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REVIEW

Sphagnum mosses, the impact of disturbances and anthropogenic management actions on their ecological role in CO_2 fluxes generated in peatland ecosystems

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Abstract

Mosses of the genus *Sphagnum* are the dominant vegetation in most pristine peatlands in temperate and high-latitude regions. They play a crucial role in carbon sequestration, being responsible for ca. 50% of carbon accumulation through their active participation in peat formation. They have a significant influence on the dynamics of CO_2 emissions due to an efficient maximum potential photosynthetic rate, lower respiration rates, and the production of a recalcitrant litter whose decomposition is gradual. However, various anthropogenic disturbances and land use management actions that favor its reestablishment have the potential to modify the dynamics of these CO_2 emissions. Therefore, the objective of this review is to discuss the role of *Sphagnum* in CO_2 emissions generated in peatland ecosystems, and to understand the impacts of anthropogenic practices favorable and detrimental to *Sphagnum* on these emissions. Based on our review, increased *Sphagnum* cover reduces CO_2 emissions and fosters C sequestration, but drainage transforms peatlands dominated by *Sphagnum* into a persistent source of CO_2 due to lower gross primary productivity of the moss and increased respiration rates. Sites with moss removal used as donor material for peatland restoration emit twice as much CO_2 as adjacent undisturbed natural sites, and those with commercial *Sphagnum* extraction generate almost neutral CO_2 emissions, yet both can recover their sink status in the short term. The reintroduction of fragments and natural recolonization of *Sphagnum* in transitional peatlands, can reduce emissions, recover, or increase the CO_2 sink function in the short and medium term. Furthermore, *Sphagnum* paludiculture is seen as a sustainable alternative for the use of transitional peatlands, allowing moss production strips to become CO_2 sink, however, it is necessary to quantify the emissions of all the components of the field of production (ditches, causeway), and the biomass harvested from the moss to establish a final closing balance of C.

KEYWORDS
anthropogenic disturbances, climate change, CO_2 emissions, drainage, paludiculture, recolonization, restoration, *Sphagnum*

1 | INTRODUCTION

Peatland ecosystems store 30% of the organic carbon (C) accumulated in the planet's soils (500–700 Gt C) (Leifeld & Menichetti, 2018; Turetsky et al., 2015; Yu et al., 2011), and during the Holocene, they have contributed to net climate cooling because peat accumulation has provided a global and persistent sink for carbon dioxide (CO_2) (Frolking & Roulet, 2007; Turetsky et al., 2015; Yu, 2011). They currently correspond to the terrestrial ecosystem that performs the largest sequestration of atmospheric C in the long term, storing a substantially greater amount of C than that captured in global forest biomass (Beaulne et al., 2021; Dunn & Freeman, 2011; Hugelius et al., 2020; Parish et al., 2008).

The C exchange between a peatland and the atmosphere (net ecosystem exchange, NEE), is the result of the net balance between CO_2 inputs from photosynthetic uptake (gross primary productivity, GPP), and its release driven by the decomposition of organic matter (OM) through plant respiration (autotrophic respiration) and soil biological community (heterotrophic respiration), which together represent the ecosystem respiration (R_{eco}) (Järveoja et al., 2018; Laine et al., 2012; Lunt et al., 2019; Rankin et al., 2022).

When considering plant composition in a peatland, the cover of mosses of the genus *Sphagnum* is considered relevant for C sequestration (Jassey & Signarbieux, 2019; Kasimir et al., 2021; Piatkowski et al., 2021; Thompson & Waddington, 2008), due to an efficient maximum potential photosynthetic rate, lower rates of respiration (Riutta et al., 2007), and continuous litter inputs whose degradation is slower than those produced by vascular plants (Glenn et al., 2006; Holl et al., 2019; Kuiper et al., 2014; Laine et al., 2012; Mazziotta et al., 2018; Purre et al., 2019), due to the presence of polyphenols, and different polysaccharides in their cell walls (Hájek et al., 2011; Norby et al., 2019; Taskila et al., 2016; Turetsky et al., 2008). In addition, they exhibit a high cation exchange capacity and a proven efficiency in water retention, generating an acidic, humid, and anaerobic environment (Clymo, 1963; Clymo & Hayward, 1982; Kotska et al., 2016; Leroy et al., 2019; Li et al., 2021). These conditions inhibit the action of decomposer extracellular enzymes (Hájek et al., 2011; Kim et al., 2021; Li et al., 2021; Nordström et al., 2022), slowing the degradation of OM and favoring C sequestration in the long term, estimated to be responsible for ca. 50% of C accumulation in peatlands (Piatkowski et al., 2021; Rydin & Jeglum, 2006).

Considering that *Sphagnum* is one of the dominant vegetation constituents in peatlands (ca. 50–90% of peat; Lunt et al., 2019; Newman et al., 2018; Turetsky, 2003), it is estimated that they can contribute to storing ca. 10% of the planet's soil organic C in the form of semi-decomposed litter (Clymo, 1987; Laing et al., 2014; Robroek et al., 2009; Turetsky, 2003), therefore, the responses of peatlands to future environmental changes and anthropogenic disturbances may be dictated by their functionalities and characteristics (Mazziotta et al., 2018; Turetsky et al., 2015).

Despite the significant ecosystem services provided by peatlands, it is estimated that 15–20% of their global extent has experienced degradation processes due to anthropogenic disturbances (Joosten, 2009; Joosten & Clarke, 2002; Ma et al., 2022; Parish et al., 2008; Urák et al., 2017; Waddington et al., 2015), generating 5–10% of annual global anthropogenic CO_2 emissions (Friedlingstein et al., 2014; Loisel & Gallego-Sala, 2022). Human pressures have transformed the biogeochemical functionality of peatlands from a C sink in the long term to CO_2 emitters in the short term (Evrendilek, 2014; Munir et al., 2015; Page et al., 2022). Anthropogenic disturbances that directly affect *Sphagnum* mosses are related to drainage, commercial extraction, or removal of the moss as donor plant material for restoration of degraded peatlands, which have the potential to affect their ecological functionality and modify C dynamics. However, land use management actions favorable for their restoration, such as assisted reintroduction, recovery of appropriate conditions for their natural recolonization, or *Sphagnum* paludiculture could counteract the negative impacts of degradation.

