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Review

Paludiculture as a sustainable land use alternative for tropical peatlands: A review

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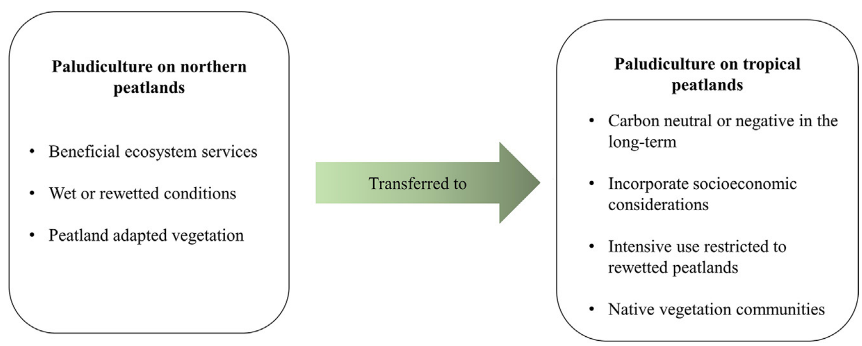
HIGHLIGHTS

- Paludiculture or wet agriculture is a sustainable land use alternative on peatlands.
- Paludiculture should be carbon-neutral or negative in the long-term.
- Wet and rewetted peatlands represent two different management pathways.
- Vegetation selected for paludiculture should be sourced from native species.
- Paludiculture in the tropics is heavily influenced by socioeconomic considerations.

GRAPHICAL ABSTRACT



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ABSTRACT

Peatlands cover approximately 4.2 million km^2 of terrestrial land surface and store up to 700 Pg of terrestrial carbon. Preserving the carbon stocks in peatland is therefore crucial for climate change mitigation. Under natural conditions, peatland carbon storage is maintained by moist peat conditions, which decreases decomposition and encourages peat formation. However, conversion of peatlands to drainage-based agriculture in the form of industrial plantations and smallholder farming has resulted in globally significant greenhouse gas emissions. Paludiculture, loosely conceptualized as biomass production on wet peatlands with the potential to maintain carbon storage, is proposed as a sustainable, non-drainage-based agriculture alternative for peatland use. However, while the concept of paludiculture was developed in temperate ecoregions, its application in the tropics is poorly understood. In this review, we examine common definitions of paludiculture used in literature to derive key themes and future directions. We found three common themes: ecosystem services benefits of paludiculture, hydrological conditions of peatlands, and vegetation selection for planting. Ambiguities surrounding these themes have led to questions on whether paludiculture applications are sustainable in the context of carbon sequestration in peat soil. This review aims to evaluate and advance current understanding of paludiculture in the context of tropical peatlands, which is especially pertinent given expanding agriculture development into Central Africa and South America, where large reserves of peatlands were recently discovered.

Abbreviations: BCR, benefit-cost ratio; FAO, Food and Agriculture Organization; GHG, greenhouse gas; IPCC, Intergovernmental Panel on Climate Change; IRR, internal rate of return; NPV, net present value; NTFP, non-timber forest product; PSF, peat swamp forest; REDD+, Reducing Emission from Deforestation and Forest Degradation; UNFCCC, United Nations Framework Convention on Climate Change.

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Contents

1. Introduction

Peatlands are carbon-rich wetlands formed from the reduced decomposition of vegetation biomass due to waterlogged anaerobic conditions (Page and Baird, 2016). Globally, peatlands are distributed across all continents, varying in structure and function driven by local climate, vegetation, hydrology, and geomorphology. In the tropics, peat soils are generally classified as soils possessing approximately 50% carbon by dry weight (Andriesse, 1988; Page and Baird, 2016). Peatlands play an important role in regulating the carbon cycle. Although they cover ~3%, or 4.23 million km^2, of Earth's surface land area (Xu et al., 2018b), they store an estimated 700 Pg of soil carbon, twice the amount stored in tropical forest trees (Bonn et al., 2016; Page and Baird, 2016; Yu et al., 2010). Peatlands hold also ~10% of the global freshwater (Joosten and Clarke, 2002), and have important hydrological functions. These include providing 3.83% of all potable water stored in reservoirs (Xu et al., 2018a), attenuating flooding in nearby areas (Lupascu et al., 2020b), contributing to river base flows (Bourgault et al., 2014; Hooijer, 2005), maintaining groundwater levels in superficial aquifers (Hooijer, 2005), and buffering against saltwater intrusion (Hooijer et al., 2012a, 2012b; Silvius et al., 2000). Additionally, peatlands also function as producers of biofuel energy, habitats for wildlife, archives of paleo-information, and cultural landscapes possessing aesthetics and spiritual values (Kimmel and Mander, 2010). The protection of peatlands is therefore an urgent priority for climate change mitigation and conservation (Leifeld and Menichetti, 2018; Page et al., 2011).

Peatlands are primarily threatened by industrial monoculture, small-scale agriculture, and forestry expansion that lead to long-term loss of their stored carbon (Lilleskov et al., 2019; Roucoux et al., 2017; Van Asselen et al., 2013; Wijedasa et al., 2018). Approximately 11% (509,000 km^2) of global peatlands experienced some degree of degradation due to human activity, the majority of which are concentrated in the tropics (242,000 km^2) (Leifeld and Menichetti, 2018). The first stage of establishing peatland agriculture often entails draining peatland to stabilize underlying peat and create dry conditions for plant growth (Comte et al., 2012; Landry and Rochefort, 2012). The lowering of water table exposes peat to oxygen and facilitates previously-inhibited aerobic bacterial decomposition of organic material, releasing vast quantities of stored carbon as CO_2 gas (Hooijer et al., 2012a; Wijedasa et al., 2017; Wösten et al., 1997). It is estimated that CO_2 emission from drainage in Southeast Asia alone amounts to approximately 1.3–3.1% of emission from global fossil fuel combustion (Hooijer et al., 2010; Wijedasa et al., 2018), and more than 900 g CO_2 m^{-2} year^{-1} are emitted for every 10 cm of drainage (Couwenberg et al., 2010). Following drainage, high rates of decomposition, consolidation (i.e. settling) and compaction cause the peat surface to subside below its original elevation. Initial subsidence is approximately 1–1.5 m in the first five years post-drainage (Hooijer et al., 2012a; Hoyt et al., 2020), upon which biological decomposition becomes the predominant driver of longer-term subsidence at a rate of 5 cm per year (Andriesse, 1988; Hooijer et al., 2012a; Hoyt et al., 2020; Wösten et al., 1997; Wijedasa et al., 2018). Drainage also increases fire susceptibility and leaching, thereby driving additional carbon loss (Hergoualc'h and Verchot, 2011; Moore et al., 2013; Turetsky et al., 2015; Waddington and Price, 2000). In the tropics, approximately 1.48 Pg CO_2 are emitted annually from degraded peatlands through the processes of oxidation, leaching, and biomass burning (Leifeld and Menichetti, 2018), with the amount of carbon lost contingent upon the type of land use change and management practices (Couwenberg et al., 2010; Hergoualc'h and Verchot, 2011; Murdiyarso et al., 2010).

Presently the conversion of peat swamp forest (PSF) for economic development is especially accentuated in Southeast Asia (Gaveau et al., 2019; Miettinen et al., 2012; Wijedasa et al., 2018). Satellite estimates indicate that 50% of forested peatlands in Peninsular Malaysia, Sumatra, and Borneo had been converted to industrial plantations and other forms of managed systems such as smallholder agriculture by 2015; a stark contrast against 1990 where only 11% of peatlands were converted (Miettinen et al., 2016; Wijedasa et al., 2018). Between 1990 and 2010, conversion of PSF to smallholder agriculture and oil palm plantations in Southeast Asia released 1.46–6.43 Pg of CO_2, with another projected 4.43–11.45 Pg CO_2 emitted by 2030 under current development trajectories (Wijedasa et al., 2018). In other parts of the tropics such as South America and Africa, newly-discovered peatlands are being threatened by mining and hydropower dam operations, oil and gas extraction, road construction, habitat fragmentation, and climate change (Dargie et al., 2019; Draper et al., 2014; Lilleskov et al., 2019; Murdiyarso et al., 2019).

