Recent Applications of Peat Resources Utilization and Its Environmental Impacts Mitigation – A Review

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In this paper, the pathway from peat resources to their various forms of utilization, to their environmental impacts, and to their mitigation is discussed. Overall, research gaps related to various fields of peat studies are identified and recommendations for future research are presented. Global peat reserves are large, but they remain mostly unused. Peat can also be valorized in various less GHG-intensive applications beyond combustion. Heterogenocity of peat may constraint many of these uses, warranting new innovations to support their feasibility. The theme of GHG emissions from peatlands is complex. The future of northern peatlands as carbon sinks remains uncertain. Paludiculture, an emerging research field, and its GHG mitigation potential is also discussed. When discussing peat production, one of its main adverse effects typically disclosed is its impact on the water quality of natural aquatic systems. Thus, different traditional and novel peat bog drainage water treatment methods are extensively compared with each other — a topic not previously presented in the literature. Peatland restoration and its novel applications, as well as economic aspects, are also addressed. Socioeconomic aspects of peat use, closely linked to climate, food, and rural livelihoods, are currently under vigorous research, and are also examined.

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INTRODUCTION

A mire is an ecosystem with a high groundwater level, where organic material accumulates and forms peat through the slow biodegradation of wetland plants under anaerobic conditions (International Peatland Society 2025). Peat formation is a process that takes several thousand years (Loisel and Gallego-Sala 2022). It precedes the formation of fossil fuels (Nadon 1998). The reaction sequence that precedes the formation of coal from peat is known as coalification—a geological process in which organic material is transformed into coal. During the biological stage, peat is formed, which gradually transforms into coal under the influence of relatively high temperatures and significant pressure. The formation of coal is further enhanced by geological time and tectonism (Marsh and Rodríguez-Reinoso 2006; Flores 2014). The degree of maturation reached during the hardening process of organic material increases gradually and can be determined based on the measured C/H ratio, as well as the residual concentrations of oxygen, sulfur, and nitrogen (Marsh and Rodríguez-Reinoso 2006).

Peat biodegradation refers to the decomposing of plant residues, resulting in the formation of humic substances. This reaction is influenced by temperature, pH, and moisture conditions. The degree of peat biodegradability indicates the environmental conditions under which it was formed. (Drzymulska 2016). Main contents of anaerobic biodegradation reactions are 50 to 70% methane (CH_4) by volume, 30 to 50% carbon dioxide (CO_2), and traces of other gases (Jameel *et al.* 2024). Peatlands are typically under anaerobic conditions, although the hydrological dynamics of the upper layer of peat (the acrotelm) are directly related to fluctuations in the water table. Thus, the acrotelm is not always saturated (Holden and Burt 2003).

Since peat is generally classified as a non-renewable fuel (although theoretically it is slowly renewable), comparing its emissions to those of other biomass sources is problematic, especially because it affects its sustainability status (economic and ecological implications) (Vainio *et al.* 2024). Regarding this, rates of peat formation are controlled by a combination of climatic (temperature, moisture), hydrological (water-table depth) and vegetation/ peatland-specific factors (Fenton 1980; Omar *et al.* 2022; Swindles *et al.* 2025). Tropical high temperatures favour organic matter decay, but high moisture content promotes peat formation from plant remnants. Tropical peatlands generally accumulate peat at a rate of 1 to 5 mm/a (and up to 10 mm/a), a rate much higher than that of boreal and subarctic peatlands (Omar *et al.* 2022).

Fenton (1980) studied three Antarctic moss banks in detail — two of which were dominated by *Polytrichum alpestre* and one by *Chorisodontium aciphyllum* — finding a moss bank growth rate of 0.9 to 1.3 mm/a. In a recent study by Swindles *et al.* (2025), apparent peat accumulation rates over the last two millennia in 28 well-dated European peatlands ranged between 0.05 to 4.48 mm/a (mean 1.18 cm/a), supporting the common estimate of ~1 mm/a. Highest rates occurred in Scandinavia and the Baltic, lower in Britain, Ireland, and Continental Europe. Summer temperature significantly controlled peat accumulation, and higher rates are often linked to wetter conditions inferred from water-table depth.

Peatlands function as carbon sinks in much the same way as forests where trees grow. Mires must be drained before they can be used for peat extraction, which has socio-economic impacts, and also introduces environmental risks—some of which can be managed through legislative changes. In this essence, protection of natural water bodies is a key aspect that needs to be taken into account. As a matter of fact, when peat production is discussed, one of its main adverse effects typically disclosed is its impact on the water quality of natural aquatic systems. Thus, more effective solutions for treatment of peat water should be developed (or the operation of current methods enhanced). It should also be noted that the human-centered exploitation of bogs has led to their large-scale degradation. For example, 25% of Europe's peatlands are degraded, and within the EU the share is around 50% (≈ 120,000 km²); degradation increases from north to south (Tanneberger *et al.* 2021). Therefore, recommissioning and protecting them is important, taking into account the related costs. Peat also has multiple uses (typically producing less CO_2) beyond combustion, which are discussed in this article.

The aim of this paper is to provide broad insights into recent applications of peat resources and the mitigation of environmental impacts. Socioeconomic aspects of peat utilization are also discussed. Research gaps are identified and recommendations for future research are presented. To the best of our knowledge, no recent comprehensive review has been published that covers all the thematic areas presented in this article. Particularly papers with a similar framework (from peat resources to their various forms of utilization, to their environmental impacts, and to their mitigation) have been missing.

CHEMICAL CHARACTERIZATION OF PEAT

Factors influencing peat formation and properties include climatic conditions, as well as geological, geomorphological, and hydrological factors (Hu and Ma 2002). The moisture content of dead plants strongly affects the quality of organic matter present in peat. Under moist conditions, microbes convert non-humic substances such as hemicellulose, cellulose, lignin, pectin, bitumen, waxes, resins, nitrogen-rich materials, lipids, amino acids, unsaturated and saturated fatty acids, various types of starch carbohydrates, oils, organic sulfur compounds, balsams, bioterpenes, and tannic acids into stable humic substances (Trckova *et al.* 2005). The formed humic substances, *e.g.* humic, fulvic, ulmic acids, and humins, make up the majority of the partially decomposed peat (Kocabagli *et al.* 2002; Janos 2003; Perminova *et al.* 2003). Peat compresses well, and its moisture content is high, while its shear strength is low, and it has low load-bearing capacity (Zainorabidin and Wijeyesekera 2007; Zainorabidin *et al.* 2007).

The same metals and non-metals found in the surrounding soil are also present in peat. Nearly all (about 90%) of the total elements found in peat ash consist of Si, Al, Fe, Ca, Mg, Na, and P (Hu and Ma 2002). These elements are also found in rainwater, along with *e.g.* K and Ti. The original pH of peat is around 3.3 to 3.8 (Rahman and Ming 2015). According to Rezanezhad *et al.* (2016), peat decomposition is widely assessed using the von Post scale (H1 = least, H10 = most decomposed). Changes in peat composition with age are presented in Table 1.

Peat is classified into three main types. The first is fibrous peat, which is slightly decomposed and still has a recognizable plant structure. Hemic or semi-fibrous peat has a moderate degree of decomposition. The most decomposed peat, which is located at the bottom of the bog, is saprine or amorphous peat, which no longer has a recognizable plant structure (Gowthaman *et al.* 2022). Of these, surface peat (fibrous peat) is used for purposes other than generating electricity and heat, for which the well-decayed bottom peat is used (Arvola 2015).

Table 1. Changes in Peat Composition with Age (adapted from Arvola 2015)

Peat component	Slightly decomposed (H1–2)[1] [%]	Moderately decomposed (H5–6)[1] [%]	Highly decomposed (H9–10)[1] [%]
Cellulose	15 to 20	5 to 15	0
Hemicellulose	15 to 30	10 to 25	0 to 2
Lignin	5 to 40	5 to 30	5 to 20
Humic substances	0 to 5	20 to 30	50 to 60
Bitumen, wax, and resin	1 to 10	5 to 15	5 to 20
Nitrogenous substances	3 to 14	5 to 20	5 to 25

1 = The von Post scale classifying peat decomposition (H1 = least decomposed, H10 = most decomposed) (Rezanezhad *et al.* 2016)

An accurate understanding of lignocellulosic biomass composition is crucial, when its valorisation (especially chemical production) is targeted. On a dry basis, biomass (*e.g.*, energy crops, grasses, softwood, hardwood, and various secondary streams) typically consists of 35 to 51% cellulose, 20 to 33% hemicelluloses, and 13 to 30% lignin.

Furthermore, there is variance in the proportions of these components between different types and subtypes of lignocellulosic biomass (Segers *et al.* 2024).

GLOBAL PEAT RESERVES

Most of the global carbon resources are located in mires, with an estimated 500 ± 100 Gt C in northern peatlands. Northern peatlands cover an area of approximately 3.2 million km² (Loisel *et al.* 2017). Mires occur in all climate zones and continents, covering a total area of 4.23 million km², corresponding to 2.84% of the Earth's surface (Xu *et al.* 2018). This is also visible in Fig. 1, in which global peat distribution is presented. Peatlands sequester twice more carbon than all the world's forests combined (Dunn and Freeman 2011). Boreal and subarctic mires are found over large areas with abundant soil carbon reserves, as frozen and thawing peat. The major global peatland complexes are located in in the circum-arctic zone, with Western Siberian Lowlands (Russia), and the Hudson and James Bay Lowlands (Canada) containing a particularly high area of peatland (Xu *et al.* (2018).

Peatlands are found (Fig. 1) across over 180 countries (Parish *et al.* 2008). The distribution of peatlands varies depending on different environmental conditions. The largest peatland areas are in Canada (1.13 million km^2) and Russia (1.37 million km^2) (Xu *et al.* 2018). Peat growths occurs in rather wet conditions in the Nordic countries, because the winters at high latitudes are long (with snow-cover), whereas temperatures are moderate in the summer. Peatlands cover vast areas in the Nordic countries, including 66,680 km² in Sweden, 94,000 km² in Finland, and about 23,700 km² in Norway (Nordic Joint Committee for Agricultural Research 2008) Other areas with significant peatlands are located (Fig. 1) near to the equator (*e.g.*, the peat wetlands or swamp forests in Indonesia, Amazon, and the Congo Basin) (Xu *et al.* 2018).

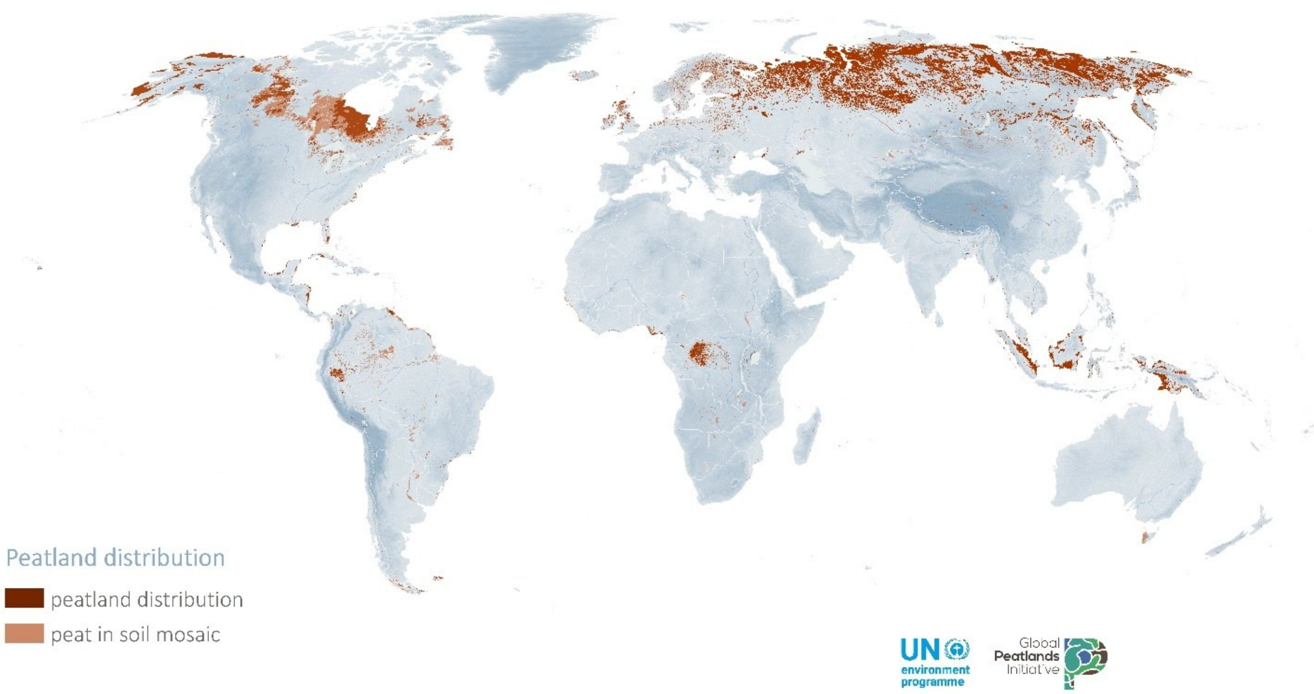

Fig. 1. The Global peatland Map 2.0 (Adapted from United Nations Environment Programme 2021).

UTILIZATION OF PEAT

Peat may be combusted for energy. However, in recent history, peat has played a significant role in national energy mixes mainly in a few countries, particularly Finland and Ireland, but its importance has declined markedly in recent years (Lempinen and Vainio 2023). In addition to combustion, peat has many different applications, some of which may be challenging to replace with other materials. Figure 2 shows a diagram of the global distribution of different forms of peat usage.