Based on what was previously mentioned, we have considered it relevant to address, through this review, how *Sphagnum* mosses influence and play a role in CO_2 emissions generated in peatlands affected by anthropogenic disturbances and by management actions that foster their restoration. Therefore, the objectives of this review are (i) to systematize and summarize the existing knowledge about the role and influence of *Sphagnum* mosses on CO_2 fluxes in peatland ecosystems, (ii) to enrich the understanding of the impacts of different disturbances and favorable land use management practices on *Sphagnum* mosses, and their effects on CO_2 fluxes in peatland environments, and (iii) to provide guidance for the development of future research based on the identified knowledge gaps.

2 | DIRECT ANTHROPOGENIC DISTURBANCES EXERTED ON *SPHAGNUM* MOSSES AND THEIR IMPACTS ON CO_2 FLUXES IN PEATLAND ECOSYSTEMS

Disturbances in peatland ecosystems are defined by Connolly and Holden (2011) as "any natural or anthropogenic process that interrupts the natural growth trajectory of these environments", affecting their hydrology, vegetation, and C stocks. In this context, the main anthropogenic disturbances on peatland ecosystems are presented in Figure 1.

In turn, anthropogenic disturbances that directly influence the ecological functionality of *Sphagnum* are mainly linked to drainage,

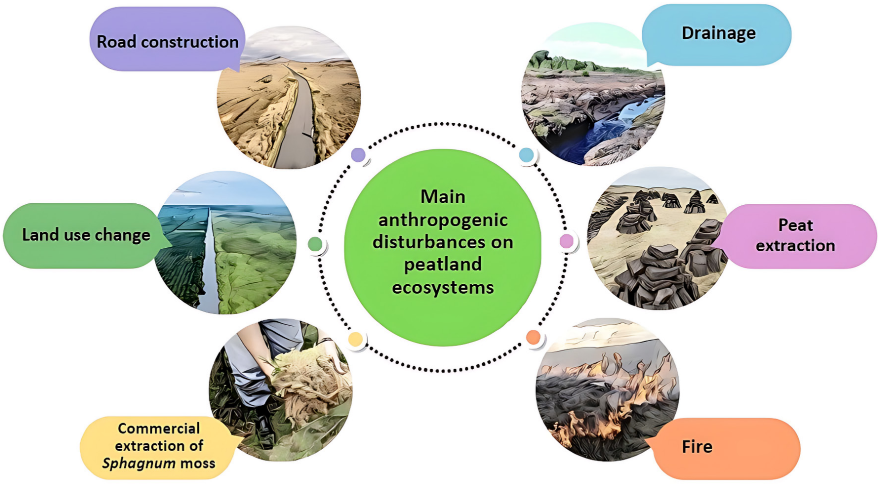

FIGURE 1 Most relevant anthropogenic disturbances affecting the conservation and maintenance of ecosystem services provided by peatland ecosystems.

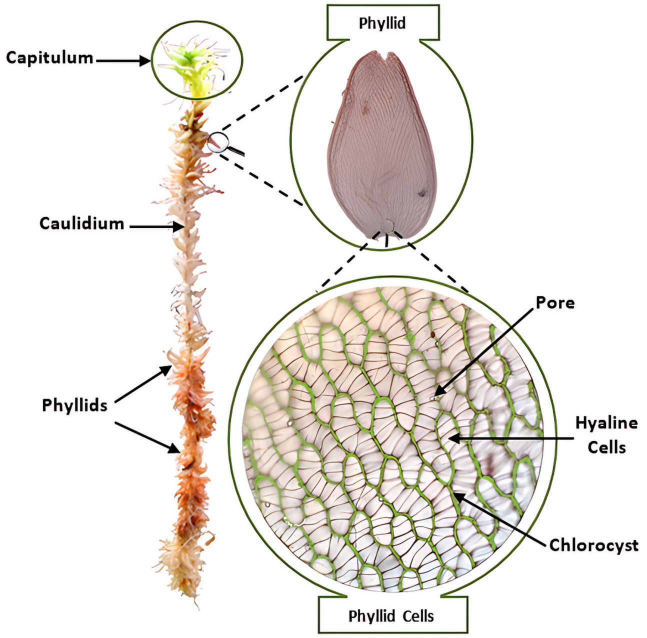

FIGURE 2 Morphology and basic anatomy of *Sphagnum* (*S. magellanicum*) considering the whole plant, the phyllids and component cells of a phyllid. Photosynthetic cells (chlorocysts) and hyaline cells that contribute to maintaining the water status of the moss are differentiated. Modified from Weston et al. (2015).

due to the fact that the moss requires a high and stable water table (WT) to maintain moisture in the apical zone of the caulidium (called capitulum; Figure 2), and to be able to activate photosynthesis (Ketcheson & Price, 2014; McNeil & Waddington, 2003; Robroek et al., 2009; Strack & Price, 2009; Tuittila et al., 2004). In addition, the total or partial extraction of the moss stratum can be used as donor plant material in the restoration of degraded peatlands (Murray et al., 2017), or for commercialization and export (Díaz & Silva, 2018; Silvan et al., 2017; Valdés-Barrera et al., 2019), considering different uses and applications where the biomass and the functional properties of moss can be used, highlighting its use as a substrate for plant production (horticulture and orchid production; Gaudig et al., 2013; Krebs et al., 2017; Oberpaur et al., 2010; Taskila et al., 2016; Whinam et al., 2003; Zegers et al., 2006).