The role of peatland carbon storage in climate change mitigation has led to an interest in managing peatlands sustainably to protect underlying carbon stock (Hermanns et al., 2017). This include efforts to restore and rehabilitate degraded or abandoned peatlands through revegetation (Dohong et al., 2018; Hytönen et al., 2018). An oft-proposed method is to manage the land under paludiculture (Fig. 1). Paludiculture is loosely defined as the “sustainable production of biomass on wet and rewetted peatlands” (Wichtmann and Joosten, 2007), where biomass refers to any form of material derived from biological origins. Although the use of wet peatland resources dates back many decades, the application of the term ‘paludiculture’ is a recent phenomenon. Given the importance of peatlands as carbon sinks, sustainably using peatlands entails maintaining a neutral carbon balance, with long-term prospects of turning these ecosystems into carbon-negative sinks (Wichtmann and Wichmann, 2011). This goal is reflected in the adoption of paludiculture in climate change-related policies by organizations such as the Ramsar Convention on Wetlands and FAO (Wichtmann and Couwenberg, 2013).

Despite its expanding usage, paludiculture as a concept lacks clear guiding principles. The literature on sustainable agriculture has long recognized the ambiguities in defining broad concepts, such as

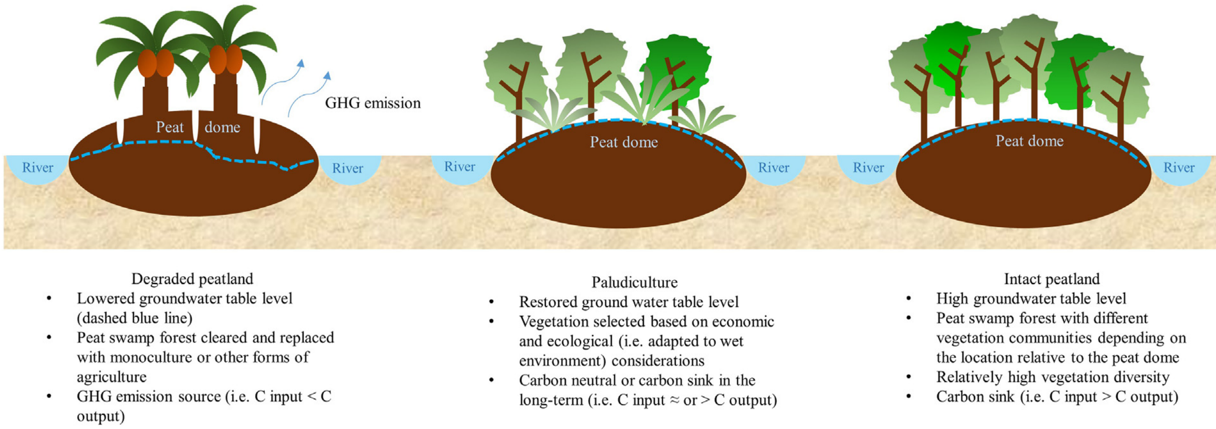

Fig. 1. Peat landscape gradient. Paludiculture land uses range along a continuum of vegetation types and complexity, hydrological intactness, and land use intensity.

sustainability in agriculture, that vary based on the environmental, socio-economic, and political context (Pannell and Schilizzi, 1999). Such variations can promote debates and highlight synergies between different schools of thought, which is key to advancing a concept (Velten et al., 2015). Nevertheless, concerns have been raised about the misappropriation of paludiculture to peatland-degrading practices such as partial drainage or higher water table systems with net positive carbon balance, thus eroding the ecological integrity and long-term existence of peatlands (Budiman et al., 2020; Giesen, 2013; Giesen and Sari, 2018; Sari et al., 2018; Tata, 2019; Taylor et al., 2019). It is through critical examination of the common definitions and application of paludiculture that we can unravel perceptions and priorities about the concept and assess how these definitions influence practice and research.

This review aims to answer the following questions: what is paludiculture and what are our current knowledge regarding the potential application of paludiculture in the tropics? We examine existing paludiculture definitions to derive key themes and highlight contentious points. We then review the current literature about peatlands and paludiculture within the framework of the key themes. Due to a scarcity of paludiculture research from the tropics, we draw upon research conducted in northern peatlands – which have a longer history of paludiculture development and potential influence on current definitions – where relevant, to identify areas of research gaps for tropical paludiculture. Finally, we provide recommendations for future paludiculture research in the tropics.

1.1. Northern compared to tropical peatlands

The development of paludiculture is grounded in research from northern peatlands. Biophysical, meteorological, and socioeconomic differences between ecoregions makes it challenging to transfer a concept from northern latitudes to the tropics. Understanding the commonalities and disparities between northern and tropical peatlands is a crucial first step for determining the extent that paludiculture is applicable to the tropics and future research priorities (Table 1).

Peatlands, in general, share several characteristics: they are wetland ecosystems with more carbon input than output and act as long-term carbon sinks, the function of which can be rapidly reversed by disruptions to the hydrology and surface vegetation (Joosten and Clarke, 2002). Although Bacon et al. (2017) argued that peatlands are governed by the same processes at large between the north and the tropics, there exist clear distinctions between the ecoregions. The majority of the world's peatlands occur in the northern hemisphere, encompassing both boreal and temperate regions. In low relief (i.e. poor draining) environments, such as those found in Europe, North America, and Russia, peatlands form under conditions of high precipitation and low temperature (Page et al., 2009). Both boreal and temperate peatlands are predominantly formed from bryophytes and graminoids, displaying slower rates of accumulation and decomposition relative to the tropics (Frolking et al., 2001; Wieder and Vitt, 2006). Global estimates indicate that northern peatlands cover 3,794,000 km^2, but only store 448.9 Gt C at a density of approximately 118,318 t C km^{-2} (Leifeld and Menichetti,

Table 1
Characteristics of boreal, temperate, and tropical peatlands.

Characteristics	Boreal	Temperate	Tropical
Topography	Located on low and high altitudes	Located on low and high altitudes	Predominantly located on lowlands
Predominant vegetation types	Bryophytes (e.g. *sphagnum*, moss), graminoids (e.g. sedges)	Bryophytes (e.g. *sphagnum*, moss), graminoids (e.g. sedges)	Woody tropical trees
Climate	Cooler temperatures; precipitation rate greater than evapotranspiration rate	Seasonal fluctuations in temperature and humidity; precipitation rate less than evapotranspiration rate	Constant high temperature and humid environment; high precipitation and evapotranspiration rate
Physical and geochemical characteristics	Range from nutrient-poor ombrotrophic bogs to nutrient-rich minerotrophic fens; slower decomposition and accumulation rate; lower hydraulic conductivity; low bulk density	Range from nutrient-poor ombrotrophic bogs to nutrient-rich minerotrophic fens; slower decomposition and accumulation rate; lower hydraulic conductivity; low bulk density	Nutrient-poor; accelerated decomposition and accumulation rate; high hydraulic conductivity; low bulk density
Biodiversity	Low but specialized biodiversity specific to peatland types	Low but specialized biodiversity specific to peatland types	High biodiversity, many of which can co-habit in other ecosystem types
Population size	Relatively population-sparse landscapes	Relatively population-sparse landscapes	Relatively population-dense landscapes
Major drivers of degradation	Peat mining, grazing, biomass production, drainage	Peat mining, grazing, biomass production, drainage	Deforestation, agriculture, drainage
Examples of peatland type	Bogs, fens	Bogs, fens, alluvial mires	Peat swamp forests

2018). Despite majority of peatland area being found in the boreal ecoregion, paludiculture research is concentrated on temperate peatlands. Hence, for the rest of this review, 'northern peatlands' will be used synonymously with temperate peatlands.