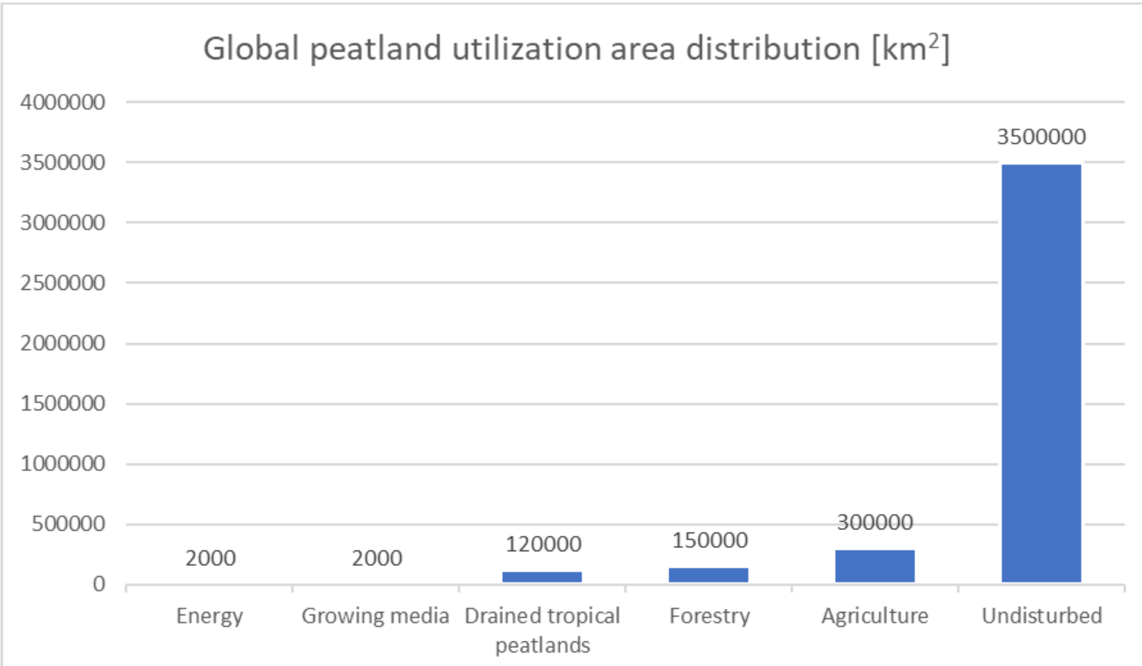

Fig. 2. Global peatland utilization area distribution (World Energy Council 2013; Clarke and Rieley 2010)

For many purposes, rapidly renewable peat from the less decomposed surface layer is used (Arvola 2015). In Finland, approximately 1.5 to 3 million cubic meters of peat are used for these purposes annually (Bioenergia ry 2024). In Finland and the Nordic countries, peatlands have been extensively utilized for energy production and forestry. Nevertheless, this is not significantly reflected in Fig. 2, as the peatland area in these countries is relatively small compared to the global total peatland area. Surface peat can be used as a growing medium, for absorbing urine and manure, for absorbing solvents and oil in both land and water areas; at a certain moisture content, peat becomes hydrophobic and begins to repel water. It can also be used as an oil spill adsorbent, a composting aid, for the purification of liquids and gases, in landfill capping, as a filler and reinforcing material, as a raw material for textiles (spun or felted into wool or silk) as well as for spa and therapeutic peat (Leinonen 2010; Perdana *et al.* 2017; Bioenergia ry 2024). In Finland, the main applications of peat are as bedding material in livestock farming, as a growing medium, and in landscaping and green area maintenance, and relatively small amounts are used for other purposes (Bioenergia ry 2024).

In addition, separating peat into fractions and using them can be a viable solution, for example, in the production of plastic composites. Favorable properties of peat in terms of mechanical processing are: fibrous, easy to grind, hydrophobic, antiseptic, adsorption/absorption capacity, cheap (Arvola 2015). Various possibilities for peat usage have been presented in Fig. 3. Some of the peat applications presented in Fig. 3 are in real-world use, while others are still more on the level of ideas and need more research to support the feasibility of producing/using them in the given application. According to Arvola (2015), peat composites have great potential, and peat products such as insulation and acoustic boards are already produced. Environmental peat, such as growing media and bedding materials, are also well-known products, as discussed earlier. To the best of our knowledge, chemical modification, thermochemical processing and chemical fractionation products with peat remain mainly of academic interest and have been studied less, mostly because of constraints in technical (*e.g.* high moisture content) and economic feasibility of such products, also as indicated by Arvola (2015). Combustion is the main utilization method for peat.

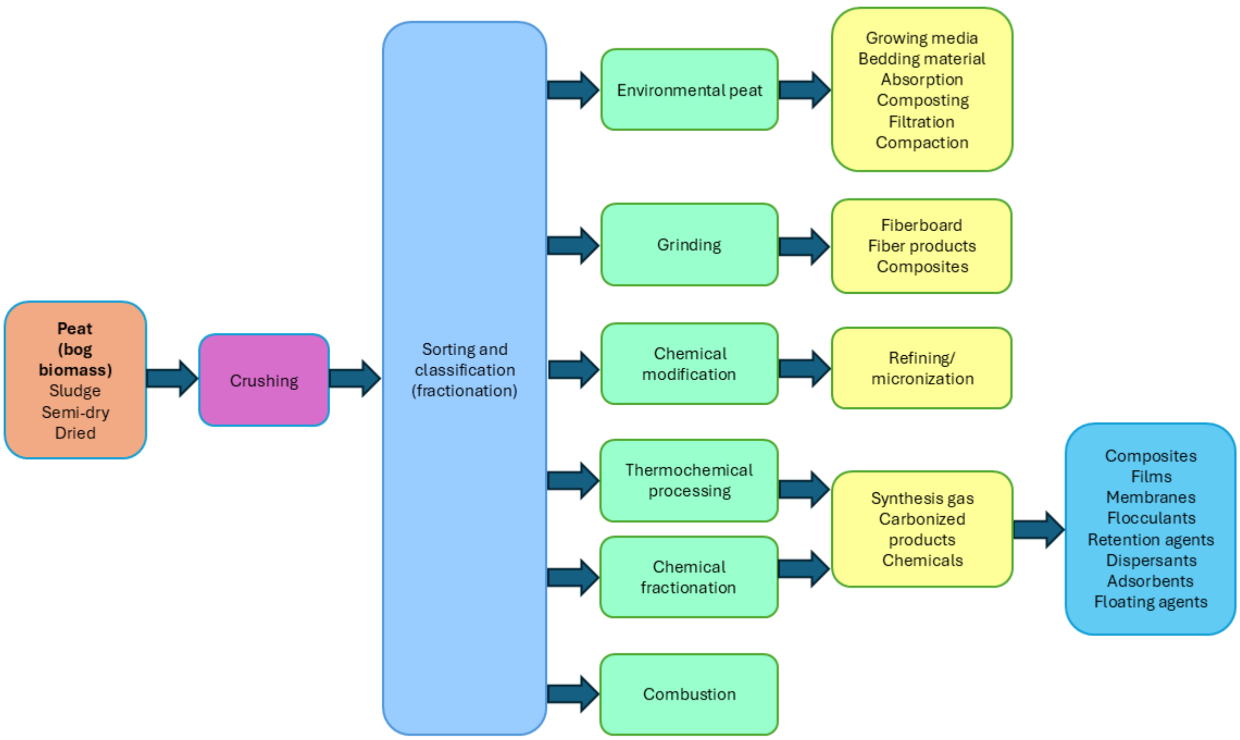

Fig. 3. Refining of peat and bog biomass. Adapted from Arvola (2015).

On the whole, to enhance real-world usability of peat refining pathways presented in Fig. 3, further investigation and new innovations are encouraged. When exploring new possibilities for peat valorization, it is important to consider what can be produced, at what cost, and its value to the user. Existing alternatives and what the new product would replace should be assessed, along with technological readiness, potential need for new technology, and related development and investment costs. Previous studies and their outcomes should be reviewed, key stakeholders identified, and the idea evaluated for political, economic, social, technological, environmental, and legal feasibility (Arvola 2015). The use of finely ground peat and Sphagnum moss would be technically (and likely also economically) feasible in polymer composites, particularly in polyurethanes, as functional filler materials (Koivuranta *et al.* 2017; Ämmälä and Piltonen 2019). However, further research on the subject is warranted.

A recent review discussed the varying lignocellulosic biomass valorisation pathways currently utilized or under development. These include variations in biomass composition, possible end-products, pretreatments, and conversion methods. Each valorisation pathway was found to have its own implications and potential. The process was found to be complex due to high multi-factor dependency. Optimal routes were not identified for linking biomass sources to end-products (Segers *et al.* 2024).

Many other biomasses are often considered rather homogenous compared to peat. Humic materials and inorganic substances contained in peat complicate direct industrial refining and value-adding chemical production. This makes the value-creation processes for other lignocellulosic biomasses more efficient and easier to optimize than for peat.

Use as Bedding Material and Nutritional Supplement

Certain properties of peat moss advocate its potential suitability for use in the poultry industry. Its efficient moisture management abilities (quick absorption and release of excess moisture) are beneficial in poultry houses. The natural pH value of peat is low (3.0 to 4.0) (Lee *et al.* 2021). Peat is beneficial in controlling ammonia and reducing bacterial populations in the bedding material. Peat moss can absorb water almost 8 times its own weight, while chopped wheat straw can absorb about 7 times its own weight (Shepherd *et al.* 2017). Peat can also absorb about 30 g NH_3/[kg dry peat] (Abbès *et al.* 1993). One study evaluated four different bedding types for cattle (long straw, chopped straw or without additives, and chopped straw/peat mix), and according to the report, emissions were lowest with the chopped straw/peat mix (Misselbrook and Powell 2005). The poultry industry uses bedding materials such as sawdust, rice husks, sugarcane pulp, bagasse, chopped straw, paper mill by-products, sand, wood shavings, corn cobs, oat hulls, dried leaves, and coffee bean husks (Toghyani *et al.* 2010; Ramadan *et al.* 2013).

In one of the few studies comparing peat to wood shavings, peat bedding was found to be healthier for broiler footpads (de Baere *et al.* 2009). A large Danish study (Kyvsgaard *et al.* 2013), on the other hand, found that differences in footpad condition didn't differ much when using wood shavings and peat. However, footpad condition was found worse when straw litter was used.

The stable's air quality, as well as management and composting of manure, may be enhanced with proper selection of bedding material with specific beneficial qualities. In a comparative study of bedding materials, wood chips, straw, peat, hemp, flax, sawdust, shredded newspaper, and various mixtures—peat/wood chips, peat/sawdust, and peat/straw—were used. Peat and peat-based mixtures showed the best performance in absorbing ammonia, water holding capacity (retention), and fertilizing ability of manure. Straw, flax, hemp, and peat were found to contain higher amount of fungi and bacteria than newspaper and wood-based materials (Airaksinen *et al.* 2001).

Poultry house litter contains nutrients, which are essential for plant growth (N, P, K, S, Ca, Mg, B, Cu, Fe, Mn, Mo, and Zn), proposing its use as an excellent fertilizer (Subramanian and Gupta 2006). Using peat as a feed supplement has increased, and peat products are commercially available. This is particularly due to peat's growth-stimulating ability in piglets and sows, as wells as its ability to prevent enteric diseases in them. Indeed, because peat contains biologically active substances and is readily available, in addition to agriculture, it has also been used in human and animal medicine (Trckova *et al.* 2005).

Combustion and Use as a Chemical Precursor

Prior to combustion, peat must be dried. This may be done either by solar drying on the open fields of bogs, or by using various industrial drying methods. Before drying, peat may contain up to 95% water, whereas typically drained bogs have a moisture content of 89 to 91%. Three commercial peat types are commonly presented: milled peat (moisture content of 40 to 50 percent), air-dried sod peat (moisture content of 30 to 40 percent) and artificially dried compressed peat briquettes (moisture content of 10 to 20 percent). Large-scale mechanized extractions of dried peat are typically used to produce milled peat. The effective calorific value of milled peat at operating moisture content is about 10,5 MJ/kg. Sod peat and peat briquettes are produced on a smaller scale, applying either dry or wet conditions and manual, semi-mechanical, or mechanical methods. (Andriesse 1988).

Wood material contains more potassium and sodium than peat. Soluble alkaline substances are released into the gas phase during combustion, causing corrosion of the combustion pipes. Chlorine-containing ash causes hot corrosion of heat transfer surfaces. Peat can be used for co-combustion with wood at a ratio of about 10 to 20%. If these combustible wood materials mainly consist of logging residues, a higher proportion (30 to 40%) of peat is recommended (Orjala *et al.* 2004). Peat, coal, and municipal wastewater sludge are suitable for co-combustion with biomass, because they help prevent corrosion (Kassman *et al.* 2011). The protective effect of peat is primarily due to sulfur and, to some extent, aluminosilicates – both can remove alkaline chlorides (mainly KCl) (Aho 2012):

$$SO_3 + 2\ KCl + H_2O \rightarrow K_2SO_4 + 2\ HCl \tag{1}$$

$$Al_2O_3 * 2\ SiO_2 + 2\ KCl + H_2O \rightarrow K_2O * Al_2O_3 * 2\ SiO_2 + 2\ HCl \tag{2}$$

Lundholm *et al.* (2005) demonstrated that all peat fuels prevented agglomeration in the studied range of 760 to 1020 °C, and even with 5% peat fuel, significant effects were observed. The results also indicated that the mechanism for preventing agglomeration of bed material varies between different peat fuels. Possible mechanisms include peat minerals trapping alkalis, *e.g.*, calcium raising the melting temperature, and sulfur reacting with alkali metals to prevent agglomeration by increasing the melting point and reducing viscosity.