2.1 | The impact of drainage and its influence on CO_2 emissions in peatlands dominated by *Sphagnum* mosses

Hydrological alteration generated by drainage is one of the most generalized anthropogenic disturbances and the one that produces the greatest damage on a peatland because primary productivity and OM degradation are closely linked to WT position (Evans et al., 2021; Huang et al., 2021; Laiho, 2006; Loisel et al., 2021; Strack & Price, 2009). The decline in WT generally alters the dynamics and C balance of the ecosystem (Chimner et al., 2017; Goodrich et al., 2015; Minkkinen et al., 1999; Strachan et al., 2016), because increased oxygen availability in the unsaturated surface stratum (acrotelm) favors respiration and oxidation of OM, increasing CO_2 emissions (Figure 3) (Huang et al., 2021; Ma et al., 2022; Peichl et al., 2014; Strack et al., 2014; Swails et al., 2022). On the other hand, anaerobic conditions fostered by elevated WT favor the accumulation of recalcitrant compounds derived from *Sphagnum* by inhibiting OM degradation through a mechanism known as "enzymatic latch" (Abbott et al., 2013; Freeman et al., 2001; Kim et al., 2021). Several studies have suggested that C losses generated by WT reduction depend essentially on the quality of the substrate (Hogg et al., 1992; Laiho, 2006; Limpens & Berendese, 2003), that is, on its ease to being degraded, a condition limited for the substrate formed by *Sphagnum* litter due to its higher resistance to decomposition. In turn, it has been suggested that in temperate peatlands dominated by *Sphagnum*, WT depth is the most important factor influencing litter decomposition (Górecki et al., 2021).

According to Rankin et al. (2022), most of the variability in CO_2 exchange in a peatland comes from changes in GPP and autotrophic respiration. Therefore, because CO_2 sequestration or emissions are generated from the balance between GPP and R_{eco} (Bragazza et al., 2009; Gažovič et al., 2013; Huang et al., 2021), the impact of

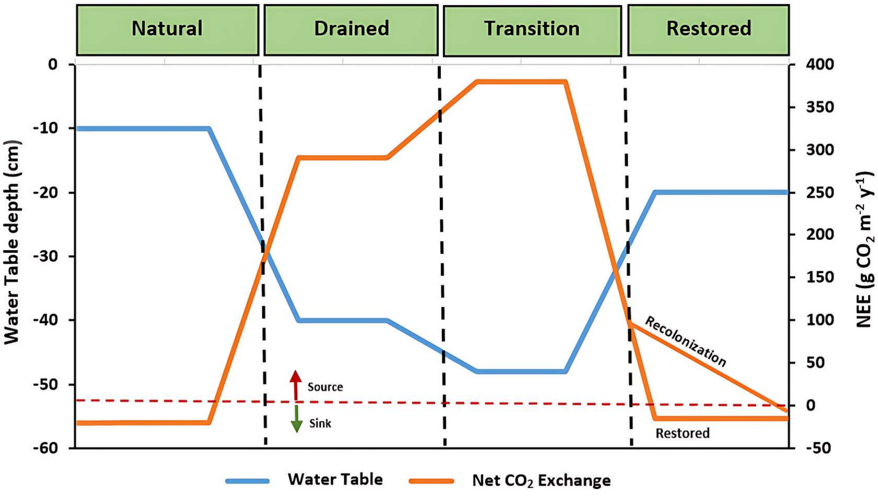

FIGURE 3 Conceptual model illustrating and comparing pristine natural sites with sites subject to the effects of drainage, abandonment (transitional peatlands), and restoration on water table behavior and net ecosystem exchange ($g\,CO_2\,m^{-2}\,year^{-1}$). Modified from Waddington & Price (2000).

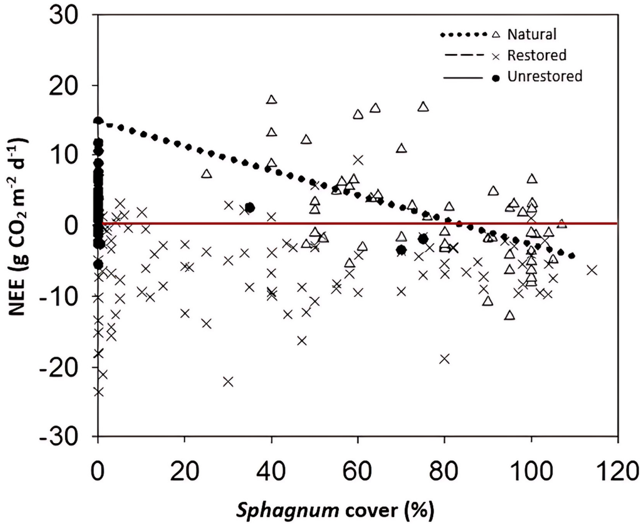

FIGURE 4 Net ecosystem exchange ($g\,CO_2\,m^{-2}\,day^{-1}$) versus *Sphagnum* cover levels for natural, moss layer transfer restored (MLTT) sites, and unrestored sites in peatlands located in eastern (Quebec) and western (Alberta) Canada. Negative values indicate CO_2 uptake by the ecosystem. Obtained from Strack et al. (2016).

drainage and the influence of *Sphagnum* mosses on CO_2 emissions will be addressed by considering an approach on both components.

We follow the atmospheric sign convention to indicate the CO_2 fluxes or NEE magnitudes reported in our paper, where negative values indicate CO_2 uptake by the ecosystem, while all fluxes from the ecosystem to the atmosphere are defined as positive. In turn, WTs are indicated by negative values when these are below the surface.