Tropical peatlands are mostly found in the lowlands, often located between river bodies and near the coast. While some higher altitude tropical peatlands do exist, their conversion to agriculture is limited. Tropical peatlands are dominated by woody trees with greater primary productivity, resulting in rapid rates of peat soil formation (Wüst et al., 2007). This process is contrasted by accelerated decomposition rates under a high temperature climate, causing degraded tropical peatlands to contribute more substantially to global GHGs emissions (Page and Baird, 2016; Page et al., 2009). Although tropical peatlands cover only 587,000 km^2, they store 119.2 Gt C at a density per unit area of 203,066 t C km^{-2} (Leifeld and Menichetti, 2018).

The different origin of peat-forming plants between northern (i.e. moss/sedge) and tropical peatlands (i.e. woody trees) further results in variations in peat characteristics and responses to disturbances. Tropical PSF trees possess extensive root structures creating large pore spaces, leading to higher hydraulic conductivity and lower bulk density near the surface, and greater water retention capacity in depths of more than 50 cm (Page et al., 2009; Wösten and Ritzema, 2001). Similar patterns of hydraulic conductivity and bulk density are also found in northern peatlands, albeit in smaller magnitudes due to smaller pore spaces. On average tropical peat bulk density can exceed 0.2 g cm^{-3} for sapric peat (FAO, n.d.) and hydraulic conductivity varies between 0.001 and 13.9 m day^{-1} depending on land use (Kurnianto et al., 2019), whereas the bulk density and unsaturated hydraulic conductivity of northern peatlands are 0.02–0.254 g cm^{-3} and 0.07–1.04 m day^{-1}, respectively (Rezanezhad et al., 2016). When disturbed, positive feedback between drainage, tree-clearing, and fires induce vegetation composition to shift towards more fire-prone grasses and flood-tolerant ferns in the tropics, thereby exacerbating disturbance risks (Hoscilo et al., 2011; Miettinen et al., 2013; Page et al., 2009). Vegetation shift towards vascular plants as opposed to bryophytes due to drainage and peat mining disturbances is observed in northern peatlands (Girard et al., 2002; Lachance and Lavoie, 2004). These dynamics pose challenges for restoring peatlands and their ecological functions: Estimates indicate that peat accumulation may be restored after 20 years in bryophyte-dominated temperate peatlands (Graf et al., 2012; Lucchese et al., 2010), whereas the process has not been studied and could take longer in the tropics depending on tree species (Budiman et al., 2020; Harrison et al., 2020).

Another unique characteristic of tropical peatlands is the complexity of PSFs and associated biodiversity. Over 1400 plant species (Giesen, 2013; Giesen et al., 2018; Posa et al., 2011) and 700 fauna species (Posa et al., 2011) have been documented in Southeast Asia PSF alone, compared to more depauperate, but specialized species found in northern peatlands (Joosten and Clarke, 2002; Warner and Asada, 2006). For example, a review of Canadian peatland plant species yielded 711 species, with the majority being herbaceous species and bryophytes (Warner and Asada, 2006). Surveys of a temperate Himalayan peatland found 460 plant species and 165 bird species, some of which are listed as threatened (O'Neill et al., 2020). In comparison, much of the ecology and diversity of tropical peatlands remains understudied, particularly for recently discovered areas in the Congo Basin (Dargie et al., 2019; Xu et al., 2018b) and South America (Gumbricht et al., 2017; Lähteenoja and Page, 2011; Lilleskov et al., 2019; Murdiyarso et al., 2019; Roucoux et al., 2017).

Aside from biophysical differences, the development of peatlands in different ecoregions has followed slightly disparate pathways. Peat mining for biofuel production and livestock grazing are documented extensively in northern ecoregions. In places such as Eastern Europe, large areas of peatlands have been abandoned after the peat extraction industry declined (Joosten et al., 2012). Currently, agricultural use of peatland is ongoing in Europe, North America, and Southeast Asia. Unlike its northern counterparts, tropical peatlands have attracted concerns for the extent, scale and speed by which they are converted to industrial monoculture, especially in Southeast Asia (Dohong et al., 2017; Joosten et al., 2012; Miettinen et al., 2012). Large-scale biomass production such as oil palm (*Elaeis guineensis* Jacq.) and pulpwood plantations has received the bulk of attention due to their destructive impacts on the carbon stock of peatlands (Evans et al., 2019; Hooijer et al., 2010; Miettinen et al., 2012; Murdiyarso et al., 2019). Moreover, unlike the low population density of northern areas, relatively high population density in parts of the tropics may result in land use competition against paludiculture (Lilleskov et al., 2019). Practitioners should take these unique characteristics of tropical PSF into consideration to guide paludiculture projects and research.

2. Key themes of paludiculture

As paludiculture is a nascent but rapidly growing concept, there are great uncertainties about what it is. We noted that the overarching definition of paludiculture as the "sustainable production of biomass on wet and rewetted peatlands" (Wichtmann and Joosten, 2007) can refer simultaneously to a set of management practices, a form of land use, and/or an agriculture system. Closer examination of how paludiculture is defined in the literature reveals three overlapping themes (Table 2): paludiculture ecosystem services (Sections 2.1 and 2.2), the hydrological conditions of peatlands (Section 2.3), and the endemicity of vegetation selected for paludiculture (Section 2.4). There is, however, limited consensus on how these themes are applied in practice. We elaborate on the key themes and points of contention below:

2.1. Paludiculture ecosystem services

Ecosystem services benefits (Millennium Ecosystem Assessment, 2003) is a frequently cited theme in defining paludiculture's normative goals. Strong emphasis is placed on provisioning services such as

Table 2
Examples of the key themes and overlapping definitions that highlight the characteristics of paludiculture. Bolded texts highlight the parts of the definitions that correspond to the theme.

Theme	Paludiculture characteristics	Example
Ecosystem services of paludiculture	Paludiculture can provide multiple ecosystem services.	"*Paludiculture, defined as 'the agricultural use of wet and rewetted peatlands', can contribute to the* ***mitigation of agricultural GHG emissions*** *but still allow the* ***production of renewable raw materials*** *and thus offer an economically viable alternative for farmers*" (Schlattmann and Rode, 2019).
Peatland hydrological condition	Paludiculture should only be practiced on rewetted peatlands	"*[…] cultivating wetland plants as biomass crops on* ***reflooded peatlands*** *to reduce [GHG] emissions and mitigate climate change*" (Ren et al., 2019).
	Paludiculture can be practiced on intact and rewetted peatlands	"*Paludiculture has been identified as an alternative management strategy consisting [of] the cultivation of biomass on* ***wet and rewetted peatlands***" (Silvestri et al., 2017).
Source of vegetation	Any vegetation that can tolerate general wetland conditions are suitable as paludiculture crop	"*Paludiculture […] is agriculture on wet or rewetted peatlands, with cultivation of* ***wetland species*** *[…] in a way that preserves the peat body*" (Temmink et al., 2017).
	Paludiculture crops need to be sourced from native peatland species	"*The rehabilitation of [PSFs] using* ***native peatland tree species****, also known as paludiculture*" (Tata et al., 2018).

biomass production (Pouliot et al., 2015; Wichmann, 2017) and regulatory services such as greenhouse gas (GHG) emission reduction (Surahman et al., 2018; Vroom et al., 2018). Supporting and cultural ecosystem services are cited less in definitions and rarely the subject of study. Although the measurement and valuation of ecosystem services have been extensively researched for wetlands, little has been done for peatlands under paludiculture management, much less ways to translate these services into policies (Wichmann et al., 2016). The holistic valuation of ecosystem services, including supporting and cultural services, in economic terms is crucial towards forming policies that promote paludiculture, but which is difficult due to their non-consumptive nature (Bonn et al., 2016; Wichmann et al., 2016). Accounting for these ecosystem services is further complicated by the ambiguous inclusion of paludiculture under general peatland restoration or rehabilitation efforts, which may include non-paludiculture production systems such as mixed-rubber agroforestry or small-scale drainage-based crops such as vegetables (Giesen and Sari, 2018; Tata, 2019).