Peat ash contains plant nutrients, *e.g.*, phosphorus, making it a potential fertilizer in forestry. For example, previously, Finland's energy production generated 300,000 to 400,000 tonnes of peat and mixed peat-wood ash annually (Moilanen *et al.* 2012). However, peat energy use in Finland has halved from about 16 terawatt h (TWh) in the last decade, also reducing peat ash production (Bioenergia ry 2024). Compared to wood ash, nutrient concentrations and liming capacity of peat ash are low; nitrogen is mostly lost during combustion and nearly absent in well-burned ash (Hytönen 1998).

Fischer-Tropsch (F-T) diesel is produced from the syngas generated by peat gasification using the F-T synthesis to produce hydrocarbons for combustion. The production chain involves several stages that must be considered: peat extraction, production, transportation, gasification, refining into F-T diesel, storage and distribution, and conversion into mechanical work (Kirkinen *et al.* 2007b).

In the F-T process, which is a catalytic chemical reaction, gasification-derived syngas (containing carbon monoxide (CO) and hydrogen (H_2)) is converted into various hydrocarbons with different molecular weights, according to the following equation (Darmawan and Aziz 2022).

$$(2n+1)\ H_2 + n\ CO \rightarrow C_n H_{(2n+2)} + n\ H_2O \tag{3}$$

In addition to F-T diesel production, peat can be chemically or thermochemically refined into various other products (Arvola 2015). A conceptual diagram of various known chemical engineering processes that can be used to refine peat into new products is presented in Fig. 4.

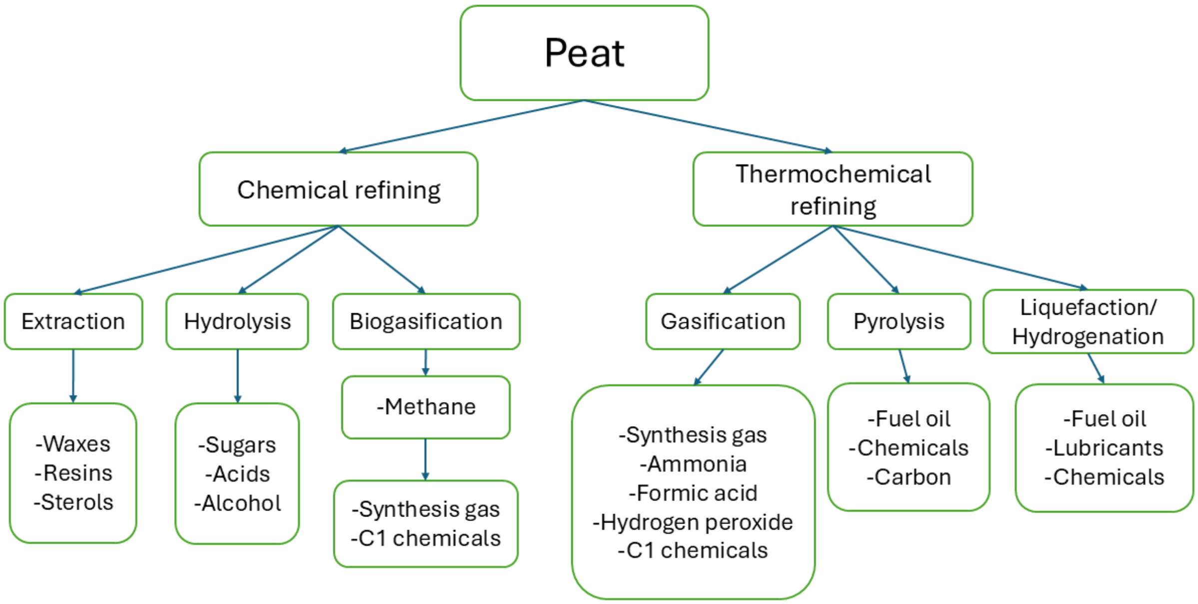

Fig. 4. Various chemical processes for refining peat into new products. Adapted from Arvola (2015)

GREENHOUSE EFFECTS

Combustion of peat produces carbon dioxide. Along with CO_2, peat production and harvesting also produces methane and other greenhouse gases (GHGs). On the other hand, in the course of photosynthesis, plants produce sugar and oxygen from water and carbon dioxide, using sunlight for carbon-capturing from the atmosphere. The carbon-compounds formed this way are mostly stored in the soil. When discussing peat utilization, it should be taken into account that peat's biodegradation releases methane on-site (Kirkinen *et al.* 2007[a]). This biodegradation is anaerobic, as discussed earlier. The production of biofuels can also alter the ecosystem's long-term carbon storage, thus impacting the carbon footprint of energy production (Kirkinen *et al.* 2008). The extraction of peat for combustion affects GHG emissions from peatlands, especially methane emissions (Kirkinen *et al.* 2007[a]).

Recent studies show that approximately 1100 to 1500 Pg of organic carbon is contained in the northern permafrost regions, of which about 500 Pg is seasonally thawed, while approximately 800 Pg remains permanently frozen (Hugelius *et al.* 2014). Around 20 to 60% of the organic carbon in permafrost is stored in peat (Schuur *et al.* 2008). In the laboratory incubation studies of Turetsky (2004), it was found that peat from thawing permafrost regions emitted substantially more GHGs than peat from frozen permafrost under similar conditions. This indicated that permafrost-driven changes in peat quality have an impact on its decomposition rate (Turetsky 2004). Approximately 15% of peatlands have been drained and deforested worldwide, with the main reason being commercial agricultural use, resulting in 1.3 Gt CO_2/yr emissions, not considering the significant emissions originating from peat fires (Mishra *et al.* 2016). However, a general estimate of global annual anthropogenic CO_2 emissions is 40 Gt. Turetsky (2004) suggested that permafrost patterning (including the thaw of permafrost decades or centuries in the past) has important influences on soil chemistry, with feedbacks to ecosystem processes such as decomposition.

As a consequence of climate change, a potential future risk in need of more discussion and research is the increasing vulnerability of some peat bogs to fires, which will be hard to put out. Such fires ignite more easily than flaming combustion. They are dominated by smoldering combustion, which may also persist in wet conditions. Climate change and human-induced drying are lowering the water table in peatlands and increasing the frequency and severity of peat fires. Burning of deep peat layers affects older soil carbon that has not participated in the active carbon cycle for centuries or even millennia, and thus determines the importance of peat fire-induced emissions to the carbon cycle and climate feedbacks (Turetsky *et al.* 2015). Lin *et al.* (2021) estimated that at a boreal region warming rate of 0.44°C/decade, the amount of carbon lost due to boreal peat fires in warmer soil could increase from 143 Mt (in 2015) to 544 Mt (in 2100), reaching a total of 28 Gt in the next century.

Peatland Climate Modelling

Peatlands have played an important role in the GHG composition in the atmosphere for most of the Holocene, leading to an estimated net cooling of ~0.5 W/m^2 (Frolking and Roulet 2007). Northern peatlands began developing ~16.5 thousand years ago, expanded rapidly between 12 and 8 thousand years ago due to high summer sunlight and rising temperatures, and contributed to early Holocene CH_4 peaks and modest CO_2 declines. They also likely influenced CH_4 and CO_2 fluctuations during earlier warm and cool periods (MacDonald *et al.* 2006). Even though peatlands have likely influenced Holocene climate, their role has only recently been included in climate assessments (Frolking *et al.* 2010). Earlier on, only a few studies had incorporated peatland carbon sinks into Holocene climate models, with limited or no climate-peatland feedbacks. However, many recent studies (Zhuang *et al.* 2020; Zhao and Zhuang 2023; Zhu *et al.* 2025) have added these feedbacks into the models applied in them.

Current policies make overshooting (climate change pathways where global temperatures temporarily exceed a target) the 1.5 °C temperature goal of the Paris agreement likely. Since northern peatlands are vast carbon storages and are warming faster than the global average, there is a risk for them to accelerate climate change by releasing more carbon into the atmosphere. Global warming causes peatlands' net carbon uptake to increase, but higher methane emissions largely offset this. Peatlands have been found to decrease the remaining carbon budget by 40 $GtCO_2$ (16 to 60 $GtCO_2$), or 8.6%, if the 1.5°C temperature goal is exceeded. Thus, this emphasizes the importance of better incorporating peatlands into climate models (particularly those concerning overshoot scenarios) to assist improving future political decisions (Zhu *et al.* 2025).

The future of northern peatlands as carbon sinks is uncertain. In a recent study (Zhao and Zhuang 2023), northern peatlands were predicted to turn from sinks to sources around 2050, sooner than previously estimated (after year 2100), highlighting their vulnerability to climate change. Furthermore, according to Zhuang *et al.* (2020), permafrost degradation and other disturbances may increase peat decomposition and alter peatland areas, complicating sink–source assessments, despite earlier research indicating northern peatlands will remain carbon sinks this century. On the whole, the models applied in these studies need to be developed further to reduce the uncertainties related to the question of peatlands function as carbon sinks.

Peatland Restoration and Management for Greenhouse Gas Mitigation

Northern peatlands are a major but highly variable source of atmospheric methane (CH_4), and both management and restoration strongly affect CH_4 exchange with the atmosphere. A systematic review and meta-analysis (87 studies, 186 sites across northern peatlands) showed that CH_4 emissions are mainly controlled by water table depth, plant community composition, and soil pH, with the highest emissions occurring in fens. Natural northern peatlands were found to emit CH_4 highly variably, with a 95% confidence interval of 7.6 to 15.7 g C m^{-2} $year^{-1}$ for the mean and 3.3 to 6.3 g C m^{-2} $year^{-1}$ for the median. The overall annual average was found to be (mean ± standard deviation) 12 ± 21 g C m^{-2} $year^{-1}$. Temperature alone is a weak predictor, but its interaction with hydrology, vegetation, and soil properties is critical. Drainage significantly reduces CH_4 emissions (on average by 84%), whereas restoration through rewetting or vegetation recovery increases CH_4 emissions by about 46% compared to pre-management levels. To assess the overall climate impact of peatland management, both net ecosystem exchange and carbon exports must be considered (Abdalla *et al.* 2016).

A search conducted by Haddaway *et al.* (2014) identified over 26,000 articles, and screening of available full texts yielded 93 relevant articles (110 independent studies). 39 studies were excluded from the critical review, leaving 71 for synthesis. The results show that in boreo-temperate lowland peatland systems drainage increases N_2O emissions and ecosystem CO_2 respiration, but reduces CH_4 emissions. Second, naturally drier peatlands emit more N_2O than wetter ones. Finally, restoration was found to increase CH_4 emissions.

A review paper by Kumar *et al.* (2020) discussed fundamentals of mitigating the major GHG emissions (CO_2, CH_4, and N_2O) from tropical peatlands in Southeast Asia and their impact on global climate change. Also, in their study it was found that rewetting peatland may increase CH_4 emissions, however stating that more research is needed to determine whether peatlands act as GHG net sinks or net sources. Only a few studies were found to have examined the effectiveness of liming in reducing peat soil acidity. Furthermore, there is a shortage of data on CO_2 concentrations in drainage and forest fire areas, N_2O fluxes in agricultural areas, as well as the contribution and reduction of CH_4 in tropical peatlands. It was concluded that these topics warrant further research to help develop a framework for GHG emission measurements and mitigation in tropical peatland regions.

Waddington and Day (2007) observed that seasonal CH_4 emissions did not differ before and immediately after peatland restoration. However, three years after restoration, seasonal CH_4 emissions at the restored site were 4.6 times higher than at the peatlands prior to restoration. Ponds and ditches at the restored site were seasonal hotspots for CH_4 emissions. However, the emissions from grass vegetation were the dominant source of CH_4 from the restored peatland due to its large area. The CH_4 fluxes from the studied peatland represented 14% of the area's CO_2 equivalent losses. This study highlights the significance of vegetation succession in the CH_4 flux from peatlands.

Fertilizing low-yield wetlands can mitigate climate change over decades. The productivity gains from fertilization lead to increased forest CO_2 sinks, which significantly outweigh soil CO_2 net emissions. Previous research also has not found short-term rise in GHG emissions from soils in drained peatlands under forestry (Ojanen *et al.* 2019).

Decommissioned peatlands could be used for forestry, serving as effective carbon sinks. GHG emissions can also be diminished after production by cultivating plants like hemp on peatlands, which sequester carbon throughout the growing season. In Fig. 5, ongoing small-scale cultivation testing in former peatland in Halsua, Finland, is presented. Hemp's growing season lasts from spring to late autumn, allowing it to be harvested and take full advantage of the entire thawed ground period. Crop rotation with hemp improves soil health and reduces pests. This is an interesting topic, demanding more research, especially related to its sosio-economic impacts as well as its application on varying locations globally.

Fig. 5. Cultivation of hemp in a former peatland (photo taken in October 2024).

Paludiculture for Greenhouse Gas Mitigation

Paludiculture, which means wetland farming where crops are grown on rewetted peatlands, is a rather new idea related to climate warming countermeasures and has recently gained global interest. According to Tan *et al.* (2021), the long-term effects of paludiculture should be carbon-neutral or carbon-negative. They also noted that paludiculture development is based on northern peatland research, and the subject has scarcely been researched in tropical peatlands. Thus, more research on tropical paludiculture is warranted.

Native species should be selected as vegetation sources for paludiculture (Tan *et al.* 2021). Various crops can be cultivated. For example, in a Latvian study (Ozola *et al.* 2023), paludiculture of Sphagnum spp., black alder (*Alnus glutinosa*), common reed (*Phragmites australis*), reed canary grass (*Phalaris arundinacea*), cattail (*Typha latifolia/angustifolia*), and sweet flag (Acorus calamus) were assessed and found to have promising results for upcoming large-scale implementation by private enterprises. As noted by Gaudig *et al.* (2014), large-scale implementation of Sphagnum farming requires extensive know-how, from initial species selection up to the final production as well as using growing media derived from Sphagnum biomass in horticulture.