2.1.1 | Impact of drainage on gross primary productivity (GPP) of *Sphagnum* mosses and its effects on CO_2 emissions in peatland ecosystems

Sphagnum is a key vegetation component in many peatlands because CO_2 uptake increases when a higher abundance of moss is present (Brown et al., 2017; Järveoja et al., 2016; Laine et al., 2016; Purre et al., 2019), generating a net CO_2 sink condition with coverage of ca. 75% to 80% (Figure 4) (Glenn et al., 2006; Strack et al., 2016; Swenson et al., 2019), which decreases after persistent WT reduction, significantly affecting the C balance (Laiho, 2006).

The relationship of *Sphagnum* with WT is subject to its condition as a plant lacking conductive tissues, therefore, they depend on water transport by capillarity from the WT to the capitulum for CO_2 absorption, that is, they require a high and stable WT to reduce capillary stresses, which is why they are highly sensitive to water level reductions (Bengtsson et al., 2020; Price et al., 2003; Robroek et al., 2009; Strack et al., 2009; Strack & Price, 2009; Thompson & Waddington, 2008). In general, peat soils have a capillary fringe that reaches the surface when the WT is located between −30 and −40 cm (Laiho, 2006; Price & Whitehead, 2001), therefore, a deeper WT will decrease the moisture content in the moss by reducing the effectiveness of capillary transport to the surface (Price & Whitehead, 2001; Strack et al., 2009; Waddington et al., 2015). Despite this, *Sphagnum* tissues contain dead and empty cells when they are functionally mature (hyaline cells) (Kremer et al., 2004), with the capacity to store water within the phyllids, managing to maintain the hydration conditions required for photosynthesis (Weston et al., 2015) (Figure 2). It has been indicated that the maximum photosynthetic rate (P_{max}) in *Sphagnum* starts to decrease once the water content in the capitulum reaches 85% (Jassey & Signarbieux, 2019; Taylor et al., 2016), a condition that could be generated after ca. 30 days of continuous drought (Lees et al., 2019). In turn, Robroek et al. (2009) determined that *Sphagnum* samples assimilated C after 16 days with high WT after 23 days with drought treatment, but not to the degree of assimilation prior to desiccation.

Sphagnum photosynthesis significantly influences C sequestration in a peatland, as ca. 16% of the annual net C fixed by this moss genus remains stored in the peat (Jassey & Signarbieux, 2019). In turn, it has been indicated for an oligotrophic and minerotrophic peatland (poor fen) located in northern Sweden, that the moss stratum can contribute ca. 70% of the GPP of the aerial portion (Järveoja et al., 2018), and ca. 30% of the average seasonal gross photosynthesis in a poor fen in southern Finland (Riutta et al., 2007). However, water stress caused by drainage reduces *Sphagnum* P_{max} due to decreased volumetric moisture content of the moss, limiting net CO_2 uptake and GPP, which may contract the CO_2 sink (Adkinson & Humphreys, 2011; Jassey & Signarbieux, 2019; O'Neill et al., 2022; Robroek et al., 2009; Sulman et al., 2010). In this regard, Riutta et al. (2007) report that decreases in WT by −14 and −22 cm caused decreases of 14% and 22%, respectively, in gross photosynthesis in an ombrotrophic peatland (bog) in southern Finland, due to reductions in *Sphagnum* photosynthesis over those exhibited in sedges and shrubs. Experimental studies by McNeil and Waddington (2003) with *Sphagnum* cores subjected to a 7-day drying period increased moss GPP from 2.2 g $CO_2\,m^{-2}\,day^{-1}$ to a peak of 3.2 g $CO_2\,m^{-2}\,day^{-1}$ with a volumetric water content (VWC) of 28%, decreasing sharply to 0.3 g $CO_2\,m^{-2}\,day^{-1}$ with a VWC of 6%, managing to recover pre-drying GPP levels after 20 days of saturation (VWC = ca. 55%).

Several studies have indicated that bogs dominated by *Sphagnum* can become a persistent source of CO_2 due to a reduction in moss GPP driven by the decline in WT (Strachan et al., 2016; Strack et al., 2009; Wu & Roulet, 2014; Zhao et al., 2016). In this regard, McNeil and Waddington (2003) state that a bog drained in Quebec (Canada), with a WT of −31.3 cm, recorded decreases of 59% in GPP (from 6.9 to 2.8 g $CO_2\,m^{-2}\,day^{-1}$) compared to surrounding sites with a WT of −21 cm, causing a sink condition only when WT remained elevated. On the other hand, continuous 12-year records of NEE and GPP measurements in a poor fen dominated by *Sphagnum* in Sweden indicate that the second lowest annual GPP rate (283 g C m^{-2}) was adjusted with the deepest average annual WT recorded over the period (−20.8 cm), generating the lowest CO_2 sink over the 12 years (Peichl et al., 2014).

Opposite results are reported by Kokkonen et al. (2022) for a poor fen, a mesotrophic fen, and a bog in Finland, where decreases in WT between −15 and −20 cm at all sites generated increases in moss P_{max} in the fens ecosystems, especially in the mesotrophic fen. On the other hand, experiments with *Sphagnum* mesocosms obtained from different microtopographic positions of a bog located in Michigan (USA) showed that, under drainage conditions (WT = −25 cm), the GPP of mosses established in hollows was more affected than those forming hummocks, increasing CO_2 emissions in the former (O'Neill et al., 2022). Hummock-forming mosses offer greater resistance to maintain GPP under drainage conditions thanks to a high capitulum density and increased capillary transport efficiency in water retention, allowing them to respond favorably to water stress, while retaining stability in C fluxes (Adkinson & Humphreys, 2011; Bengtsson et al., 2016; O'Neill et al., 2022; Robroek et al., 2009). However, O'Neill et al. (2022) demonstrated that they can become sources of CO_2 when low WT is combined with an intense or repetitive seasonal drought condition.