Cultural services reflect how communities value and perceive peatlands, which can act as an opportunity or obstacle for implementing paludiculture in relatively population-dense areas (Comberti et al., 2015; Giesen and Sari, 2018; Thornton, 2017). Paludiculture may align with traditional, small-scale practices, thus generating knowledge for the management of peatlands in a way that sustain ecosystem functions and processes in the long-term. For example, indigenous communities such as the Urarina nation in Peru developed a unique set of terminology for distinguishing peatlands by their underlying hydrology, topography, and vegetation (Schulz et al., 2019a). This knowledge can inform future land use mapping and landscape-scale carbon measurements by providing finer-scale and contextualized land cover classifications. Likewise, peatland communities in Sumatra, Indonesia, practiced the timber harvesting method of *ongka* – a combination of wooden sleigh and rails sometimes accompanied by a small-scale canal system – to transport logged woods out of PSFs (Kuniyasu, 2002). The traditional system had less impact on the PSF compared to commercial methods of logging and transport. Such practices and knowledge are developed and refined after decades of coexistence with peatlands and can serve to inform modern management. Other overlooked dimensions of cultural ecosystem services that can align with paludiculture systems include rural development, health, land tenure, paleo-ecological knowledge, and gender equity (Gasparatos et al., 2011; Joosten and Clarke, 2002). However, as these cultural services are non-tangible and non-consumptive (Wichmann et al., 2016), they are often disregarded in market valuation and ecosystem services assessments, leading to the erosion of cultural values and ultimately, the peatland environment.

Different methods of accounting for ecosystem services have been developed depending on the type of service. Wichmann et al. (2016) proposed using market values to capture provisioning services, replacement or damage costs for regulating services, whereas cultural services dependent on decisions involving individual perceptions and choices are calculated through contingent valuation and choice experiments (Turner et al., 2016; Wichmann et al., 2016). Valuing cultural services is also necessarily context-specific. Schaafsma et al. (2017) noted that although traditional Dayak cultivation practices do not rely strongly on canals, community members were against building canal blocks for peatland rewetting due to perceived flood risk. They were unlikely to engage in alternative land uses such as paludiculture unless benefits can be directly accrued to the communities and enhance self-sufficiency. To take another example, the *ongka* system did not damage PSF when logging intensities were restricted to small-scale local use. But when the same system was applied for large-scale industrial logging, it led to the removal of vast quantities of trees, often prior to natural regeneration, resulting in longer term and more extensive degradation. Moving beyond a static view of ecosystem services, temporal, geographical, and scalar changes also need to be incorporated into the valuation processes (Wichmann et al., 2016) – for example, paludiculture systems can create additional values for the communities in the future after the biomass is ready for harvest.

Following the valuation of ecosystem services, questions have been raised regarding methods to translate ecosystem services into tangible policies. Presently most policies focused on accounting for carbon loss arising from land use changes. An example is the Intergovernmental Panel on Climate Change (2014) amendment to the 2006 national GHG inventories guidelines, which acknowledged paludiculture as a specific type of activity on rewetted organic soil. Yet the narrow emphasis on carbon can result in the undervaluation of other paludiculture-related ecosystem services (Comberti et al., 2015). Efforts are underway to develop more holistic methods of accounting for all ecosystem services types, including co-benefits and trade-offs between services. An example is the bundling or layering ecosystem services for markets (Lau, 2013). The former creates market packages for multiple ecosystem services generated from paludiculture; the latter is concerned with separating and marketing individual ecosystem services (Bonn et al., 2014; Lau, 2013; Reed et al., 2017).

Thus far few markets exist for peatland ecosystem services and most of them are concentrated within northern peatlands, an example being the UK Peatland Code certification targeted towards peatland restoration for climate benefits (Bonn et al., 2014). There are, however, existing carbon market instruments, such as carbon credits or carbon taxes, that can be applied to peatlands and paludiculture (Bonn et al., 2014), although these require a better understanding of the mechanisms regulating ecosystem carbon balance to sufficiently capture the carbon values of peatland.

2.2. Paludiculture ecosystem services: a carbon balance perspective

At the national and international level, substantial interests exist to develop paludiculture for reducing GHGs emissions and increasing carbon storage as ecosystem services. The potential of paludiculture to fulfill the REDD+ and UNFCCC requirements is frequently cited (Murdiyarso et al., 2019; Sofiyuddin et al., 2012; Tata and Susmianto, 2016), although this is contingent upon multiple factors such as: groundwater table level, the type of GHG considered, the dominant vegetation, and management practices (Couwenberg et al., 2011).

One mechanism for GHGs reduction in paludiculture is through the process of rewetting: reinstating groundwater table level to reduce aerobic peat decomposition (Furukawa et al., 2005; Hergoualc'h and Verchot, 2011; Karki et al., 2014, 2016; Page and Baird, 2016). In reality, the dynamic between groundwater table and GHG emission is complex. For example, anaerobic conditions can substantially decrease CO_2 emission, but support root respiration and the egression of CO_2 (Jauhiainen et al., 2016). Emission reduction may also be offset by increases in CH_4 emissions due to methanogenesis, or the decomposition of organic matter under anaerobic conditions (Couwenberg et al., 2010; Furukawa et al., 2005). CH_4 is a more potent GHG than CO_2 but has received less attention within tropical peatland research as it is not typically released from peat surface (Couwenberg et al., 2010; Pangala et al., 2013) and tropical peatlands are assumed to produce less CH_4 compared to northern peatlands (Jauhiainen et al., 2008). Recent studies identifying alternative sources of CH_4 from tropical peatlands and model estimates indicate, however, that tropical peatlands may in fact produce a substantial amount of CH_4, potentially 2–3 times that of northern peatlands (Lupascu et al., 2020a; Melton et al., 2013; Pangala et al., 2013; Welch et al., 2019), with seasonal fluctuations between CH_4 sink or source contingent upon precipitation and changes in surface water table level (Melton et al., 2013; Welch et al., 2019).

The effect of rewetting on the emission of another potent GHG of concern – N_2O – is less evident. N_2O emissions are positively correlated to external nitrogen inputs and nitrate concentrations in soils (Kandel et al., 2013; Karki et al., 2014), which may be released upon rewetting and induce eutrophic conditions (Giannini et al., 2017; Harpenslager et al., 2015; Van De Riet et al., 2013). Legacy effects from past and current agriculture activities can pose a challenge for GHG regulation in peatlands. Monoculture plantations such as oil palm and acacia rely on heavy fertilizer input to thrive in nutrient-poor peatlands, contributing to elevated N_2O emissions (Takakai et al., 2006). The use of paludiculture vegetation to improve water quality by absorbing residual nutrients on abandoned agriculture sites can offer potential solutions to the issue of N_2O emission (Giannini et al., 2017; Vroom et al., 2018), although it has not been explored in the tropics.

Vegetation indirectly affects GHG emissions through altering the chemical composition of litter and the release of GHG via specialized gas-exchange vegetation structures, such as the aerenchyma or lenticel (Lupascu et al., 2020a; Welch et al., 2019). The effects on GHG are species-dependent. For example, planting reed canary grass at −10 to −20 cm groundwater table level can reduce N_2O emission by up to 86%, but substantially increase CH_4 by over 70% (Karki et al., 2015). *Sphagnum* and cattail were found to lower CH_4 emissions post-rewetting (Günther et al., 2017), but had negligible effects on N_2O emissions (Vroom et al., 2018). Still others documented modest impacts where the water table had not reached equilibrium (Palmborg, 2012). In tropical forests, Pangala et al. (2013) measured the rate of CH_4 diffusion of a number of wetland species and demonstrated that tree stems can be a major source of CH_4 emissions. They determined that trees with greater wood density and lower lenticel count resulted in lower gas exchange and emission. Furthermore, *Shorea balangeran*, a native PSF species widely harvested for timber production and promoted as a paludiculture crop, had the highest CH_4 emission within the study site. Their results were corroborated by Welch et al. (2019) showing that tree stems act as conduits of CH_4 and N_2O emission. Factoring tree stem emissions into the carbon balance of paludiculture land use will reveal a clearer picture of the extent of how these systems perform in relation to other land uses and assist in vegetation selection (see Section 2.4).