An Indonesian study (Uda *et al.* 2020) found that among the studied crops, sago (*Metroxylon sagu*), banana (*Musa paradisiaca*), pineapple (*Ananas comosus*), water spinach/kangkong (*Ipomoea aquatica*), kelakai/edible fern (*Stenochlaena palustris*), illipe nut/tengkawang (*Shorea* spp.), dragon fruit (*Hylocereus undatus*), mangosteen (*Garcinia mangostana*), and sweet melon/melon (*Cucumis melo*) were found as the most suitable alternatives for local paludiculture. However, it was stated that precaution is necessary when planting crops requiring low drainage.

Myllyviita *et al*. (2024) studied climate change mitigation potential of paludiculture in Finland. They found that rewetting reduces emissions from drained peatlands, and paludiculture can provide renewable raw materials as an alternative to peat. Paludicrops were used instead of peat for animal bedding and growing media. Results indicated that compared to current peat use, paludiculture could save 352,000 t CO_2-eq by 2050, mainly through reduced land-use emissions. Most of the paludiculture emissions came from crop cultivation (300,000 t CO_2-eq), with a carbon sink of 48,000 t CO_2-eq. It was suggested that paludiculture is unlikely to exceed peat-related emissions but that it does not fully offset abandoned cropland emissions, and that afforestation or restoration could yield greater savings.

A four-year experiment conducted in Finland, in which the groundwater level was raised gradually and CO_2, CH_4, and N_2O (nitrous oxide) emissions were measured. The results showed, among other things, that a 10 cm increase in water level reduced CO_2 emissions by ~0.87 Mg CO_2-C ha^{-1} per year. CH_4 fluxes varied from uptake to emission as the water table rose. Nitrous oxide emissions ranged between 2 to 33 kg N_2O-N ha^{-1} yr^{-1}; emissions were initially high on bare soil but declined by the end of the experiment, likely due to more plant cover and higher water table limiting aerobic soil conditions. Overall, the area remained a net carbon source, highlighting the emission-related challenges of highly decomposed peatlands (Lång *et al.* 2024).

In Denmark, Nielsen *et al.* (2024) found that in nutrient-rich peatlands, paludicultivation can be a carbon sink right from the start, but in nutrient-poor bogs the result was the opposite. Since the results on paludiculture for GHG mitigation still seem to be variable and unclear, more research is needed on the subject. Furthermore, the effect of paludiculture on peat water quality should be studied in future, also in conjunction with novel peat water treatment technologies.

REHABILITATION OF DECOMMISSIONED PEATLANDS

Decommissioned peatlands may be rehabilitated (or restored) and used for various purposes as well as to mitigate their detrimental environmental impacts, including reducing GHG production and biodiversity promotion, or for paludiculture. Furthermore, peatland rehabilitation may help to restore ecosystems and improve water quality, leading to *e.g.* decreases in soil erosion. Soil amendments used for the reclamation of decommissioned peatlands generally aim to improve soil structure and optimize nutrient availability. However, these areas, where peat extraction has ceased, can be challenging for farming due to poor aeration, compaction, and weak nutrient cycling (Kuokkanen *et al.* 2019). Peat itself can also be used for soil improvement and cultivation.

Soil Improvers for Peatland Rehabilitation

Certain soil properties can be influenced by adding various soil amendments. These properties include water retention, aeration, temperature, nutrient retention and availability, cation exchange capacity (CEC), structure and pore stability, as well as microorganisms, insects, and pests (Shinde *et al.* 2019). Biochar, made from various organic materials can improve soil structure by enhancing aeration, water retention, and nutrient availability. It can also have a variable effect on the GHGs emitted by the soil (He *et al.* 2017). Compost and other sources of organics, *e.g.* manure compost, can be used to amend peat soils, resulting in an improved soil structure and increased organic matter content, acting as an enhanced basis for microbial activity as well as promoting nutrient cycling. Sand or clay can also be added to decommissioned peatlands to improve water-physical properties of the peat soil, with sand enhancing, *e.g.*, aeration and nutrient retention, and clay improving the growing media rewetting ability (Michel 2009; Zakharova *et al.* 2020).

Tailored peatland fertilizers are beneficial for peatland reclamation, since they contain essential nutrients such as nitrogen (N), phosphorus (P), and potassium (K). Thus, soil growing-conditions will improve, supporting plant growth. Adding organic matter and microorganisms can enhance biological activity and promote nutrient recycling (Deru *et al.* 2023). Schulte and Kelling (1998) studied various soil amendments, including crop residue, manure, compost, peat, sawdust, sewage sludge, green manure, cover crops, and topsoil. Organic matter content of the soil was increased by all of the soil amendments studied. Materials with low C:N ratio (< 28:1) release nitrogen upon decomposition, while materials with high C:N ratio bind nitrogen. Plants can utilize nutrients once they are converted into inorganic forms. Additionally, soil amendments influence the soil microbial balance. Babla *et al.* (2022) investigated the addition of carbon to soil mixed with organic materials and found that soil amendments made from waste materials can be pelletized, allowing organic matter and nutrients such as N and K to slowly dissolve into the soil. Therefore, the physical, chemical, and biological characteristics of the soil are improved, contributing to sustainable agriculture.

A new innovation, presented in Fig. 6, suggests utilizing valuable biomasses from industrial side streams as fertilizers and soil improvers on peatlands converted to carbon sinks. Kuokkanen *et al.* (2019) studied the use of soil amendments in non-food potato production.

Industrial side streams and by-products can serve multiple purposes in non-food potato production. However, the common practice of monoculture in potato cultivation creates significant challenges, such as soil compaction, depletion of humic substances, and a decline in the soil's nutrient retention capacity. These changes, in turn, lead to increased nutrient leaching, which not only reduces nutrient availability for crops but also results in economic losses and imposes additional burdens on the environment (Hansen 1996). Environmental impacts of peatland non-food potato cultivation can be minimized by using suitable soil amendments, which offer a promising solution due to their chemical purity. Soil particle bonds help plants access water and nutrients, with medium-sized pores retaining water and larger pores allowing root growth.

Coarse soils with large pores are preferred for potato cultivation, and adding biotechnologically modified fiber sludge enriched with nutrients can improve soil health and act as fertilizer (Kuokkanen *et al.* 2019). Biotechnologically modified fiber sludge decomposes at a slow rate and serves as a substrate (Kuokkanen *et al.* 2018). Previous studies have successfully tested five industrial by-products as soil amendments for decommissioned peatlands: fiber sludge, biocarbon, hygienic biodigestate, paper mill sludge and gypsum waste (Kuokkanen *et al.* 2019).

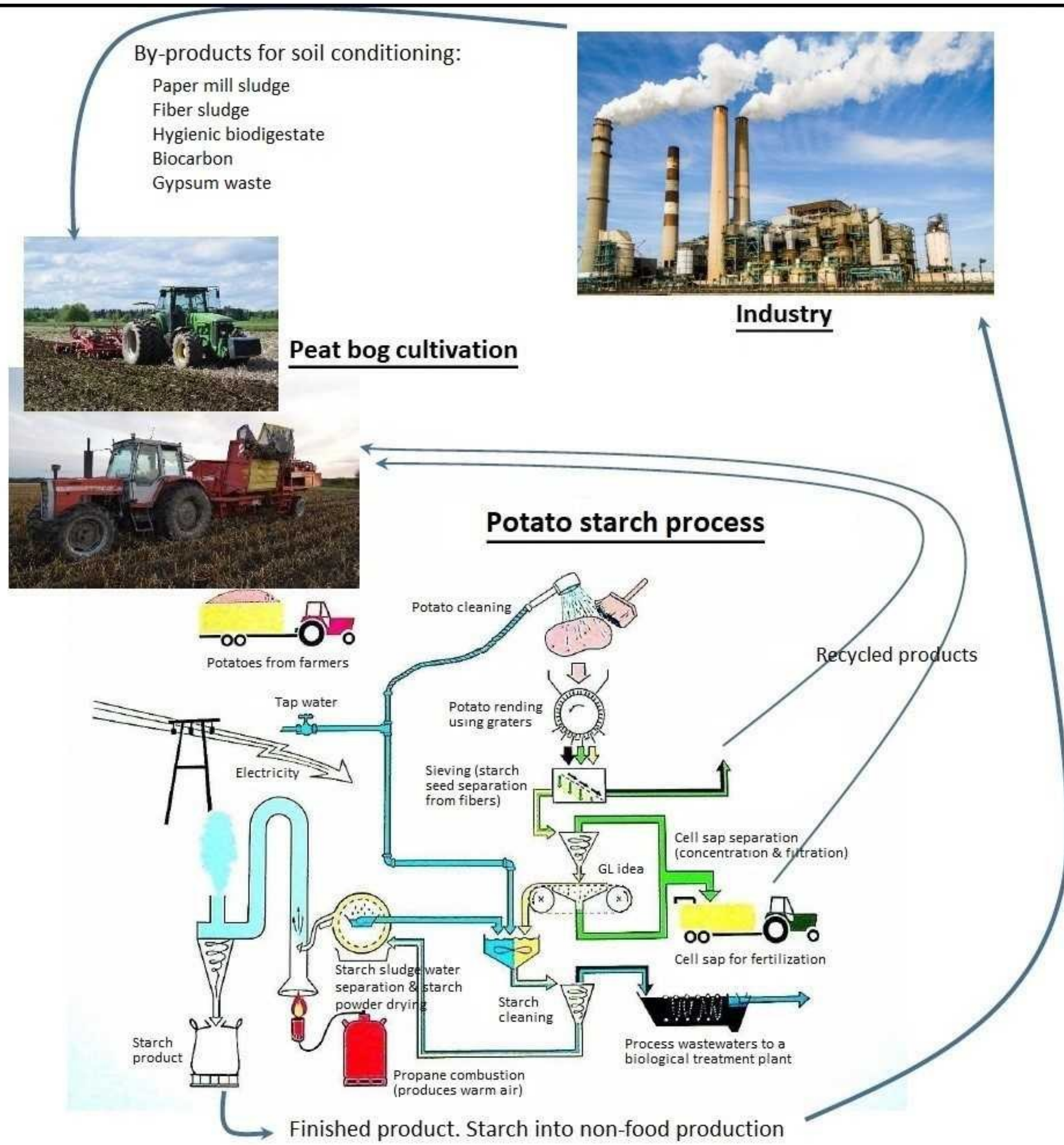

Fig. 6. The circular economy cycle of non-food potato production in decommissioned peatlands (Kuokkanen *et al.* 2019)

In the future, it would be valuable to further investigate the long-term functionality of the soil amendments described above. For example, gypsum waste takes many years to fully dissolve in the soil, whereas the properties of fiber sludge can be enhanced through enzymatic treatment and should be tested under a variety of soil conditions. In such studies, economic aspects should also be scrutinized in detail, since this kind of analysis is still lacking.

Stabilizing peatlands for construction or road building is crucial and has been highlighted in recent studies. For instance, Affam *et al.* (2023) studied adding of sodium, calcium and zinc compounds along with biochar to peat soil to increase its strength and stabilize the soil, finding them effective. Similarly, Vincevica-Gaile *et al.* (2021) suggested that the addition of limestone could improve peatland stabilization, and they also proposed the use of ash from various sources. Untreated waste, products made from recycled materials, and waste from biological treatment of municipal waste, landfills, and industrial processes can also be utilized as raw materials. Additionally, composite materials are useful for soil reinforcement, especially for road or building construction. Bamboo has also been studied for peatland improvement for similar purposes (Talib *et al.* 2021), while Jumien *et al.* (2023) examined the use of cement and quarry dust as additives, finding them effective as well. Altogether, more research is needed on the subject areas presented in this chapter.

Costs of Peatland Restoration Projects

It is important to assess peatland conditions before and after restoration projects to evaluate both economic and ecological outcomes. Such studies are lacking (although on the rise), especially concerning long-term effects/results, and should be carried out more in the future. The models used in these studies should be developed further. According to an earlier paper on the subject (Moxey and Moran 2014), capital costs of peat land restoration were assumed to fall in an illustrative range of 250 to 12,500 €/ha (indicated in the literature). The large variation in estimates is due to the differences in the prices of restoration methods and the measures that have to be resorted to. This is influenced not only by the methods used, but also by the site (*e.g.* remoteness) to be restored and the objectives of the project. Some projects have clearly cheaper solutions, for example, only blocking grip drains. Other projects have to resort to more expensive solutions, such as vegetation restoration and, in addition, lost agricultural income.

In a Finnish study (Rehell *et al.* 2014), restoration measures of a wet, swampy mire were found to cost about 800 €/ha on the treated area, but when impacts over the wider affected area are considered, the cost dropped to roughly 90 €/ha. These calculations excluded planning and supervision costs. According to a study reviewing peatland restoration costs in Lithuania and other countries, restoring 1 ha of peatlands in Lithuania (including works such as dam installation, vegetation removal, and drainage destruction) costs about 800 €/ha, while in some countries, such as Germany, costs may reach 3,000 €/ha (Greifswald Mire Centre 2020).

Glenk *et al.* (2022) analysed a database of peatland restoration activities from 142 projects and 323 sites in Scotland, of which 300 were suitable for analysis. Data came from two types of forms completed by applicants and grantees: application forms and final reporting forms. Restoration costs varied widely by activity type and initial peatland condition. Based on reported actual costs, the mean restoration cost was 2000 €/ha (median 1200 €); excluding outliers, the mean decreased to 1400 €/ha. On average, project management accounted for about 10% and non-monetary contributions for about 8% of total costs. Forest-to-bog restoration roughly doubled per-hectare costs compared to other activities.