In general, *Sphagnum* gross photosynthesis exhibits a unimodal response to WT variation (Tuittila et al., 2004), in which case different studies indicate that its optimal position should be maintained between −5 and −20 cm to maximize moss GPP (Brown et al., 2017; Riutta et al., 2007; Strack et al., 2009; Tuittila et al., 2004) (Figure 5). According to Tuittila et al. (2004), *Sphagnum* photosynthesis is restricted above an optimum WT of −12 cm due to the reduction of water available for metabolic processes; however, a very high WT can saturate the moss with water, reducing the rate of gas diffusion toward the chlorocysts, limiting CO_2 fixation. In this regard, Brown et al. (2017) state that there were no differences in moss GPP between sites with WT of −10 cm and surrounding areas with WT of −20 cm in a bog recolonized by *Sphagnum* species in Canada. On the other hand, sites with WT between −30 and −55 cm reduce the capillary force necessary for water transport, generating detrimental effects on *Sphagnum* GPP (McCarter & Price, 2015; Peichl et al., 2014; Price & Whitehead, 2001; Strack et al., 2009; Strack & Price, 2009). However, other research has indicated that *Sphagnum* does not limit its productivity and CO_2 sequestration when WT is at a depth of less than −40 cm (Ketcheson & Price, 2011; Taylor et al., 2016).

2.1.2 | Impact of drainage on ecosystem respiration (R_{eco}) in peatlands dominated by *Sphagnum* mosses and its effect on CO_2 emissions

The moss stratum seems to play an important role in the R_{eco} of a peatland, since according to Turetsky et al. (2008), it is the *Sphagnum* species and not the environmental conditions that allow controlling the early stages of peat decomposition, and the initial R_{eco} rates, a situation associated with their recalcitrant litter inputs. Other studies indicate that in ombrotrophic peatlands, the moss layer can inhibit the respiration of microorganisms in the underlying peat by reducing CO_2 fluxes (Rankin et al., 2022). Likewise, in bogs where a community with diverse functional types of plants is established, *Sphagnum* represents the lowest contribution to seasonal R_{eco} (15%), compared to those recorded in sedges (40%) and shrubs (45%) (Riutta et al., 2007).

Drainage of peatlands dominated by *Sphagnum* can generate increases in R_{eco} fluxes and increased CO_2 released by oxidation of labile OM and some of the recalcitrant residual peat derived from moss litter (Rankin et al., 2022; Valdés-Barrera et al., 2019; Wilson et al., 2016), due to decreases in their phenolic concentrations, stimulation of bacterial growth, and increases in phenoloxidase enzyme activity (Abbott et al., 2013; Fenner & Freeman, 2011; Freeman et al., 2001). Furthermore, Tuittila et al. (2004) report that the respiration rate of *Sphagnum* reaches its maximum with a WT located at −12 cm, decreasing and stabilizing with deeper WTs. Moreover, Purre et al. (2019) indicate that bogs dominated by *Sphagnum* in Estonia with a WT between −45 and −50 cm explain a small part of the variation in R_{eco} (<3%), which increases substantially (40%) with a WT of −16 cm. However, R_{eco} rates were higher at sites with deeper WT (WT = −48; $R_{eco} = 128$ mg $CO_2\,m^{-2}\,h^{-1}$), than in areas with higher WT (WT = −16 cm; $R_{eco} = 98$ mg $CO_2\,m^{-2}\,h^{-1}$). According to Rankin et al. (2022), the relations between R_{eco} and WT increase when the latter is above −35 cm (less deep), while they break down below this level. On the other hand, Riutta et al. (2007) indicate that WT

