerland is small, the distribution of these soils and the resulting local dependence of agriculture on them means that the problems encountered will also occur in other countries with high population density or extensive organic soils.

In regions where peatland restoration is not viable, alternative measures must be sought. One possibility is mineral coverage in combination with raised water table, although the effectiveness of this procedure in reducing GHG emissions needs to be quantified. A second alternative is paludiculture. Although we show that paludiculture is - with the exception of paddy rice cultivation - economically challenging in Switzerland, it could be made more attractive to farmers by changing economic conditions. Exploring mechanisms for economic change is beyond the scope of this article, but one potential mechanism may be farm subsidies which, on average, make up 20 % of farm incomes (Jan *et al.* 2024) and which are already used in part to promote measures benefitting soil conservation and biodiversity. Another potential mechanism is carbon farming schemes, such as those implemented by the UK Peatland Programme and the German Moorfutures scheme. Exploring financial mechanisms to promote the re-wetting of organic soils is thus a research need in Switzerland. However, land area is finite and in countries where land for agriculture is scarce, this presents an additional challenge. Such countries cannot necessarily find extra land to grow the calories that are lost if organic soils are rewetted. The challenge becomes more acute if current diets are to be maintained and imports of feed or food are not to be increased. Alternative approaches including increased productivity on mineral soils, dietary changes, and/or reducing food waste could reduce the necessary agricultural surface. For example, at the global scale, Poore & Nemecek (2018) showed that halving the consumption of animal products would reduce agricultural land use by 51 % and reducing consumption of oils, sugar, alcohol and stimulants by 20 % could reduce land use for these products by 39 %. For Switzerland, von Ow *et al.* 2020 showed that merely adhering to the national recommended diet could achieve a 48 % reduction in environmental impacts, including land competition. Such changes will be necessary to address this challenge.

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AUTHOR CONTRIBUTIONS

CWG originated the work, developed the workflow conceptually and put together the new map, carried out analyses and wrote the manuscript; JL contributed to the study's concept, the analyses and the manuscript.

DATA AVAILABILITY

The spatial data of the new organic soil map (2 data sets) are available from the Zenodo Data Repository:
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SUPPLEMENT containing Tables S1 to S12 is provided as a separate download (pdf file).