Use of Peat for Soil Improvement and Cultivation

Peat itself can be used for soil improvement. It is important in commercial horticulture, as it supports plant growth, helping plants to trap CO_2 from ambient air and return it to the soil. The carbon is then stored to the soil due to its high content of organic matter. The part of peat used for soil improvement and as a growing medium comes from the surface layers of peatlands, not from the deeper layers suitable for combustion. This material can be used for soil amendment, as a sorbent or bedding material or as compost (Järvinen and Hänninen 1992). The use of peat as a growing medium is being replaced. Several organic materials, such as wood bark, wood fibers, coconut fiber, other coconut-based products, and green compost, are suitable alternatives to peat (Gruda *et al.* 2024).

Ma *et al.* (2022) demonstrated that a BHA (bentonite-humic acid) addition improved and better-preserved soil ecosystem balance and agricultural crop yield. This was reached by effective soil hydrological regulation, soil enzyme functioning, as well as nutrient exchange in surface and subsoil, leading to efficient water and nutrients usage by oat crops, and a corresponding increase in grain protein, crop yield, efficiency of water use, and overall productivity response per unit of nitrogen applied (nitrogen partial factor productivity). Liu *et al.* (2022) investigated the effect of peat and bentonite additions on substrate-less remediation in heavy metal-contaminated areas. They found that the additions improved plant growth and reduced the bioavailability of heavy metals. Cao (2019) noted in his study that adding peat significantly altered the physico-chemical properties of sandy soil.

Peat is uniform in quality, porous, and resistant to compaction, making it a durable substrate, especially in dry soils. Compost, though variable in composition and prone to compaction, is more nutrient-rich and biologically active. Peat has low nutrient content but enhances soil CEC and suits acidophilic plants, whereas compost is generally neutral to slightly alkaline. Peat is difficult to re-wet once dry and typically free of weed seeds; well-processed compost can be similar. Compost is commonly used as a top dressing, while peat is less ideal due to surface drying and moisture absorption. Despite its lower cost, peat use raises environmental concerns due to its slow regeneration, carbon emissions, and peatland degradation, potentially delaying the shift to sustainable alternatives (Government of Ireland 2019).

TREATMENT OF PEAT BOG DRAINAGE WATER

Regarding negative aspects of peat production, peat bog drainage water (PBDW) generation is one of the main issues mentioned due to its potentially adverse environmental effects. The humic substances (HS) content in water bodies varies seasonally and annually, mainly influenced by rainfall. In one study (Tuukkanen *et al.* 2017), water quality data were gathered and analyzed from 15 peat extraction sites in Finland (located in different parts of the country) during years 2011 to 2012 (336 total samples were taken). Table 2 was compiled based on the data presented in the article.

Table 2. Peat Extraction Runoff Water Quality Data Variation

Parameter	Range of variation	Average	Median
COD_{Mn}	10–170 [mg/L]	45 [mg/L]	40 [mg/L]
P_{tot}	15–840 [µg/L]	96 [µg/L]	65 [µg/L]
N_{tot}	0.5–3.6 [mg/L]	1.6 [mg/L]	1.4 [mg/L]

The fluctuation in PBDW pollutant concentrations and load has been associated with the intensity of drainage, soil geochemical properties, and the rate of runoff. Due to the wide variations in PBDW quality, selection of suitable water treatment methods is challenging. Thus, different treatment methods designed to comply with the requirements of the peat production process, local hydrology and geology, the sensitivity of the receiving water bodies and current legal standards are necessary at different sites. This is the situation at least in Finland, where regulatory authorities determine purification requirements for each case separately (Heiderscheidt 2016). PBDW is derived from extraction of peat and is typically mildly acidic and colored. It contains the nutrients phosphorus (P) and nitrogen (N), in addition to HS and total solids (TS) (Kuokkanen *et al.* 2015). Excessive nutrient inputs into aquatic systems cause oversaturation, triggering algal blooms, which in turn leads to eutrophication (Zhao and Sengupta 1998; Bektaş *et al.* 2004; Alvarez-Vázquez *et al.* 2014). This will lead to the oxygen level of water being depleted and water light penetration being hindered, thus being detrimental to the organisms present in the aquatic environment as well as causing biodiversity decrease (Bektaş *et al.* 2004; Alvarez-Vazquez *et al.* 2014). HS are residues of biological substances (by microbial activity), of which the organics contained in natural waters are mostly comprised of (Jones and Bryan 1998; Seida and Nakano 2000; Yildiz *et al.* 2007; Ghernaout *et al.* 2009).

The main components of HS in water are humic acids (HA), fulvic acids and humins. HA are weakly acidic aliphatic and aromatic compounds, which contain various functional groups (*e.g.*, -COOH and phenolic -OH groups), and their presence in natural water causes the water color to become darker (O’Melia *et al.* 1999; Motheo and Pinhedo 2000; Prado and Airoldi 2003; Seredynska-Sobecka *et al.* 2006; Naddeo *et al.* 2007).

In addition to esthetic problems, this will further prevent light from reaching deeper parts of the water body. Overall, HS are non-homogenous and have very complex chemical structures and a high molecular weight (several hundreds g/mol or larger), and their physicochemical properties are not clearly defined (Hesse *et al.* 1999; Motheo and Pinhedo 2000). Because of their high stability, HS are also resistant to microbial attack (Motheo and Pinhedo 2000). HS are the end-product of nature’s biodegradative and oxidative processes, making them essentially non-biodegradable; however, their high aromatic and aliphatic residue content allows them to be readily aggregated and precipitated by charge neutralization (Jones and Bryan 1998; Ødegaard et al. 1999; Yildiz *et al.* 2007; Yildiz *et al.* 2008). Water containing HS may also be treated using other potential methods, such as physicochemical, biological, and membrane processes, *etc*. Usage of peatland buffer areas and/or wetlands are examples of classical methods that are of particular interest in HS removal. Traditional PBDW treatment methods have often limited effectiveness. Therefore, new methods are needed. For example, electrocoagulation (EC) has also been suggested to be an effective new method to treat PBDW (Kuokkanen *et al.* 2015; Postila 2016). While theoretically possible, capturing methane from peat water would require precise control and closed systems, which is currently impractical. In addition to treatment technologies, spatial planning tools have also been developed to support water protection in peatland forestry. Niemi *et al.* (2022) introduced a GIS-based method for identifying suitable locations for water protection structures, such as overland flow fields and uncleaned ditch sections. Using high-resolution lidar data, the method enabled detailed terrain analysis and was tested in the Kovesjärvi catchment area (Parkano, Western Finland). Field evaluations confirmed most suggested sites as suitable, highlighting the method’s applicability in forest management planning.

Basic and Enhanced Methods

Numerous water purification methods have been developed or modified for use in the conditions found at peat extraction sites, aiming to reduce the load of pollutants caused by this type of land use (Kløve 2001; Heiderscheidt 2016).

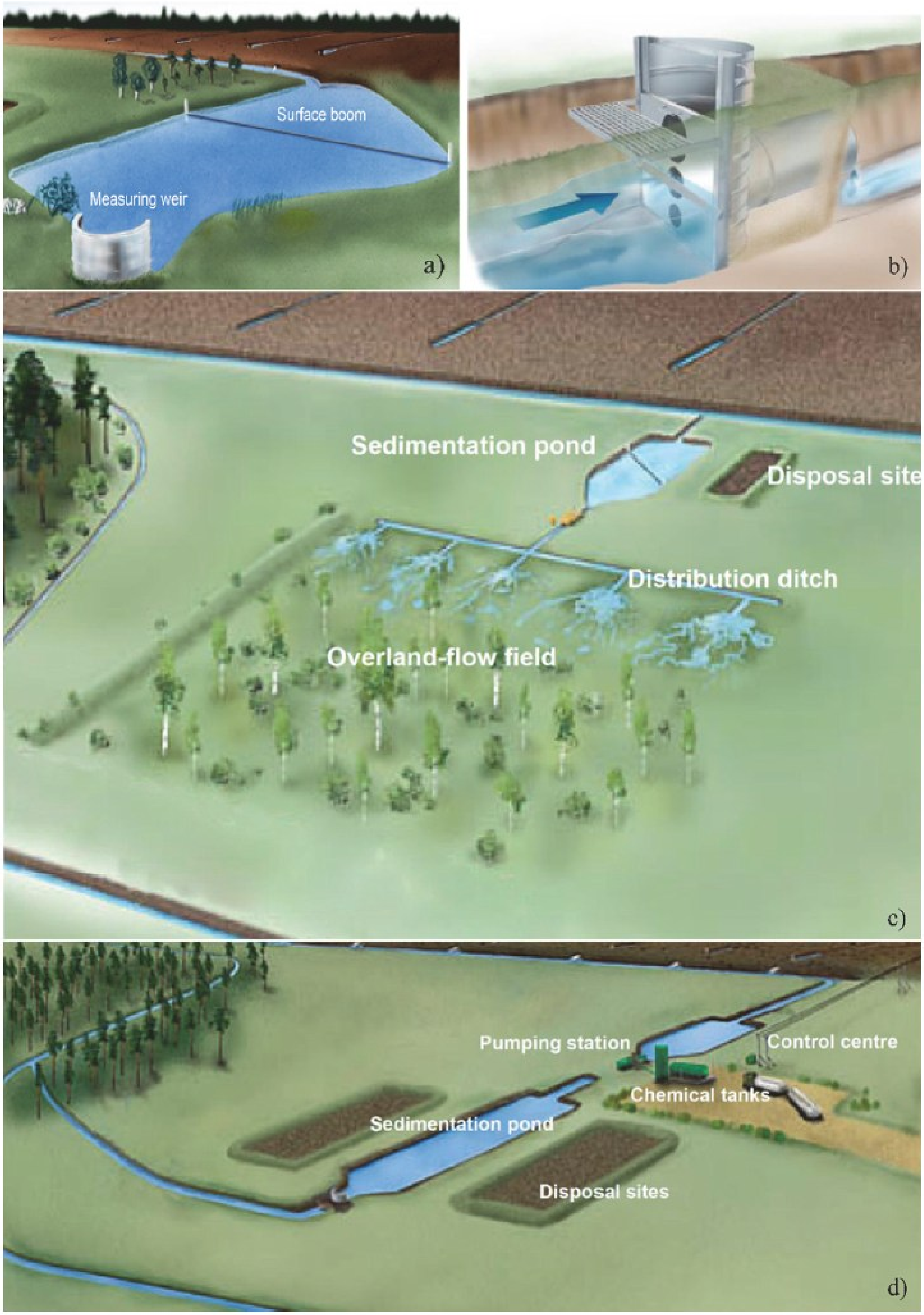

Fig. 7. The most commonly used water treatment methods for peat extraction areas in Finland. a) Constructed sedimentation pond, b) peak runoff control dam, c) overland flow field and d) chemical purification (Heiderscheidt 2016)

For example, the primary basic and enhanced water treatment approaches applied at Finnish peat extraction sites are presented in Fig. 7. In these methods, the flow of PBDW is generally directed by existing drainage networks. Structures for load retention are then placed within the ditch network or at its outlet. To this end, a main ditch or alternatively a channel collecting the outflow may be used (Heiderscheidt 2016; Postila 2016).

As reported by Marttila and Kløve (2009), peak runoff control dams are able to reduce 61 to 94% of suspended solids (SS), 45 to 91% of total nitrogen (N_{tot}), and 47 to 88% of the P_{tot} present in PBDW.

An overland flow field (Fig. 7c), such as treatment wetlands or constructed wetlands, is considered to be among best available techniques for treating runoff water from peat extraction areas, and due to its low maintenance requirement, it is the most widely used method to treat PBDW. In this approach, PBDW is directed into the surface layer of a peatland area, which is either pristine or has been previously drained. The vegetation of the surface layer then separates solids from the water, acting as a mechanical filter. In addition to physical (sedimentation and filtration) removal of pollutants, chemical and biological (especially nitrification and denitrification) processes in the peat layer contributes to the removal (dissolved nutrients) efficiencies. Typically, 60% of SS is removed, while removal efficiencies of 25% are reached for N_{tot} and P_{tot} (Heiderscheidt 2016; Postila 2016).

Treatment wetland purification efficiency if affected by several environmental and soil characteristics, which include, *e.g.*, residence time (Wörman and Kronnäs 2005), soil properties (Pant and Reddy 2003; Liikanen *et al.* 2004;), climate (Kadlec 1999; Werker *et al.* 2002; Kuschk *et al.* 2003), redox dynamics (Niedermeier and Robinson 2007), biological factors such as microbes and plants (Huttunen *et al.* 1996), pH (Grybos *et al.* 2009), and hydraulic load (Braskerud 2002). However, simplified, it is generally known that the TA/CA ratio (treatment wetland area as a proportion of total catchment area) determines the purification efficiency of an overland flow field. According to Postila (2016), in Finland, the recommended TA/CA is >3.8%, which can be seen to be a very large figure. A general rule of thumb of the required overland flow field TA/CA ratio is 1 to 5% (Guieysse 2018).

Chemical coagulation (CC) is a classic water/wastewater treatment method used at wastewater treatment plants all over the world. In CC (Fig. 7d), aluminum or iron (ferric) salts, such as aluminum sulfate (referred to as alum), ferric sulfate (the overall best option in this application (Heiderscheidt 2016)), and ferric chloride are typically used to promote coagulation and remove the pollutants from water/wastewater. Both natural and synthetic coagulant aids and enhanced coagulants may also be applied.

After coagulant chemicals are added and mixed, the water is routed toward a sedimentation pond, where solids and dissolved substances are precipitated. CC typically reduces SS by 30 to 90%, N_{tot} by 30 to 60%, and P_{tot} by 75 to 95%. It is used at sites with sensitive ecosystems or limited space for wetlands. However, high implementation and maintenance costs, along with the need for precise control, limit its use. The freezing of liquid coagulants and fluctuations in efficiency, pH, and metal concentrations are technical challenges (Heiderscheidt 2016).