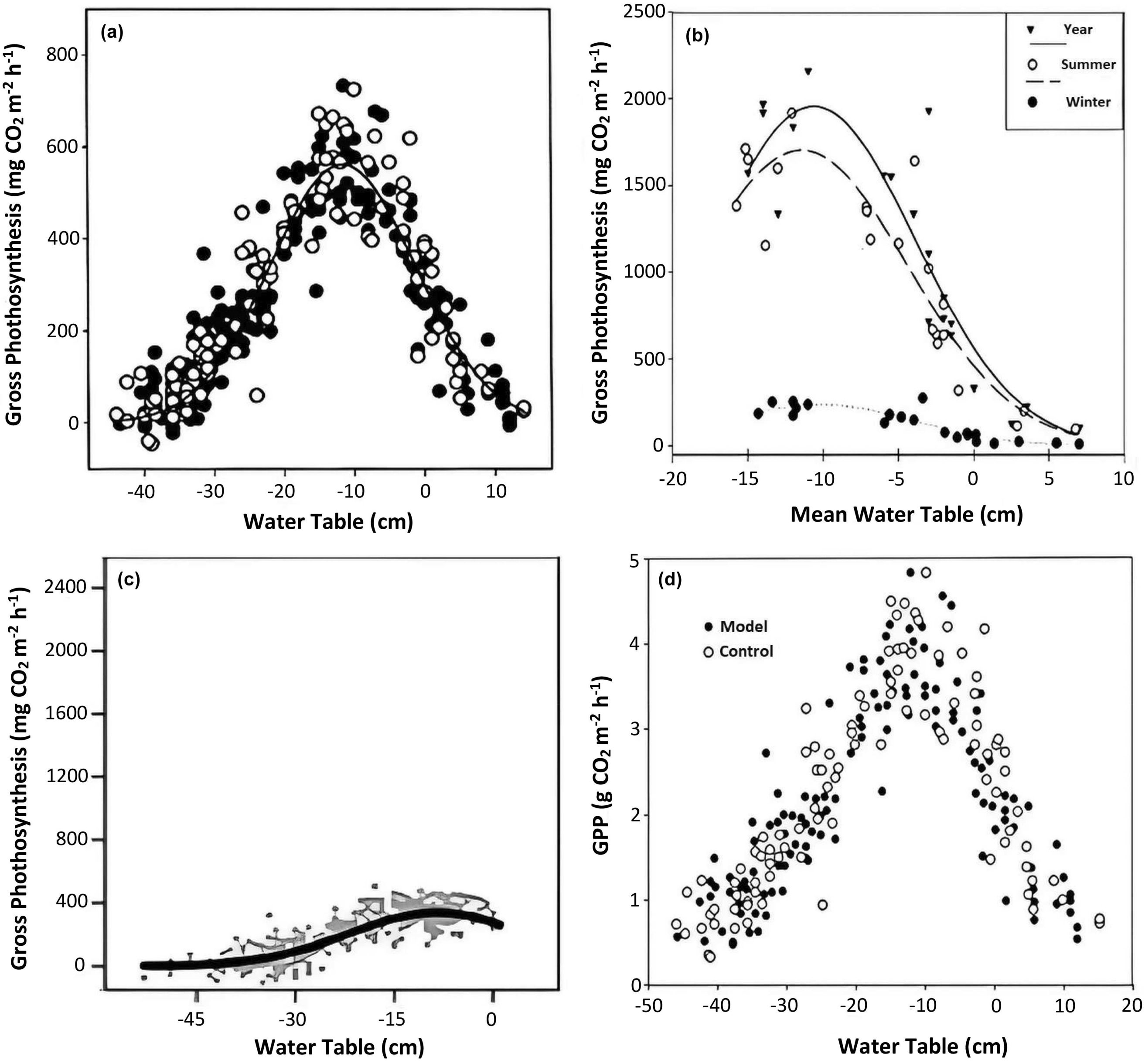

FIGURE 5 Relationship between gross photosynthesis/GPP of *Sphagnum* mosses (mg CO_2 m^{-2} h^{-1} or g CO_2 m^{-2} day^{-1}) and water table depth taken from various studies. (a) Tuittila et al. (2004); (b) Laine et al. (2007); (c) Riutta et al. (2007); (d) Thompson and Waddington (2008).

affects R_{eco} only in very humid conditions, and the impact is reduced when WT is below a threshold of −15 cm. Despite this information, maintaining a high and stable WT would allow less variability in R_{eco} due to the suppression of OM decomposition (Lund et al., 2010). In contrast, low and fluctuating WT can generate ca. 75% increases in moss litter decomposition due to increased fungal populations that are able to reduce their phenolic concentrations and activate hydrolase enzymes (Kim et al., 2021).

R_{eco} dynamics with respect to WT appear to be similar in peatlands located in the southern hemisphere, since according to the results obtained by Holl et al. (2019) for a bog located in the Argentine Patagonia dominated by *Sphagnum magellanicum* Brid., lower R_{eco} rates (339 ± 3 g CO_2-C m^{-2} $year^{-1}$) were associated with periods with higher WT, as opposed to deeper WT that drove increases in R_{eco} (392 ± 3 g CO_2-C m^{-2} y^{-1}) and to a lesser extent in net photosynthetic rates, generating a 72% decrease in CO_2 sequestration. A similar situation is described by Cai et al. (2010) for a fen with high *Sphagnum* cover in Canada, where an increase of −20 cm in WT depth increased R_{eco} by 52%, and photosynthesis only by 35%, raising CO_2 emissions by 200%.

Opposite results are reported by McNeil and Waddington (2003) for a bog subjected to hydrological restoration with a WT of −0.9 cm that presented a R_{eco} 10 to 25% higher than that of a drained surrounding area with a WT of −35.8 cm, a condition similar to that reported by Brown et al. (2017) for a bog restored with *Sphagnum*, where the mean R_{eco} was significantly higher in sites with WT of −10 cm than in those with WT of −20 cm ($t_{10.7}=3.7$, $p=.003$). In turn, Kokkonen et al. (2022) indicate that a WT decrease of −10 cm in three different peatland types in Finland did not affect *Sphagnum* respiration, although this was higher in a poor fen site than in a rich fen and a bog located in the same experimental area. Experimental studies using peat cores obtained from a bog in Michigan (the United States) have confirmed this dynamic, as WT decreases to −25 cm showed no effect on respiration of *Sphagnum* established in lawns and hummocks. However, CO_2 emissions increased significantly with lower WT in lawn mosses (ANOVA with a p-value = .0086) (O'Neill et al., 2022), due probably to a faster decomposition rate compared to mound mosses (Hájek, 2009; Taskila et al., 2016; Turetsky et al., 2008), a difference associated with higher relative amounts of metabolic carbohydrates in the former (lawns), and structural carbohydrates in the latter (hummocks) (Turetsky et al., 2008).

Finally, experimental studies conducted by Lees et al. (2019) with peat cores dominated by *Sphagnum* subjected to 80 days of drought that reduced WT, managed to decrease R_{eco} by 45% with respect to the initial record, however, after sudden rewetting R_{eco} increased by 340%, generating pulses of elevated CO_2 emissions; a situation also reported by McNeil and Waddington (2003) for *Sphagnum* cores kept in complete drought for 7 days, where subsequent rewetting increased peat and moss respiration from 472% to 1386%. In this sense, it has been suggested that R_{eco} pulses after sudden rewetting would be dominated by *Sphagnum* respiration (Strack & Price, 2009). However, studies that have used peat cores covered only by *Sphagnum* record higher CO_2 release during drainage (WT = -40 cm) than during slow rewetting (WT = -25 and -20 cm), due to increased R_{eco} associated with higher microbial activity (Blodau & Moore, 2003; Strack & Price, 2009).