Measurements of GHG emissions from intact peatlands against paludiculture systems have suggested that the latter may not reduce as much emissions as previously envisioned. Mander et al. (2012) investigated the impacts of planting reed canary grass (*Phalaris arundinacea* L.) on abandoned peat extraction area, fen meadow and natural bog in Estonia. They found that the natural bog emitted higher CH_4 (24 kg CH_4 ha^{-1} $year^{-1}$) than the planted plots (1.75–3.56 kg CH_4 ha^{-1} $year^{-1}$), although the CO_2 emission from the fertilized planted plot (21,938 kg CO_2 ha^{-1} $year^{-1}$) was higher than the natural bog's (8,448 kg CO_2 ha^{-1} $year^{-1}$). Compared to bare peat, plots planted with reed canary grass had lower global warming potential. This pattern is observed in another study of GHG balance at a rewetted peatland in Ireland (Wilson et al., 2016), where the rewetted site had a significantly reduced CO_2 emission factor of -104 ± 80 g CO_2-C m^{-2} $year^{-1}$, but higher CH_4 (9 ± 2 g CH_4-C m^{-2} $year^{-1}$) than natural sites. The authors concluded that even after a decade of rewetting, the global warming potential of rewetted sites was still higher than intact peatlands.

Even though there is a lack of paludiculture systems in the tropics, comparisons of restored sites and wet agriculture systems with intact PSF show an equally complex GHG dynamic. A pre- and post-rewetting study on PSF and deforested burnt sites in Central Kalimantan found greater CO_2 fluxes in the PSF compared to the deforested site, although no significant changes were detected before and after rewetting (Jauhiainen et al., 2008). The deforested site was also a small source of CH_4 emissions (0.197–0.275 g CH_4-C m^{-2} $year^{-1}$), though notably less than the annual CH_4 release documented in northern peatlands (e.g. *Sphagnum*-bogs release 2–15 g CH_4-C m^{-2} $year^{-1}$) (Couwenberg et al., 2010; Jauhiainen et al., 2008, 2016). Another extensively studied wet agriculture system in the tropics is rice planting. Lowland rice fields produce less CO_2 and N_2O fluxes (30.1 ± 8.1 mg C m^2 h^{-1} and 3.77 ± 3.89 μg N m^{-2} h^{-1}, respectively) than drained PSFs (266.2 ± 33.6 mg C m^2 h^{-1} and 25.36 ± 8.43 μg N m^2 h^{-1}, respectively) and natural PSFs (94 mg C m^{-2} h^{-1}) (Furukawa et al., 2005). The finding is supported by simulations showing that jelutong agroforestry and rice farming would reduce CO_2 emissions by ~13–21% (Surahman et al., 2018). Hergoualc'h and Verchot (2012) noted that rice fields produce up to $10{,}760 \pm 6020$ kg C m^{-2} $year^{-1}$ in CH_4 emission, whereas intact PSF in Southeast Asia produce 2860 ± 970 kg C m^{-2} $year^{-1}$. CH_4 emissions can increase and peak around the rainy season following the conversion of secondary PSF to paddy field (Inubushi et al., 2003). Similar wet agriculture systems such as papyrus (*Cyperus papyrus* L.) cultivation can also contribute to biomass production and carbon sequestration, but are conducive to methanogenesis (Pacini et al., 2018; Saunders et al., 2014).

Finally, land use practices may also affect the carbon balance of peatlands and paludiculture. Here we focus on traditional communities and their use of peatland resources (Table 3), which generally have less impacts as opposed to industrial plantations and may be integrated with elements of sustainable practices and governance. For example, the Mestizo communities of the Peruvian Amazon peatlands actively plant native palm tree seedlings to restore the palm stock. They also avoid wasteful tree-felling practices during the harvest of palm products (Schulz et al., 2019b). In Indonesia, some peatland communities limit cultivation to shallow peatlands (<1–3 m) and in areas near the coast where there is external nutrient input from the ocean (Limin et al., 2007; Surahman et al., 2018). Deeper peatlands are generally avoided due to nutrient-poor conditions and difficulty with controlling the water table level (Noor, 2001).

Notwithstanding, care should be taken to not conflate all local-level practices as sustainable. Historically, community-scale overexploitation of PSF species is related to shifts in market forces, governance, and territorial disputes, driving the species to the point of endangerment (Potter, 2005). Notable examples include the overexploitation of Gemor (*Alseodaphne coriacea* Kosterm.) for mosquito repellant (Joosten et al., 2012) and jelutong (*Dyera* sp.) for sap production (Potter, 2005). When local resources are depleted, communities are likely to expand further into the PSF frontier or switch to another resource for their livelihoods rather than actively restore the resources. More controversially, small-scale drainage systems such as the *handil* (small canal used for drainage and transportation of produce) and

Table 3
Examples of local communities that utilize peatland resources across the tropics.

Geographic location	Communities	Type of peatland use	Example of management practice/reference
Cuvette Central, Congo Basin	Bantu people	Fishing; small-scale farming; hunting and gathering of resources	Most agriculture plots are located close to *terra firme* forests rather than PSF (Dargie et al., 2019)
Loreto, Peru	Urarina indigenous nation and mestizo communities	Hunting and gathering of resources	Mestizo community members practice sustainable harvesting and cultivation of native palm species and are aware of their limited resources compared to the indigenous communities (Schulz et al., 2019b)
South East Pahang, Malaysia	Jakun tribe	Hunting and gathering of resources; fishing	Sundari, 2005
Indonesia	Dayak communities	Small-scale farming	Dayaks limit cultivation to shallow peatlands using native species such as sago and jelutong (Limin et al., 2007; Tata and Susmianto, 2016)

tatah (small canal made to increase access into forests) built by some indigenous communities on the island of Borneo may release higher GHG emissions relative to undrained peatlands (Limin et al., 2007; Osaki et al., 2016; Suyanto et al., 2009). Tata (2019) studied various mixed smallholder agroforestry systems in Jambi and Central Kalimantan, Indonesia, and found that peat subsidence and CO_2 emission were present in all systems, including plots with native PSF species but where water table level was only partially restored.

These non- or partial-drainage uses of peatlands challenge underlying assumptions of paludiculture, which conflate wet peat production with reduced GHGs emission and the protection of other natural peatland ecosystem services; and beg the question of whether paludiculture land uses should expand beyond a simple criteria defined by production on 'wet or rewetted' peatlands (Section 2.3). From a carbon balance perspective, the overall GHG balance from paludiculture should be comparable to intact peatlands, if it is to be treated as an alternative to drainage-based production. Observations of the relationship between vegetation, hydrology, and carbon, and the length of time it takes to restore degraded peatlands, have led to suggestions that paludiculture should only be implemented on rewetted peatlands, whereas undisturbed peatlands are preserved intact to avoid irreversible ecosystem damage (Giesen and Sari, 2018; Joosten and Clarke, 2002; Wichtmann and Joosten, 2007).