For treating cold peat water, specialized coagulants, *e.g.* polyaluminum chloride (PACl) or polyaluminum ferric silicate (PSAF), could be used, since it has been shown that the water temperature does not significantly affect the removal efficiencies using these chemicals on natural waters (Guan *et al.* 2011, Bian *et al.* 2026). However, research related to this is scarce, particularly using peat water (no studies were found). Thus, research efforts are warranted.

Novel Methods

New methods for the treatment of PBDW have focused on enhancing traditional techniques and applying innovative approaches. These approaches offer promising alternatives alongside conventional methods, especially in areas with limited space or resources. Hybrid wetlands integrate different moisture conditions, including surface-flow and subsurface-flow wetlands, resulting in more effective nitrogen and organic matter removal. For example, Hernández-Rodríguez *et al.* (2022) used *Phragmites australis* to achieve up to 95% ammonium (NH_4^+) and 80% COD removals (from domestic wastewater) with a short hydraulic retention time, while two conventional wetlands (one baffled and one non-baffled) with twice the footprint removed only 34 to 50% NH_4^+ and 54 to 73% COD. Although applicable to PBDW treatment, literature on hybrid wetlands for this purpose is lacking, highlighting the need for research.

A recent line of research in Finland focuses on developing hybrid wetland systems that incorporate microbial electrochemical technologies (METs) to improve treatment efficiency in cold climate conditions. These systems use electroconductive materials to support microbial processes, improving the efficiency of organic matter and nitrogen removal. Attention is also given to optimizing performance, selecting suitable materials, and minimizing GHG emissions. The concept offers a promising, nature-based solution for improving water quality in peatland areas.

Another innovative approach involves combining physicochemical treatment methods to improve the removal of organic matter from peat-influenced waters. A study by Elma *et al.* (2022) demonstrated the effectiveness of a hybrid system integrating coagulation, adsorption, and ultrafiltration processes. Coagulation with aluminum sulfate removed up to 78% of natural organic matter (NOM). Subsequent adsorption with powdered activated carbon increased the NOM removal efficiency to 92 to 96%. Final ultrafiltration using a polysulfone-based membrane achieved up to 95% NOM removal with high filtration fluxes. This integrated treatment notably reduced membrane fouling, offering a promising alternative for effective peat water purification.

The potential of using peat water as a source for potable water in peatland areas was highlighted in a recent study (Khayan *et al.* 2022). The research demonstrated that integrating various treatment methods, including aeration, sedimentation, as well as filtration with shell sand and activated carbon, can effectively condition peat water to meet clean water standards. The study showed significant improvements in water quality, with reductions in color (866.7 to 28.83 PtCo), turbidity (10.96 to 2.19 NTU), iron (Fe) levels (2.48 to 0.22 mg/L), and coliform bacteria (1068 to 20 CFU). The most effective stage of treatment was filtration and adsorption, providing a promising solution for water purification in peat-rich regions.

Hidayat *et al.* (2025) investigated the thermal activation of bentonite for treating peat water and found that activating bentonite at 500 °C effectively (200/300/400/500 °C were tested) reduced the Fe content by 66%, organic matter by 89.6%, and total dissolved solids (TDS) by 27 mg/l. The activation process enhanced the adsorptive capacity of bentonite, making it more efficient for removing contaminants from peat water. This method proved to be a promising solution for improving the quality of peat water for consumption. Various other novel single and combined methods for PBDW treatment have been summarized in a recent mini-review paper by Qadafi *et al.* (2023).

Electrocoagulation (EC)

A new method peat bog drainage water treatment, electrocoagulation (EC), was proposed by Kuokkanen *et al.* (2015). EC is a water treatment technology based on electrolysis and with over a century of development, currently gaining global commercial interest. In EC, sacrificial anodes typically made of Al or Fe are used, which dissolve into water (as Al^{3+} or Fe^{2+}), promoting coagulation. Simultaneously, a cathodic reaction generates microscopic hydrogen gas to aid flotation, removing the coagulated matter from the water as flocs onto the water surface. EC may be called a green technology, since the electron is the only (no chemicals are added, unless pH alteration or conductivity increase is needed) "chemical" used in it, leading to no secondary pollution occurring. The amount of coagulant metal dissolved is based on Faraday's law, and is thus proportional to the electric current and treatment time, allowing easy dosing. EC can be operated in batch or continuous modes, is automated, and requires minimal maintenance (Chen 2004; Kuokkanen *et al.* 2013; Kuokkanen 2016).

In Kuokkanen *et al.* (2015), optimized batch-EC treatment of synthetic wastewater removed 90% COD_{Mn} and 80% dissolved organic carbon (DOC). When applied to real PBDWs (Fig. 8) from northern Finnish peat bogs, high pollutant removal was achieved, with complete removal of P_{tot}, TS, and color, and 33 to 41% removal of N_{tot}, 75 to 90% of COD_{Mn}, and 62 to 75% of DOC/TOC (total organic carbon).

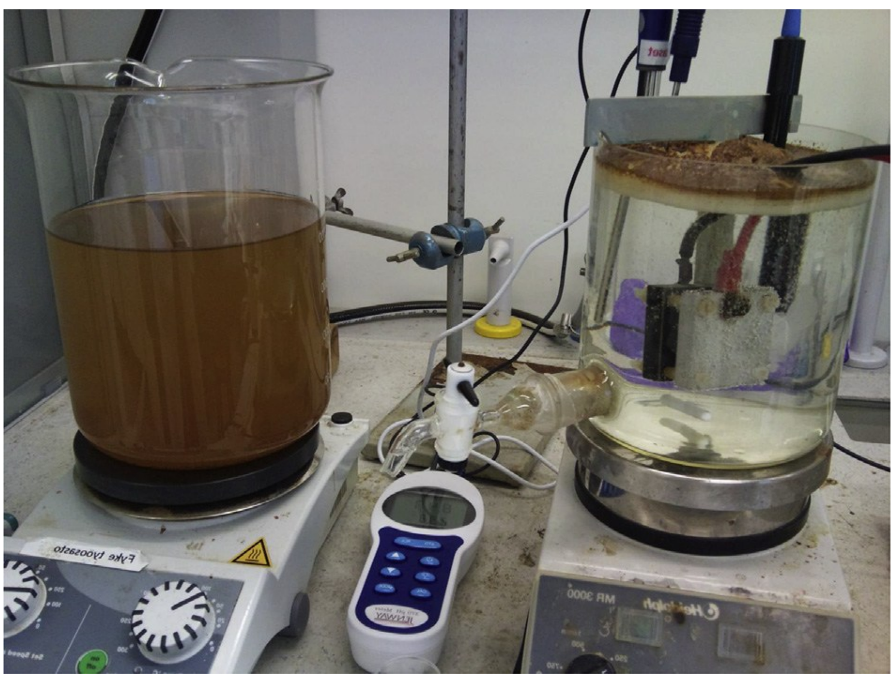

Fig. 8. PBDW from a Finnish peat bog before and after batch-EC treatment (Kuokkanen *et al.* 2015)

EC is suitable for treating cold water, unlike CC, which is less effective in cold conditions, especially with solid coagulants (Heiderscheidt 2016). Kuokkanen *et al.* (2015) found that EC also neutralizes PBDWs by producing OH^- at the cathode, whereas CC tends to lower pH. EC has low energy requirements (per cubic meter treated), making solar power viable for PBDW treatment, and can be operated remotely, which is advantageous for peat extraction sites without grid electricity. EC also has strong disinfectant properties, inactivating bacteria and microbes in water and it offers logistical advantages, as it uses metal plates instead of chemical products with variable metal content. However, EC requires sufficient water conductivity, which can be enhanced by adding NaCl. The challenge with EC is its application to high-volume flows at remote sites such as peat extraction sites, but it is particularly promising for smaller-scale cases.

In addition to Kuokkanen *et al.* (2015), a Malaysian research group has studied EC treatment of tropical peat water and published multiple papers on the subject (Rahman *et al.* 2020a,b, 2021a,b, 2024). Rahman *et al.* (2021c) developed a solar-powered EC system for the treatment of Sarawak (located in East Malaysia, in northwest Borneo) peat water. It was concluded that the batch and continuous EC systems studied could achieve 18.8% and 46.15% turbidity removal efficiencies, respectively. However, it was found that the designed systems still need improvements to meet stricter drinking water standards. Lately, the research group has focused on EC treatment of brackish peat water (intrusion of seawater into peat water, which raises water salinity) and the challenges of EC in treating this type of water, such as aluminium electrode fouling and passivation (Rahman *et al.* 2022, 2023, 2025a,b).

According to Rahman *et al.* (2020b), peat water, being an abundant water resource, is still used for domestic purposes in some coastal areas of Sarawak. They point out that if used for human consumption, HAs may cause diseases such as stomach cancer, blackfoot disease, *etc*. This highlights the importance and the possibilities of this research subject on a global scale.

When comparing COD values presented (49 to 64 mg/L) (Rahman *et al.* 2020a,b, 2021b), they were in accordance with the Finnish values presented in Table 2, although they had been measured as COD_{Cr} (which results in higher COD values than COD_{Mn}), and not as COD_{Mn}, which is typical for natural water bodies. Furthermore, as presented in the papers of Rahman *et al.* (2020a,b, 2021b,c), the following initial values for Sarawak peat waters were measured: turbidity 9 to 42.38 NTU, total suspended solids (TSS) 8 to 11 mg/L, TOC 48 to 99 mg/L, color 451 to 1080 TCU. Applying EC, mostly high removal percentages for these parameters were achieved, as presented in Table 2 of Rahman *et al.* (2021a).

Comparison of Peat Bog Drainage Water Treatment Methods

Various peat bog drainage water treatment methods have different advantages and limitations, space requirements, costs, *etc*. These have been compared in Table 3. A cost estimate based on forestry use in drained peatlands found that the marginal costs of using overland flow fields for water treatment is very low and sediment reduction costs were significantly lower than when using sedimentation ponds. However, data on the specific costs of phases involved in preparing overflow fields and sedimentation ponds remain limited and require further investigation (Miettinen *et al.* 2020).

As a whole, overland fields are well applicable in cold climate locations. However, as stated by Postila (2016), the cold-climate suitability of overland flow fields has its challenges, taking into account the need for year-round efficient purification. Microbiologically-driven purification processes are particularly prone to become slower during low temperature periods in winter (Feng *et al.* 2012). However, the water temperature effect on purification efficiency still needs more research. This is because some studies, *e.g.* Kadlec and Reddy (2001), have reported (at least partly) lowered nutrient and BOD purification efficiencies at low temperatures, while others (Mæhlum and Stålnacke 1999), have found no significant seasonal difference.

Table 3. Comparison of Traditional and Emerging Peat Bog Drainage Water Treatment Methods

Treatment method	Removal efficiency [%]	Advantages	Limitations	Cost level	Space requirement	Cold-climate suitability	References
Sedimentation pond	SS: 40	Simple	Low P & N removal, maintenance need	Medium	Moderate	Moderate	Heiderscheidt 2016; Kløve 2000; Miettinen *et al.* 2020
Overland flow field (wetlands)	SS: 60 N_{tot}: 25 P_{tot}: 25	Nature-based, low maintenance requirement	Large area requirement, water table level altering may lower upstream area growth medium quality	Low	Very high	Good	Heiderscheidt 2016; Miettinen *et al.* 2020; Postila 2016
Hybrid wetlands	NH_4^+: 95 COD: 80 (domestic wastewater)	Nature-based, low maintenance requirement, water table effects easier to control by design than in conventional wetlands	Moderate-high area requirement, design-sensitivity	Low	High	Good	Hernández-Rodríguez *et al.* 2022
Chemical coagulation (CC)	SS: 30–90 N_{tot}: 30–60 P_{tot}: 75–95	High removal efficiency, low space need	Need for precise control, efficiency fluctuation, may greatly lower water pH, chemical resupplying logistics	High	Low	Weak	Heiderscheidt 2016
Electrocoagulation (EC)	P_{tot}: >(90–95) N_{tot}: 33–41 COD_{Mn}: 75–90 DOC/TOC: 62–75	High removal efficiency, pH neutralization effect, low maintenance, less sludge (better quality) than in CC, compact and simple equipment, no chemical additions (except NaCl)	High-volume PBDW treatment untested, high-capacity grid electricity needed, sufficient conductivity needed	Medium	Low	Very good	Chen 2004; Kuokkanen *et al.* 2013; Kuokkanen *et al.* 2015
Novel single and combined physicochemical methods	Variable (mostly high/very high)	High removal efficiency, low space need	High-volume PBDW treatment untested, complexity of design (combined methods)	High-very high	Low	Variable	Elma *et al.* 2022; Khayan *et al.* 2022

Settling velocities decrease in cold water, which can reduce sedimentation pond removal efficiencies. A study on the Pohjansuo peatland in central Finland showed that nutrient and solids retention in the settling ponds weakened during spring floods (snowmelt induced SS erosion from sedimentation ponds is a well-known problem (Kløve 2000)).

As presented in Table 3, EC has not been tested in a continuous-mode real-world situation for PBDW treatment. However, it has a logistical advantage over CC in this application, since no chemicals need to be frequently transported to remote locations. Avoiding chemical dosing also eliminates pumping-related issues (*e.g.*, freezing). Thus, field testing of EC for PBDW treatment is needed.

No studies regarding treatment/valorization of sludge produced during CC or EC treatment of peat water has been conducted. Rajaniemi *et al.* (2021) have reviewed various EC sludge valorization options (*e.g.*, as a fertilizer (mainly as struvite), pigment, construction material (mainly as blocks)) presented in the literature. Nayeri and Mousavi (2022) conducted a review on CC coagulant recovery and reuse from drinking water treatment sludge, confirming that coagulants may be recovered using various methods. They found that it is possible to reuse the recovered coagulants multiple times in water and wastewater treatment (pollutant removal), or as building and construction materials, constructed wetlands substrate as well as other purposes.