2.1.3 | Final considerations associated with the impact of drainage on CO_2 emissions in peatlands dominated by *Sphagnum* mosses

Previous research results allow to establish a general preliminary consensus that drainage of peatlands dominated by *Sphagnum*

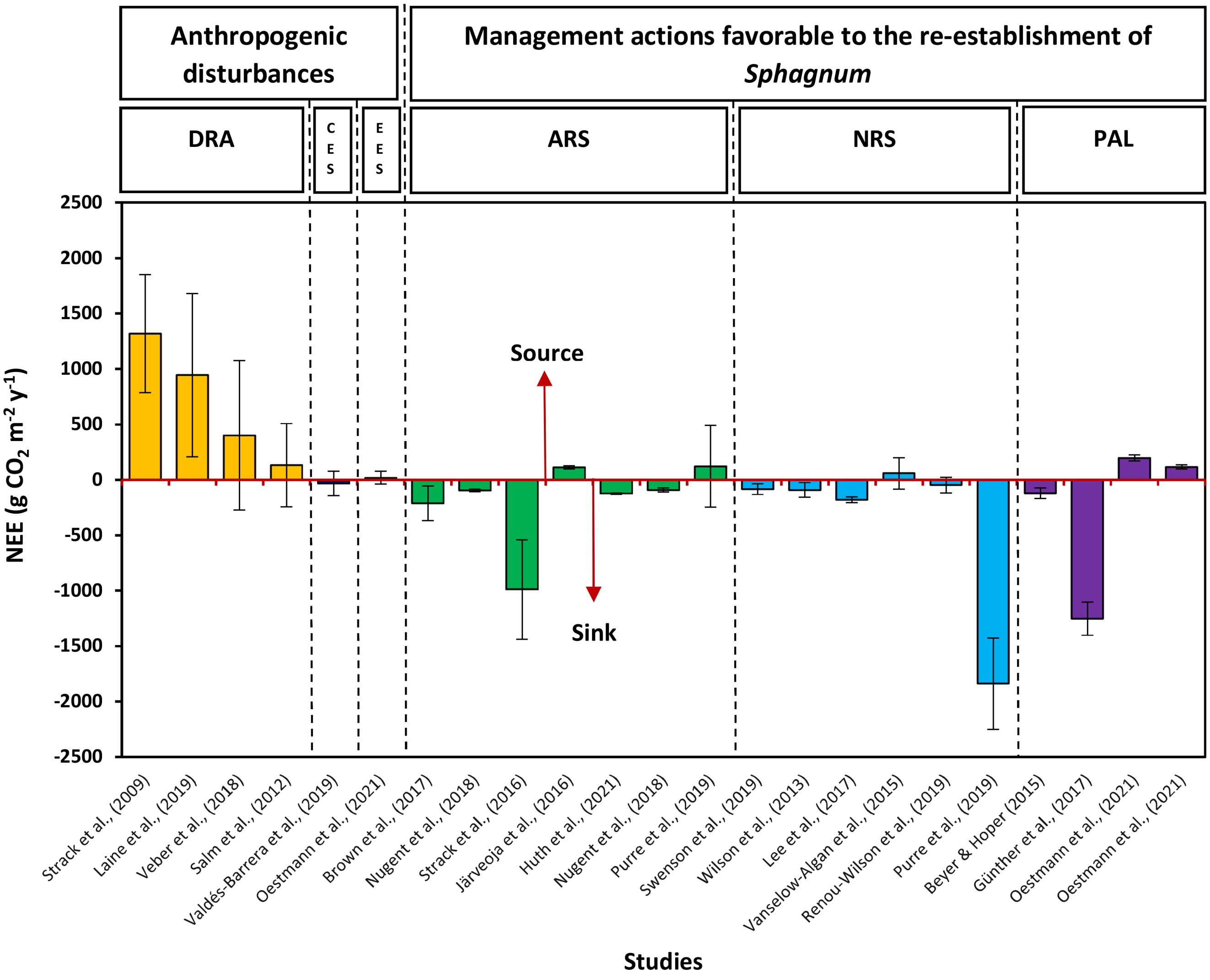

FIGURE 6 Annual mean values of NEE (g CO_2 m^{-2} $year^{-1}$) obtained from various studies in relation to different direct anthropogenic disturbances performed on *Sphagnum* (DRA, drainage; CES, commercial extraction of *Sphagnum*; EES, ecological extraction of *Sphagnum*), and management actions that favor moss reestablishment (ARS, assisted recolonization of *Sphagnum*; NRS, natural recolonization of *Sphagnum*; PAL, *Sphagnum* paludiculture).

reduces moss productivity, increases their autotrophic respiration, and increases R_{eco} rates, generating higher CO_2 emissions to the atmosphere (Figure 6; Table 1; Table S1) (Blodau et al., 2004; O'Neill et al., 2022; Riutta et al., 2007; Tuittila et al., 2004). We generated this consensus cautiously due to some contradictory results reported in the literature, indicating increases, decreases, or low differences in *Sphagnum* photosynthesis rates and R_{eco} associated with drainage (Huang et al., 2021; Kokkonen et al., 2022; Ma et al., 2022), a condition related to the complexity of the variables involved, and to the difficulty of identifying a unifying mechanism for NEE responses considering the multiple interactions between biotic and abiotic components in a peatland (Huang et al., 2021; Laiho, 2006; Laiho et al., 2004). For example, NEE quantifications by Riutta et al. (2007) in a poor fen with WT reductions of −14 and −22 cm under different vegetation covers, indicate that sites covered only by peat and *Sphagnum* mosses were a source of CO_2 to the atmosphere (203 mg CO_2 $m^{-2}h^{-1}$), while sites covered by *Sphagnum* and sedges, and those where *Sphagnum*, sedges, and shrubs were established were CO_2 sinks with NEE records of −114 and −130 mg CO_2 $m^{-2}h^{-1}$, respectively; results generated mainly from the decrease in gross photosynthesis of *Sphagnum* since it was the plant component most sensitive to WT reduction.