2.3. Paludiculture hydrology: differentiating between restored and intact baselines

Reinstating hydrological functions or processes of peatlands through restoring peat water table level is a prerequisite for paludiculture. The phrase 'wet and rewetted' peatlands – or any synonyms thereof –is used frequently in definitions to denote the peatland's hydrological state. However, this phrase risks oversimplifying the heterogeneity of different peatlands types and their degradation levels to a narrow focus on hydrology (Morley, 2013). Within this context, any form of biomass production on peatland can be considered as paludiculture so long as groundwater is maintained near the peat surface. Such generalization overlooks short- and long-term chemical, structural, biological, and hydrological disparities between rewetted and intact peatlands, with consequences for the evaluation of paludiculture projects (Graf et al., 2012; Kennedy and Price, 2005). It also overlooks spatial heterogeneity within a single intact peatland hydrological unit, where the peat dome possesses distinct hydrology from the edges near the rivers (Giesen and Sari, 2018; Wösten et al., 2006, 2008). Instead, we should recognize that 'wet' and 'rewetted' peatlands refer to two distinct management pathways: converting undisturbed, hydrologically intact peatlands to productive non-drainage use or rewetting a previously drained system and converting it to non-drainage production.

To date many paludiculture research are conducted at a pilot capacity (Dohong et al., 2018; Jauhiainen et al., 2008; Pouliot et al., 2015). In the tropics, paludiculture research focus on field evaluations of existing agroforestry systems against short-term socio-economic and environmental metrics (Budiman et al., 2020; Prastyaningsih et al., 2019; Tata, 2019). A caveat is that most of these agroforestry systems do not have fully restored water tables, which Budiman et al. (2020) termed 'compromised paludiculture'. On the other hand, the most commonly employed research method in northern peatlands is experiments comparing the impacts of paludiculture to a predetermined baseline. Baselines for evaluating the performance of paludiculture are usually set against drained peatlands (Günther et al., 2017; Hytönen et al., 2018; Kandel et al., 2013; Palmborg, 2012), and generally focus only on GHG emission reduction. Few studies explicitly compare paludiculture to intact hydrological conditions. Given that the total carbon budget of intact peatlands is negative or neutral, comparing paludiculture against hydrologically intact systems is important as it provides a clear baseline to (1) gauge the performance and progress of paludiculture projects, and (2) set goals for sustainable peatland management.

Intact peatlands have two layers – the acrotelm and catotelm – responsible for regulating water storage, discharge and carbon sequestration (Joosten and Clarke, 2002). While this distinction is not as pronounced in tropical peatlands, there is still a relatively free draining upper layer and slower draining lower layer (Baird et al., 2017). The upper layer is rapidly destroyed when peatlands are drained, requiring an extensive period to reform after rewetting (Graf et al., 2012; Joosten and Clarke, 2002). It is currently not known whether rewetted peatlands can maintain water storage and discharge capacity comparable to intact peatlands (Baird et al., 2017; Graf et al., 2012; Günther et al., 2020; Price et al., 2003; Waddington and Price, 2000). Holden et al. (2011) investigated mean water table depth and variability, and responses to precipitation between drained, rewetted, and intact blanket peatlands. They found that the rewetted site exhibited hydrological characteristics that fall between the drained and intact peatlands six years post-rewetting, suggesting that complete hydrological restoration, if possible, is likely to take a long time. Similarly, observations of hydrologically restored peatlands in Indonesia found that these ecosystems lack the ability to regulate seasonal fluxes in groundwater table levels immediately post-rewetting, resulting in the inundation and mortality of planted seedlings (Page et al., 2009; Wösten et al., 2006). Strategies to reduce flooding risks, such as planting on mounds (Lampela et al., 2018), and how vegetation growth overtime feed into the hydrology of peatlands, need to be investigated in the context of establishing and maintaining paludiculture systems.

Aside from affected hydrology, microbial activities also differ between restored and natural sites based on nutrient availability, vegetation succession types, water table, land use, drainage age, and peat thickness (Andersen et al., 2010; Mishra et al., 2014). Accelerated decomposition was found in a newly restored ombrotrophic peat in Canada where the addition of new organic matter and low shrub cover favor higher microbial activity (Andersen et al., 2010). Similarly, mixed-crop plantations were found to house more diverse bacterial community than oil palm plantations, degraded PSF, and settlements (Mishra et al., 2014). Contrary to the northern peatlands, sample measurements of microbial biomass and enzyme activities in reforested and intact peatlands in Central Kalimantan showed lower rates of decomposition at the top 0–3 cm of peat in the reforested site (Könönen et al., 2018). The authors speculated that reductions in decomposition were linked to shifts in vegetation communities towards younger trees with less litter inputs. As the trees matured, greater litter inputs stimulated more decomposition, at the same time contributing more decomposition-resistant woody materials to the deeper layers of peat. As such, peat accumulation in the tropics may take a shorter period to reinstate compared to temperate sites. These findings echo our earlier concerns regarding the distinctions between peatland hydrological conditions and potentially expanding beyond a narrow focus on the hydrology when setting baselines for implementing paludiculture research.

2.4. Endemicity of vegetation selected for paludiculture

The third theme concerns the selection of vegetation for paludiculture. Vegetation plays an important role in maintaining ecosystem carbon balance through the regulation of carbon fluxes, net primary productivity, and decomposition rates (Chen and Chen, 2018; Dieleman et al., 2015; Fujii et al., 2017; Joosten and Clarke, 2002; Ren et al., 2019; Ward et al., 2015; Yule and Gomez, 2009). Scaling up to the landscape-level, vegetation communities can influence the overall ecosystem resilience towards external stressors. The identification and selection of vegetation is thus a popular topic for many paludiculture-related research (Giannini et al., 2017; Ren et al., 2019; Silvestri et al., 2017). A common criterion for selection is the tolerance to waterlogged conditions (Giannini et al., 2017; Ren et al., 2019; Surahman et al., 2018; Vroom et al., 2018). Peat soils are generally high in acidity and nutrient-poor (with notable exceptions such as mineral-rich fens), posing additional challenges for plant growth. This is reflected in low ecosystem diversity and site-level species diversity in intact tropical PSF as opposed to forests grown on mineral soils (Giesen et al., 2018; Posa et al., 2011). It is therefore assumed that native species are more adapted to peatlands conditions and should only be considered as candidate species for paludiculture (Tata, 2019). Nonetheless, this is sometimes not in line with the economic needs of stakeholders due to low productivity and high seedling mortality (Harrison et al., 2020).

A contention of vegetation selection for paludiculture is the use of native versus non-native species. Utilizing non-native vegetation risks creating novel conditions that may not adequately support other native biodiversity. A known example is the endangered aquatic warbler (*Acrocephalus paludicola*) which is adapted to living in fen sedges. Although the species can survive in commercial reedbeds, the density of reeds and high water table (>10 cm) needed to maintain the system are unsuitable for nesting warblers (Tanneberger et al., 2009).

Another caveat of selecting non-native species is suboptimal growth under natural peatland conditions, that is, a discrepancy between potential and realized yield. Many plant species are capable of growing under wet conditions, but with reduced yield and shortened lifespan. For example, sago (*Metroxylon sagu* Rottb.), touted as a crop candidate for paludiculture (Giesen and Sari, 2018), is a type of palm species harvested for its starch in the tropics. But sago plant has lower yields of 181–816 g m^{-2} when grown on peatlands as opposed to growing on clay soil (yields ~1361 g m^{-2}; Ming et al., 2018). Given that sago requires constant nutrient input, it can only be grown along the riverine edges of PSF (Flach, 1997; Jong, 2001; Okazaki and Sasaki, 2018). Such ecological barriers limit the geographical extent on which sago can be cultivated. Other non-native species such as pineapple (*Ananas comosus* L. Merr.) and coffee (*Coffea liberica* Hiern.) have been used as paludiculture crop candidates, although these are not considered 'true' paludiculture, as they require some level of drainage to grow and do not contribute to peat formation (Giesen, 2013).