However, to the best of our knowledge, valorization of CC or EC sludges has not reached commercial interest, and landfilling remains the general solution for their disposal. More research on this subject is warranted, including possible leaching of the coagulant metals (Al or Fe) from the landfill. The varying composition of the sludges as well as their high metal content currently prohibits their use in fertilizer/soil amendment usage of the sludges on peat bogs and new innovations are warranted to allow such utilization of the sludges. It should also be noted that nutrient content of PBDW and therefore the resulting sludges are very low. Similarly, despite high removal efficiencies, the high costs of novel single and combined physicochemical methods likely limit them to mainly academic interest for now.

Peat-based Materials for Water Treatment

Peat-based adsorbents offer many advantages in water treatment, *e.g.* high sorption capacity, cost-effectiveness, ease of production, and environmental friendliness in use. Peat can be used for removing various contaminants from water. It possesses several properties necessary for an effective adsorbent for separating dissolved metals from wastewater. The mechanism of metal binding is controversial. The prevailing theories include ion exchange, complexation, and adsorption. In the adsorption process, efficiency is affected by factors such as pH, loading rates, and the presence of multiple metals in the solution (on one hand, there are more metals to be removed, but on the other hand, this leads to sorption site competition, reducing the binding of individual metals). Studies have also shown that peat can be regenerated and metals recovered through acid elution, having only a minor effect on the sorption capacity of peat (Brown *et al.* 2000).

Peat is capable of removing dissolved metals, nutrients, suspended solids, organic matter, oils, and odor compounds from household and industrial wastewater and from oil spills (Viraraghavan and Ayyaswami 1987). As such, the poor physical properties of peat—low mechanical strength, high water retention capacity, chemical instability, and its tendency to shrink and swell—generally make it unsuitable for use in large industrial filters under harsh conditions (Couillard 1994; Brown *et al.* 2000).

Table 4. Efficiency of Peat-based Adsorbents in the Treatment of Different Waters and Wastewaters

Investigated water/wastewater	Used peat chemical	Dosage [g/L]	Initial pollutant levels [mg/L]	Reduction [%]	Adsorption capacity [mg/g]	Study
Sulphide mine leachate	(*Sphagnum Acutifolia* peat, *Sphagnum Cuspidata* peat *Carex* peat)[1], peat granules[2]	20–40	Ca: 13.5 Fe: 0.8 K: 1.7 Mg: 2.3 Na: 2.3 Al: 0.3 Mn: 0.4 Cu: 0.0709 Zn: 0.298 Ni: 0.0034 Cd: 0.000821	Cu: 85–100 Zn: 38–99 Ni: 68–100 Cd: 56–100	Total (Ca, Na, K, Mg, Fe, Mn, Al): 17.7–43.3/13.1–37.0[3]	Ringqvist *et al.* 2002
Landfill leachate	(*Sphagnum Acutifolia* peat, *Sphagnum Cuspidata* peat *Carex* peat)[1], peat granules[2]	20–40	Ca: 173–247.2 Fe: 2.2–3.5 K: 652–679 Mg: 49.6–59.6 Na: 1140–1145 Al: 0.01–0.2 Mn: 0.2–0.3 Cu: 0.0216–0.045 Zn: 0.0535–0.0947 Ni: 0.0391–0.0562 Cd: 0.0006–0.0007 Pb: 0.0024–0.0031	Cu: 32–68 Zn: 89 Ni: 27–55 Cd: 37–81 Pb: 41–80	Total (Ca, Na, K, Mg, Fe, Mn, Al): 1626–1734/0[3]	Ringqvist *et al.* 2002
Laundry wastewater	(*Sphagnum Acutifolia* peat, *Sphagnum Cuspidata* peat *Carex* peat)[1], peat granules[2]	20–40	Cu: 0.72–1.21 Zn: 1.08–1.65 Ni: 0.086–0.132 Cd: 0.0054–0.011 Pb: 0.233–0.424 Cr: 0.141–0.21	Cu: 76–87 Zn: 31–65 Ni: 15–60 Cd: 13–93 Pb: 81–95 Cr: 65–81	n.d.	Ringqvist *et al.* 2002
Artificial multi-metal solutions	*Sphagnum* peat moss	60[4]	As: 0.1–1.0 Cd: 0.1–1.0 Cu: 0.1–1.0 Cr: 0.1–1.0	Cd, Cu, Zn, Ni, Pb[5]: 91–98	n.d.	Kalmykova *et al.* 2009

			Ni: 0.1–1.0 Pb: 0.1–1.0 Zn: 0.1–1.0			
Artificial ibuprofen solutions	NaOH-treated peat + Co-doping	0.1 + 2 mM PMS	Ibuprofen: 10 DOC: 8.43	Ibuprofen: 100 (1 h) DOC: 52 (2 h)/75 (5 h)	5[6] (P 850 + PMS) 13.8[6] (Co-P 850) 100[6] (Co-P 850 + PMS)	Ren *et al.* 2021
Synthetic sulphate solutions	PG-Peat	4	SO_4^{2-}: ~1000–1100	SO_4^{2-}: 43	SO_4^{2-}: 189.5 ± 2.7[7]	Gogoi *et al.* 2019
Mine water	PG-Peat	4	SO_4^{2-}: 1970–2168	SO_4^{2-}: 24–26[6]	SO_4^{2-}: 137 (± ~5[6])	Gogoi *et al.* 2019
Synthetic nitrate solution	Modified peat (epichlorohydrin, ethylenediamine, triethylamine, DMF)	4	NO_3^--N: 30/350	NO_3^--N: ~75/28[6]	NO_3^--N: ~7.5/24.2[7]	Keränen *et al.* 2013
Metallurgical wastewater	HCl-treated peat	0.5	Zn: 0.578–5.467 Ni: 0.0307–0.0424 Pb: 0.0059–0.0424	Zn: 50–70[8] Ni: 30–50[8] Pb: 60–75[8]	Zn: 0.6 (batch) Ni: 0.03–0.04 (batch) Pb: n.d.	Heiderscheidt *et al.* 2020
Metallurgical wastewater	HCl-treated peat	0.5	Zn: 0.311 Ni: 0.042 Cr: 0.022	Zn: 48–65 (batch) Ni: 50–79 (batch) Cr: 32–50 (batch)	Ni: 16 (natural peat)/21[9]	Gogoi *et al.* 2018
Spiked ditch runoff water	HCl-treated peat	0.5	Zn: 0.174 Ni: 0.094 Cr: 0.078 Cu: 0.062	Zn: 4–43 (batch) Ni: 10–38 (batch) Cr: 66–75 (batch) Cu: 36–80 (batch)	Ni: 16 (natural peat)/21[9]	Gogoi *et al.* *2018*

SEE NOTES ON THE NEXT PAGE

1 = Peat samples with low humification (von Post scale 2–4, where 1 is the least and 10 the most humified) were selected. Low-humified peat exhibits a coarser structure than highly humified peat, providing sufficient permeability for column experiments.
2 = Unknown botanical and humification degree origin.
3 = Cation adsorption/release onto peat (mmol/kg) during column experiments after roughly five bed volumes (100 mL) of leachate had passed through the column. In the mine leachate tests, *Sphagnum Cuspidata* and Carex peat were used, while in the landfill leachate tests, *Sphagnum Acutifolia* and *Sphagnum Cuspidata* peat were used.
4 = Seven identical columns (60 mm × 300 mm) packed with 60 g of peat, with varying experimental conditions.
5 = Peat was not found to remove As, regardless of the applied environmental condition. This poor removal is attributed to the occurrence of arsenate/As(V) as negatively charged ions. Chromium was initially removed effectively (97%), but removal dropped to ~80% over time and was influenced by environmental factors. Higher pH (8) and NaCl reduced Cr sorption by 30%, while draining and stagnation caused a 20% decrease.
6 = Estimate based on data given in the article at issue.
7 = Maximum experimental capacity.
8 = Removals achieved during both batch laboratory and continuous mode pilot experiments.
9 = Determined using triplicate batch shaking tests, applying 1 g/L of the HCl-modified and natural peat and nickel nitrate solutions (2–120 mg/L).
n.d. = not determined.

Table 4 presents the effectiveness of peat-based adsorbents in the treatment of various types of water and wastewater. In addition to the data in Table 3 (which is mainly based on the latest research), using peat in water treatment has been studied for decades, particularly in the Nordic countries, France, Canada, and the United States (Coupal and Lalancette 1976; Gosset *et al.* 1986; McLellan and Rock 1986; Couillard 1994; Brown *et al.* 2000).

Ringqvist *et al.* (2002) investigated metal adsorption from various wastewaters onto peat samples (Sphagnum and Carex) and compared the results with other adsorbents such as peat granules, clinoptilolite, glauconite, and steel industry flue dust. The highest metal removal efficiency was observed in sulphide mine leachate, where metals were present as free ions, with Carex peat removing up to 97 to 99% of Zn and 85 to 100% of Cu. Wastewater composition significantly affected adsorption efficiency, and variation among the peat types were associated with the source plant chemistry and environmental conditions during peat formation.

Column experiments were conducted on sphagnum peat moss for its ability to remove multiple metals (As, Cd, Cu, Cr, Ni, Pb, Zn) under varying environmental conditions simulating real-life applications. High removal efficiencies (91 to 98%) were observed for Cd, Cu, Zn, Ni, and Pb, while performance remained stable despite physical changes but was temporarily reduced by NaCl addition and DOC leaching. However, peat was ineffective at removing As and Cr within the tested pH range (6.7 to 8.0). This was likely because these elements existed as negatively charged (As(V) and Cr(VI)) ions at the studied pH (Kalmykova *et al.* 2009).

It should be noted that chromium and arsenic commonly occur in water and wastewater as Cr(VI) and As(V) (Xanthopoulou *et al.* 2025). It has been found that the adsorption effect of Cr(III) on biomasses is stronger compared to Cr(VI), *e.g.*, in the study of Chen *et al.* (2015) using biochar as adsorbent. They found that the mechanism of Cr(III) adsorption by biochar mainly involved $Cr(OH)_3$ precipitation and cation exchange with biochar, but these reactions were ineffective for Cr(VI).

Ren *et al.* (2021) developed a Co-doped catalyst from NaOH-treated peat for efficient ibuprofen degradation in contaminated water. By carbonizing (at 600/700/800/850 °C) Co^{2+}-saturated peat, they created a Co–P 850 catalyst, which, when used as a peroxymonosulphate (PMS) activator, removed 52% of DOC in 2 h and 75% in 5 h.

Gogoi *et al.* (2019) developed a bio-based anion exchanger, PG-Peat, for the removal of sulfate from synthetic solutions as well as authentic mine water with low pH. Amine and quaternary ammonium groups were grafted onto peat using polyethylenimine (PEI) and glycidyltrimethylammonium chloride (GTMAC). The optimal modification resulted in a sulfate uptake capacity of 189.5 ± 2.7 mg/g. PG-Peat showed rapid sulfate sorption within five minutes and maintained high recyclability, with slightly reduced uptake in the presence of nitrate. In another research paper on by the same study group (Gogoi *et al.* 2021), sulphate removal from real minewater by PG-peat was also studied. They found that temperature variation (2 to 22 °C) did not affect the sorptive performance of PG-Peat.

Keränen *et al.* (2021) modified five Finnish lignocellulosic materials, including peat, for nitrate removal from synthetic solutions using epichlorohydrin, ethylenediamine, and triethylamine in the presence of N,N-dimethylformamide (DMF). Nitrate removal capacity of 24.2 mg/g (as NO_3^-–N) for peat was achieved, which is comparable to other lignocellulosic materials and commercial anion exchange resins. The sorption was rapid.

Heiderscheidt *et al.* (2020) investigated the effectiveness of mineral and biomass-based sorbents for treating metallurgical industry wastewater in continuous-flow CSTRs (continuous stirred tank reactor) followed by sedimentation, using both laboratory and pilot-scale experiments. Among others, they tested acid-modified peat (M-Peat), a magnesium-based commercial mineral sorbent (Mineral-P), and ground blast furnace slag (calcium-rich). The results showed that when initial metal concentrations were higher, metal removal efficiencies were also higher. M-Peat achieved significant removal of Zn (50 to 70%), Ni (30 to 50%), Pb (60 to 75%), *etc*., in both laboratory and pilot experiments. The addition of M-Peat had no effect on the water pH, but its poor settling properties limited its application in CSTR systems coupled with sedimentation. SS concentrations were clearly higher in M-Peat-treated outflows (62.7 to 168.2 mg/L) than with Mineral-P (4.0 to 4.2 mg/L) or slag (0.5 to 2.7 mg/L), with an initial concentration of 11.0 ± 9.4 mg/L. A higher dosage was used for M-Peat (0.5 g/L) relative to the other sorbents (0.1 to 0.2 g/L).

A similar conclusion was drawn by Gogoi *et al.* (2018), who reported that even though metal (Ni, Cr, Zn, Cu) removals were achieved from similar industrial wastewater as in Heiderscheidt *et al.* (2020) and spiked urban runoff water using HCl-modified peat, due to its observed low hydraulic conductivity, the use of such small-particle peat material in filter-type passive systems may be restricted. Column tests showed that HCl-Peat performed similarly to a commercial sorbent and mostly did not leach metals into the solutions. The acid treatment of peat was selected to decrease its natural hydrophobicity and improve its poor settling properties, an issue emphasized by Leiviskä *et al.* (2018) in their study on the importance of peat pretreatment in wastewater treatment. Furthermore, according to Batista *et al.* (2009), treatment with HCl would not have oxidised the organic content of peat, as HCl is not an oxidising agent, keeping it intact. As noted by Gosset *et al.* (1986), HCl modification can desorb metals ions originally present in natural peat, thus enhancing its metal adsorption capacity.