Selecting and cultivating native PSF species are further hampered by the lack of knowledge regarding the complex relationship between vegetation communities and the chemical and hydrological properties of peatlands. There are over 1400 plant species documented in Southeast Asian PSF, many of which lacked detailed life history information (Posa et al., 2011; Giesen, 2013; Giesen et al., 2018; Graham, 2009). Where peatlands have been rewetted, key information such as seasonal water table fluctuation can be lacking. Planting seedlings which are not tolerant to constant flooding in newly rewetted peatlands can lead to mass diebacks during the initial stages of paludiculture establishment (Giesen and van der Meer, 2009; Lampela et al., 2018). The same issue may also be observed in temperate peatlands following rewetting (Couwenberg et al., 2011). The temporary introduction of non-native species may therefore be needed to gradually shift degraded environments towards conditions which favor and initiate ecosystem processes that enable native PSF species growth (Wibisono and Dohong, 2017; Wijedasa et al., 2020). This has been observed in northern peatlands, where initial planting of non-native species creates more suitable microclimates for native vegetation growth (Pouliot et al., 2015). In Sumatra, rapid natural regeneration with similar species richness to an adjacent intact PSF was found below five-year-old unharvested *Acacia crassicarpa* Benth. plantations (Wijedasa et al., 2020). Crucially, while natural regeneration of *A. crassicarpa* occurred further away from the forest, it was absent where water table was high, suggesting that non-native species under the right conditions may catalyze natural regeneration (Wijedasa et al., 2020).

A proposed strategy for the tropics is to use the framework species method for tree species selection, which simultaneously yields multiple ecological and social functions (Graham et al., 2017). The method calls for planting a combination of vegetation that fulfils the following characteristics: native species, fast growing species to establish structure and function, flowering and fruiting species to attract pollinators and dispersers for propagation, pioneer and climax species, species tolerant of external disturbances, and species that possess economic values (Elliott et al., 2003; Wijedasa et al., 2020). Integrating tree planting into more complex agri-sylvofishery or agri-silviculture systems may also yield short-term economic and environmental benefits, although further studies are needed to assess their long-term impacts (Budiman et al., 2020). Future research into paludiculture should explore the selection of vegetation communities instead of individual species to restore the socioeconomic and ecological functions of peatlands (Taylor et al., 2019). This approach will need careful consideration and monitoring as some non-native fast-growing pioneer trees could be invasive (Lugo, 1997; Parotta, 2012). Aside from the ecological limits of paludiculture crops, vegetation selection is greatly influenced by their economic values as agriculture crops, sources of NTFPs, or biofuel materials. This is especially important if paludiculture systems are used to replace more ecologically-destructive land uses, and surrounding communities rely on peatland products for their livelihoods.

2.5. Vegetation selection for economic returns

Products from northern peatlands are commonly used for bioenergy purposes (Hytönen et al., 2018). Bryophytes, perennial graminoids and woody vegetation are often selected for paludiculture studies. Common species include *sphagnum* (Gaudig et al., 2017), common reed (*Phragmites australis* L.) (Joosten et al., 2016; Ren et al., 2019), giant reed (*Arundo donax* L.) (Ren et al., 2019; Tho et al., 2017), and cattail (*Typha latifolia* L.) (Vroom et al., 2018). Woody species such as Canadian poplar (*Populus* × *canadensis* Moench. var. Oudenberg), white willow (*Salix alba* L. var. Dimitrios) (Giannini et al., 2017), and black alder (*Alnus glutinosa* L. Gaertn.) (Huth et al., 2018) are also frequently used. Many of these species are generalist wetland species that tolerate a wide range of environmental conditions with physiological adaptations, such as aerial roots (Joosten and Clarke, 2002; Melts et al., 2019; Ren et al., 2019).

Considerations about the quantity and quality of biomass production dominate most paludiculture research in the northern hemisphere. Evaluations of plant species for their chemical composition yield a selection of species suitable for bioenergy material. For example, cattail contains low concentrations of S and Si, and high concentration of Ca in its tissues, characteristics suitable for high-quality combustion (Ren et al., 2019). Species such as the giant and common reeds may produce metal-enriched ash post-combustion used for fertilizer purposes (Wichtmann and Wichmann, 2011), although high ash and alkali metal content can also increase the likelihood of boiler corrosion and slagging during combustion (Melts et al., 2019). In addition, the timing of biomass harvest and management method may affect the paludiculture crop's mineral content and GHG emission. For example, peatlands planted with reed canary grass under a fertilized two-cut harvested system may emit 10 times the N_2O compared to one-cut and unfertilized two-cut systems (Kandel et al., 2013). Wichtmann and Wichmann (2011) therefore suggested several principles to ensure the sustainable production of biomass on peatlands. They are the preservation and protection of: important carbon sinks, food production and local biomass usage, biodiversity, ecosystem functions, and local community livelihoods.

Compared to the north, vegetation selection studies in the tropics took a socioeconomic approach (Sitepu et al., 2017). Given the widespread development of tropical peatlands for smallholder and industrial agriculture and forestry, the economic profitability of paludiculture plays a heavy role in influencing vegetation selection and management decisions. Cultivating native tree species may be undesirable due to low marketability and extended harvesting age. Sago takes nearly a decade to reach harvestable age and can only be grown in a semi-cultivated state (Bintoro et al., 2018), rendering it difficult for large-scale production. Planting native PSF species is likely to yield a low return on investment unless their unproductive period is accounted for in economic valuations (Schulz et al., 2019b).

Preliminary accounting of paludiculture outputs using the economic internal rate of return (IRR), benefit-to-cost ratio (BCR), and net present value (NPV) indicate that there is potential for paludiculture to generate positive returns (Table 4). Sago is commonly viewed as a paludiculture crop in Southeast Asia with a history of local use, commercialization and trade dating back to before the 1980s (Jong, 2018; Joosten and Clarke, 2002; Ming et al., 2018). Most sago plantations are managed in semi-wild conditions near the rivers in areas such as Sarawak, Malaysia (Bintoro et al., 2018; Naim et al., 2016), Maluku Islands (Girsang, 2018), New Guinea (Pue et al., 2018), and Riau, Indonesia (Jong, 2001, 2018). Historically, sago from Borneo was exported to Singapore whereas currently, it is being exported to Malaysia from the Meranti Islands in Sumatra for making noodles (Butler, 2010). Along with gutta-percha, a latex obtained from the tree *Palaquium gutta*, the two products comprised up to 26% of the total trade between the two regions in 1852–1853, and as much as 56% of the trade in 1865–1866 (Cleary, 1996). Economic analyses of sago indicate that the crop has market potential. The IRR for sago ranges from 12.9–19% and increases to 17.4–19.2% when environmental benefits are accounted for (Chew et al., 1998; Md Isa and Mohayidin, 1999). A similar value was reported by Ming et al. (2018) where the IRR of sago was 13% on shallow peat.

Another common paludiculture crop, jelutong, is capable of growing on peat soils of varying depths (Giesen and Sari, 2018) and is valued for its sap. Jelutong has a long trade history in Indonesia, dating as far back as the mid-19th century. Tariffs on NTFP in 2006 and sanctions on illegal tappers later stifled the trade of jelutong latex (Tata et al., 2016). Recorded net present value (NPV) for jelutong were Rp 0.17–3.26 billion per km^2 in monoculture plantations and 925–980 million per km^2 for agroforestry systems (Tata et al., 2015). When compared to other crops (e.g. smallholder oil palm) planted on peatlands, jelutong has a low NPV and high return to labor, suggesting that planting jelutong monoculture is not efficient (Sofiyuddin et al., 2012).

Based on the NPV, IRR, and BCR, paludiculture may generate a small margin of economic benefit. Yet there are two main caveats: the economic benefits of paludiculture to conventional drainage-based crops are still too low, and not all paludiculture costs and externalities are documented in the economics analyses. For example, depending on the technology used to process sago, the cost of using modern technology such as large machines can be 17 times that of semi-modern technology such as wage labor and smaller processing machines, thereby impeding the upscaling of sago production (Girsang, 2018). Other costs such as the purchase of seedlings (Tata and Susmianto, 2016) and obstacles such as changing existing negative perspectives of sago as a ‘poor man's food’ (Pue et al., 2018) prevent the widespread adoption of paludiculture. Future studies should aim to capture environmental externalities and assess paludiculture relative to other production systems in order to outline the trade-offs for better decision-making.