In the study of Leiviskä *et al.* (2018), among the tested materials, NaOH-treated peat showed the highest efficiency in nickel removal, followed by HCl-treated peat, citric acid-treated peat, and water-treated peat. The settling of HCl-treated peat was significantly improved with the use of cationic flocculants, especially those with higher charge densities, while NaOH-treated peat already showed good settling properties without additional treatment. Overall, the use of treated peat in mixing and settling systems was found feasible.

Chemical pre-treatments with HCl and NaOH do not always enhance metal adsorption by peat. For example, the Santo Amaro das Brotas peat, which has a high organic matter content, showed decreased Cr(III) adsorption after modification, indicating that its natural composition was already highly effective. In contrast, adsorption improved after modification for the Ribeirão Preto and Itabaiana peats — all three being of Brazilian origin (Batista *et al.* 2009).

The cost-effectiveness of the peat-based adsorbents presented in Table 4 is debatable, at least in the cases where multi-stage pretreatment is performed in order to obtain the adsorbents used. The studies presented in Table 4 contained no economic consideration, especially for real-world long-time use of the adsorbents, both of which warrant further investigation. As suggested by Gogoi *et al*. (2018), further research is needed on metal desorption from saturated sorbents, their potential for reuse, and appropriate disposal methods when regeneration is not feasible. Furthermore, when discussing adsorption, the issue with real wastewaters is in many cases their variability in both quality (pH, contaminant levels) and quantity. This means that laboratory-optimized results for a given sample might not actually easily translate into an industrial-scale application with stable high-level reductions. On the whole, because peat-based adsorbents have not gained commercial interest, they remain mainly of academic interest for now.

ECONOMIC AND SOCIAL IMPACTS OF PEAT UTILIZATION

Another important aspect of peat utilization is its economic and social impacts. Global research related to this is now vigorous because it is connected to climate change mitigation, food production/security, and rural livelihoods, among other issues (United Nations Environment Programme 2022; Räsänen *et al.* 2023; Mursyid *et al.* 2025). However, Räsänen *et al.* (2023) note that few studies compare the impacts of different after-uses of peat extraction sites. The importance of peatland cultivation for agriculture and the importance of peat use for horticulture is significant, and the restoration of peatlands should be carried out in a way that does not endanger people's socio-economic status or income. There is particularly a lot of discussion right now in the EU and Southeast Asia, but interest is also quickly spreading to African wetlands and South American peat forests, which are increasingly recognized as major carbon stores and livelihood sources (United Nations Environment Programme 2022). For example, in Finland, the whole peat industry is currently being driven down by policy, causing problems for the peatland farmers. As a general note, a lot of the research on this subject has been conducted using various types of queries and interviews. Therefore, the formulation of research questions is important and should receive particular attention in the future.

Even though the largest areas of global peatlands are situated in North America and Russia (see Fig. 1), a vast proportion of them are under permafrost. In Canada, peat has been extracted almost exclusively for use in horticulture (Räsänen *et al.* 2023). The share of peat in the fuel balance of Russia has dropped drastically from its peak heights, with decreased extraction leading to peatland fires occurring much more frequently (Tcvetkov 2017). Interest in peat use in these areas remains limited.

European Research

Although costly, restoring peatlands and small water bodies creates employment, enhances biodiversity and water quality, and may increase recreational and tourism potential. A study of five Finnish cases showed that large-scale restoration can generate considerable local employment, particularly in remote regions. Public surveys indicated that citizens' knowledge and opinions had remained stable over five years, yet most respondents regarded restoration as beneficial for nature and recreation, and many supported allocating more tax money to it. A targeted survey in Nuuksio National Park revealed that restoration would not notably change visitation rates, suggesting that its economic effects on tourism are limited in the short term (Juutinen *et al.* 2024).

Buschmann *et al.* (2020) examined emission reduction options for agricultural peat soils in Northern, Eastern, and Central Europe. Based on farmer interviews and expert group discussions, preferred low-emission land uses were mainly driven by economic factors such as productivity of resource systems, land value, and market incentives, while implementation depended more on user diversity and conflicts. The authors pointed out possibilities to transfer solutions between regions and discussed an institutional framework at EU, national and regional levels to facilitate implementation. Primarily EU-level incentives were found necessary for financing alternative land use.

Southeast Asian Research

Elia and Yulanti (2022) conducted a study in the village of Tumbang Nusa, Central Kalimantan, Indonesia, a region where peatland fires occur frequently. They employed a qualitative descriptive method and interviewed 45 farmers, investigating land ownership, crops, income, motivation, and perceptions of peatlands. Successful cultivation and management of peatlands have improved farmers' livelihoods. The farmers (land ownership either in a 2-ha migrant quota or purchased land) understand peatlands and their management despite past ineffective methods. Both Javanese farmers and local residents aim for sustainable income and fire prevention, considering the forests to be vital and believing that peat forests must be protected. However, as demonstrated by the study, they continually need support and guidance for the sustainable use of peatlands, which reconciles environmental protection and local development.

Yeny *et al.* (2022) conducted research, applying, *e.g.* field surveys and interviews, in Central Kalimantan to develop a food estate strategy that considers local community welfare and biodiversity sustainability. Their findings indicate that operating a food estate on degraded peatlands carries a moderate to high risk of negative impacts, and the greatest risk of impacts was deemed for communities and changes in farming practices. It was concluded that once human activities harmfully alter the biophysics of peatlands, costly and time-consuming peatland reparation efforts may be required. Irreversible consequences may also occur and must be avoided. Thus, a strong sustainability perspective must be included in the operation of a sustainable food system on peatlands. A key strategy to achieve this includes protecting natural resources and substituting foreign cultivated plants with native peat plants. This may include, *e.g.*, fragmentation prevention and maintaining habitat connectivity with corridors.

Puspitaloka *et al.* (2021) also conducted a socio-economic study on peatlands in Central Kalimantan and found that funding and institutional challenges substantially hinder restoration efforts. Indirect social costs may represent up to half of total expense. Diversified, market-based funding was found necessary. Hybrid governance through collaboration between the public and private sectors for ecosystem restoration was proposed.

Peatlands have contributed significantly (even as the main source of livelihood) to the local economy in Riau Province, Indonesia, and many communities in this mostly coastal region live on peat-dominated land (Syahza *et al.* 2020). The influence of twelve socio-economic factors (such as demographics and livelihood strategies) on local knowledge and practices related to peatland protection in Riau Province was examined by Yunus *et al.* (2025). While awareness of peatland importance was generally good, it was not well translated into practice, particularly in sustainable agriculture. Education, age, and training participation improved knowledge, whereas income and length of residence affected protection behavior. Bridging the gap between knowledge and action requires disseminating best management practices, diversifying community-based livelihood options, and providing financial incentives. Collaboration and long-term community commitment are central for sustainable peatland management. In another study (Syahza *et al.* 2020) in Riau Province, a conceptual model of sustainable peatland utilization integrating local wisdom and community welfare for the province was proposed.

Economic and Social Impacts of Paludiculture

On the whole, socioeconomic studies of paludiculture remain scarce, warranting further research. In a Latvian study (Balode and Blumberga 2024), the economic costs and benefits of different restoration strategies and alternative uses of peat using the composite index method were (based on scientific literature, reports, and local project studies) compared. The most favorable options were the production of insulation boards and paludiculture using cattail (Typha) and sphagnum. Paludiculture can provide economic benefits for landowners, and the harvested biomass can be utilized in high value-added products without peat extraction, while maintaining ecosystem services.

Wichmann and Nordt (2024) noted that paludiculture remains limited due to historical drainage and restrictive legislation in Germany. They emphasized the need for coherent policies, effective incentives, and empowered communities to enable a just transition. Although focused on Germany, their study provides a useful model for similar efforts across Europe. Country-specific conditions must be considered; for instance, forestry and peat extraction are the main causes of peatland drainage in the Baltic States and Finland, while paludiculture may be less viable in countries with many remote peatlands and low industrial demand for renewable raw materials.

Furthermore, according to Gaudig *et al.* (2018), farming sphagnum offers a clear opportunity to present a solution to pressing societal challenges. In order to do this research, industrial and policy partners should increase collaboration to scaling up sphagnum farming. However, further research is needed to assess the long-term impacts of sphagnum farming, as well as its profitability and environmental benefits outside Germany.

According to Tan *et al.* (2021), in the tropics, socioeconomic considerations heavily influence paludiculture. In an Indonesian paludiculture study (also see earlier chapter "Paludiculture for Greenhouse Gas Mitigation") by Uda *et al.* (2020), among the studied crops, sago palm, and illipe nut were found to score highest in sustainability and market scalability (in principle found able to compete with oil palm), while banana, pineapple, and sweet melon scored highest in market scalability and farmer acceptance. However, a key barrier to large-scale adoption is the long time needed for the crops to become productive—up to eight years for sago and illipe nuts—which is too long for many smallholder farmers to wait for returns. Government support, along with policy and technical assistance, is therefore essential to enable effective harvesting, processing, and marketing of these high-potential crops. Carbon trading could be a tool for generating profits when cultivating paludicultural crops.

CONCLUSIONS

Global peat reserves are large and mostly unused, having great untapped potential. Energy usage of peat has greatly diminished in recent years. Peat use for less GHG-intensive uses could be increased, but such novel alternative uses still require research and new innovations to support their feasibility. Peat is difficult to replace in some of its real-world alternative uses, such as growing media, animal litter (may be used as a fertilizer afterwards), or especially in mixed combustion with wood, as a replacement for coal. Peat ash contains nutrients such as phosphorus but is low in nitrogen, and can be used as a fertilizer in forestry or in non-food crops cultivation on decommissioned peat fields.

New peat valorization options should be assessed against existing alternatives, required technologies, and related costs. Peat is often considered rather heterogenous compared to other lignocellulosic biomasses, and thus favourable economic feasibility of chemical modification, thermochemical processing and chemical fractionation products may be hard to accomplish. There are suitable pre-treatment methods available to achieve more uniform quality in peat products, such as mechanical mixing, screening, drying, pelletizing or briquetting, thermal pre-treatment, and blending with other biomasses. More expensive methods are technically feasible, but they are used only to a limited extent, and they should be developed in a more cost-effective direction.

Opinions on the renewability of peat vary: it is typically seen as a non-renewable fuel, although some experts regard it as a slowly renewable resource, while some compare it to fossil fuels. On the other hand, peatlands function as carbon sinks. Furthermore, peatland GHG emissions can be diminished after production by cultivating various plants on them, which sequester carbon throughout the growing season. This interesting topic requires further research, particularly on socio-economic impacts and global applications.

Methane is a much more potent greenhouse gas than CO_2 (correlation depending on the time horizon). It is emitted from undisturbed peatlands but reduced by drainage, as shown in the literature. Peat combustion does not produce methane; rather, mainly CO_2 and N_2O are generated instead. The theme of GHG emissions from peatlands is complex, and the future of northern peatlands as carbon sinks remains uncertain. The models applied in studies on peatlands function as carbon sinks need to be developed further to reduce the uncertainties related to them. Climate change also causes elevated potential future hazard of peatland fires producing more GHG. Leaving peat unextracted also increases the risk of fires, which further complicates the issue.

Paludiculture (wetland farming where crops are grown on rewetted peatlands) is a rising research topic, which also requires further research, because its effect on GHG mitigation still seem to be variable and unclear. In particular, the effect of paludiculture on peat water quality should be studied in future, also in conjunction with novel peat water treatment technologies. From the perspective of peat producers, conditions would need to be made more eco- and cost-effective for cultivating various crops on rehabilitated peatlands to be worthwhile.

Decommissioned peatlands can be effectively rehabilitated, *e.g.* for utilization in non-food potato production, using various industrial by-products and organic soil amendments. Their use aligns with the principles of circular economy, diminishing environmental impact and enabling the productive reuse of waste materials. Studies on novel applications for utilizing rehabilitated peatlands, using novel types of soil amendments (including their long-term performance) are encouraged. Economic and/or socio-economic aspects should also be considered.

A key environmental concern of peat production is its potential effect on the quality of natural water bodies. Peat water may be used for consumption after proper treatment, highlighting the importance and the possibilities of PBDW treatment on a global scale. Various peat bog drainage water treatment methods have different advantages and limitations, space requirements, costs, *etc*. Data on the specific costs of phases involved in preparing overflow fields and sedimentation ponds remain limited and require further investigation. Globally, the amount of commercial EC vendors is on a significant rise. EC has not been tested in continuous real-world PBDW treatment; therefore, field testing is needed. However, power grid requirement constraints its use in many locations. New less temperature-dependent chemicals should be tested for CC in field applications. Regarding both CC and EC, sludge treatment/valorization needs to be addressed.

Peat itself may be used in water treatment. However, peat-based adsorbents remain mainly of academic interest for now. Future studies should consider long-term cost-effectiveness of peat-based adsorbents, with a particular focus on lowering pretreatment costs required. The heterogenous nature of peat material may constrain its use in this application.

Socio-economic aspects of peat use, closely linked to climate, food, and rural livelihoods, are under vigorous research. Currently, discussion is strong in the EU and Southeast Asia, with growing interest in African and South American peatlands. Socio-economic research largely relies on queries and interviews; careful formulation of research questions is important for future studies.

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Declaration of Generative AI and AI-Assisted Technologies in the Writing Process

During the preparation of this work, the authors used ChatGPT to assist in searching for relevant literature and information, as well as to condense the text. After using this tool/service, the authors reviewed and edited the content as needed and take full responsibility for the content of the publication.

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