3. Discussion and conclusion

Paludiculture is theoretically a viable option for sustainable peatland management in the tropics. Early research directions have taken divergent pathways in northern and tropical peatlands. The former is focused on experimenting with various parameters of paludiculture systems and practices to optimize ecosystem services benefits. This is made possible as lower population and economic pressures have resulted in many northern peatlands being abandoned and restored, allowing the quick revegetation of wetland species. Such conditions are not found in the tropics, where growing populations and economic development has resulted in most peatlands undergoing active use with only limited areas placed under strict conservation status. As such research on paludiculture in the tropics tends towards more exploratory approaches: investigating existing agroforestry systems to determine whether they could be considered paludiculture and interviewing stakeholders to obtain their perceptions of paludiculture.

Based on the current body of knowledge and the three themes we identified, we propose a definition for paludiculture in the tropics: a form of wet agriculture on rewetted peatlands using native PSF with long-term prospects of transforming the ecosystem into carbon-neutral or negative sinks, while providing socioeconomic benefits. We further propose several future directions of research for tropical paludiculture (Table 5). We noted that paludiculture seeks to provide and enhance multiple ecosystem services. To this end, the production of useful biomass and carbon regulation has featured prominently in research. However, a key point is that paludiculture systems must be carbon-neutral or negative in the intermediate- to long-term in order to prevent future peatland loss. In the case of carbon-positive systems, such as agroforestry on partial drainage systems, these should only be considered as sustainable in the short-term, ideally followed by eventual transition into full paludiculture. We suggest that a focus on carbon will help convene multiple aspects of paludiculture as carbon dynamic is closely tied to peatland hydrology and vegetation, and provide a reference for other paludiculture ecosystem services in terms of translating services into market schemes or policies. This is because carbon has traditionally received greater attention in climate-related policies that resulted in the development of numerous market instruments. Many of these market instruments function by levying a price on carbon emitted

Table 4
Economic valuation of sago and jelutong. No inflation adjustments were made (NPV = net present value; BCR = benefit-cost ratio; IRR = internal rate of return).

Crop	Location	NPV (USD per km^2)[a]	BCR	IRR (%)	Reference
Sago	Sarawak	N.A.	N.A.	13.1 (shallow peat of <2.5 m) 23.4 (mineral soil)	(Ming et al., 2018)
Sago	Sarawak	38,819 121,032 (including environmental benefits)	N.A.	12.9–13.8 17.4–19.2 (including environmental benefits)	(Chew et al., 1998)
Jelutong	Central Kalimantan, Indonesia	253,174 (monoculture) 590,738 (mixed planting)	7.88 (monoculture) 8.68 (mixed planting)	20 29	(Harun, 2014)
Jelutong	South Sumatra, Indonesia	992,465	3.85	9.97	(Ulya et al., 2015)
Jelutong	Central Kalimantan, Indonesia	17,064,014	2.89	24.22	(Mintarjo and Betlina, 2006)
Jelutong	Central Kalimantan, Indonesia	95,875 (monoculture) 567,579 (mixed planting)	4.28 (monoculture) 5.35 (mixed planting)	14.7 (monoculture) 24.1 (mixed planting)	(Budiningsih and Effendi, 2013)
Jelutong (monoculture)	Jambi, Indonesia	359,000	N.A.	N.A.	(Sofiyuddin et al., 2012)

[a] Historical currency exchange rate based on values reported in December of the year of publication (OFX, 2017).

Table 5
Future research directions for tropical paludiculture according to the key themes.

Theme 1. Ecosystem services benefits
• Assess synergies and trade-offs of peatland ecosystem services under paludiculture systems in comparison to other forms of peatland use. • Develop more holistic methods for measuring and valuing the ecosystem services – especially cultural ecosystem services – provided by paludiculture across spatial and temporal scales. • Examine the dynamics between hydrology, vegetation, and different GHGs (i.e. CO_2, CH_4, and N_2O) to produce a more complete picture of the carbon balance and GHG reduction potential under paludiculture. • Identify ways for translating ecosystem services provided by paludiculture into national and international policies and market instruments.
Theme 2. Hydrological conditions of paludiculture
• Conduct long-term monitoring of ecological processes and dynamics between the abiotic (e.g. groundwater table level) and biotic components (e.g. vegetation and microbial activity) of restored sites. • Conduct more comparative studies of the hydrological behaviors (e.g. seasonal fluctuations in groundwater table level) and properties (e.g. nutrient input) between paludiculture and intact PSF.
Theme 3. Vegetation selection
• Investigate how vegetation communities and processes of succession can be used to restore peatland ecological functions while providing economic profit in the long-term. • Develop better methods of factoring in environmental externalities of paludiculture systems. Additionally, more studies comparing paludiculture to other production systems with the aim of identifying the trade-offs of engaging in paludiculture is crucial for better decision-making. • Identify ways to upscale the marketability of various native PSF species. For example, certification for paludiculture products may increase the economic attractiveness of paludiculture. • Develop better understanding of the functions of different native PSF species such that the knowledge can be used to design complex paludiculture systems (e.g. framework species method). • Examine ways to integrate paludiculture with other forms of peatland production, such as silviculture and animal husbandry.

or rewarding avoided emissions. Therefore research on the better accounting of carbon dynamics and pathways in paludiculture systems is crucial for paludiculture impacts to be integrated into carbon market instruments.

In addition to regulating services, more holistic accounting for other paludiculture ecosystem services to highlight synergies and trade-offs of services is needed. Sole focus on one type of service can lead to the erosion of other services and the peatland ecosystem in general (Comberti et al., 2015). Better valuation for cultural services is important as tropical peatlands are also complex socio-ecological systems. Understanding how paludiculture systems and management practices align with and enhance cultural ecosystem services will increase its adoption, particularly at the community-level.

In regard to the second theme, we suggest that a clear distinction be made between management practices for rewetted and intact (i.e. wet) peatlands as they can have vastly different biogeochemical and physical characteristics. Intact PSF should be primarily preserved for conservation and light-weight use, such as the small-scale harvest of NTFP, with limited impacts to the ecosystem and ecological functions. Paludiculture practiced as an agriculture requiring active management is more suited for rewetted peatlands that are recovering hydrological, biogeochemical, and vegetation functions. To date long-term monitoring of the abiotic and biotic components of restored peatlands (and by extension, paludiculture) is greatly lacking in tropical regions. As tropical peatland processes take years to establish, ongoing and long-term monitoring is needed to provide better assessments of the impacts of paludiculture systems.

A recommendation for the third theme – vegetation selection – is that native PSF species should ideally be used in paludiculture as they are best adapted to intact peatlands. However, recognizing that substantial knowledge gaps exist for the cultivation of these species and that many of them are not suited towards the harsh conditions of newly rewetted peatlands, pioneer species or non-native species more tolerant of extreme environments may be planted initially to create more favorable environments for climax species. This entails a shift away from the current focus on selecting for individual plant species towards establishing complex vegetation communities with multiple ecological and economical functions. Current research shows that a few of the native PSF can be economically viable but requires substantial upscaling before being competitive to more destructive peatland uses. Future research into accounting for environmental externalities and increasing the values of paludiculture products will make paludiculture more appealing as a land use alternative.

In this review, we have identified key themes from existing paludiculture definitions and evaluated our current knowledge of tropical peatlands within the framework of these key themes. The ambiguity of the definitions suggests that there can be multiple realizations of paludiculture as an agriculture system, land use, or set of management practices. Irrespective, the implementation of paludiculture in the tropics should be guided by an understanding of the socioecological context and rooted in peatland science.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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