On the other hand, experimental studies have demonstrated that in the range of environmental conditions determined for different peatlands, CO_2 emissions in samples dominated by *Sphagnum* generally increase with greater depth of WT (Blodau et al., 2004; Blodau & Moore, 2003; McNeil & Waddington, 2003) (Table 1; Table S1). However, these types of studies account for short-term C dynamics, which may differ widely from long-term conditions, since in general, the disturbed system will always lose C after disturbance, but in the long term, they inherently vary among climates and peatland types (Laiho, 2006).

2.2 | Impacts of commercial extraction of *Sphagnum* mosses on CO_2 emissions in peatland ecosystems

Sphagnum biomass is commercially harvested from natural and seminatural peatlands in several countries around the world, mainly in Australia, New Zealand, Chile, Finland, and Germany (Díaz et al., 2008; Guêné-Nanchen et al., 2019; Silvan et al., 2017; Whinam & Buxton, 1997; Wichmann et al., 2017), due to its high economic value and global demand (Krebs et al., 2017; Whinam et al., 2003; Zegers et al., 2006). Harvesting methods vary from manual extraction and the use of forks to the use of tractors or mechanical diggers (Krebs et al., 2017; Kumar, 2017; Silvan et al., 2017), causing ecological disturbances and alterations in CO_2 emissions, especially in sites where indiscriminate extraction has taken place, which remain waterlogged, without moss regeneration, and in some cases without vegetation cover (Díaz et al., 2008; León et al., 2012). Several studies have reported slow recolonization of *Sphagnum* in harvested peatlands due to modifications of hydrological conditions, and reduced growth rate (Díaz & Silva, 2018; León et al., 2012; Valdés-Barrera et al., 2019; Whinam et al., 2003; Whinam & Buxton, 1997). However, according to Silvan et al. (2017), *Sphagnum* cover in a Finnish peatland redeveloped rapidly after harvest; a condition also affirmed by Guêné-Nanchen et al. (2019) for the case of *Sphagnum* propagule donor peatlands in Canada, where cover increased linearly from the harvest, reaching 70% in ca. 11 years after the harvest. Other studies have stated that 80% of the cover can be recovered after 1 year by extracting the top 5–10 cm in natural peatlands with high productivity (Krebs et al., 2017).

Studies addressing the impacts of this anthropogenic disturbance on CO_2 fluxes are scarce, so the global background is quite limited. Despite this, there is an initial consensus that removal of the moss stratum allows C stored in the peatland to be released to the atmosphere as CO_2 through oxidation of the exposed peat, also generating a reduction in the GPP of the site (Newman et al., 2018; Valdés-Barrera et al., 2019). Nevertheless, it is possible to recover significant portions of the sink function within a short period of time (3–4 years) after the harvest (Silvan et al., 2017; Figure 6; Table 1, Table S2).

Data related to CO_2 emissions at sites with commercial *Sphagnum* extraction have been obtained from anthropogenic peatlands in southern Chile, an activity that has been conducted since the 1990s (Díaz et al., 2008; León et al., 2012), in which case the sites subjected to harvesting show CO_2 emissions close to neutrality, that is, a small C sink (−33 ± 111 g CO_2 m^{-2} $year^{-1}$). However, adjacent areas kept intact (without moss removal) for about 40 years have higher negative NEE rates (−135 ± 267 g CO_2 m^{-2} $year^{-1}$), that is, undisturbed areas are on average a larger sink than areas with *Sphagnum* removal, due to a higher site GPP and a slightly higher R_{eco} rate, which would be associated with a more abundant woody vascular plant cover (Valdés-Barrera et al., 2019).

For the case of a peatland used as a demonstration area for *Sphagnum* biomass harvesting during 2013 and 2014 in Finland, the short-term effects for an undisturbed site evinced a net seasonal CO_2 balance of −39.0 and −19.2 g CO_2-C m^{-2} $season^{-1}$ in each year evaluated, that is, a CO_2 sink. In turn, the harvested area showed a small positive seasonal balance in 2013 of 1.7 g CO_2-C m^{-2} $season^{-1}$, and a negative balance in 2014 with −11.1 g CO_2-C m^{-2} $season^{-1}$ (Silvan et al., 2017).

2.3 | The removal of *Sphagnum* as a donor material for use in ecological restoration of transitional peatlands and its impacts on CO_2 emissions

One of the most widely used approaches for the restoration of degraded and abandoned peatlands corresponds to the Canadian moss layer transfer technique (MLTT), which considers the active reintroduction of *Sphagnum* fragments, and the blocking of drainage ditches among other activities (Graf & Rochefort, 2010; Quinty & Rochefort, 2003). For this purpose, *Sphagnum* fragments are

TABLE 1 Effects of anthropogenic disturbances on the average CO_2 fluxes ($g\,CO_2\,m^{-2}\,day^{-1}$) derived from *Sphagnum* mosses reported in the literature.

Country	Type of peatland	Type and details of anthropogenic disturbance	Method used to obtain CO_2 fluxes	Measurement period	Treatment/condition	CO_2 fluxes[a]	References
Canada	Ombrotrophic	Peatland drainage. Drainage for 7 years prior to the study	Closed dynamic chamber	Growing season (June–August) 2013 and 2014	Drained site dominated by *Sphagnum*—Shrubs WT 2013: −77.7 cm WT 2014: −62.9 cm	Year 1: −0.23 Year 2: 1.38	Harris et al. (2020)
					Drained site dominated by *Sphagnum*-Spruce WT 2013: −114.7 cm WT 2014: −114.2 cm	Year 1: −1.37 Year 2: 2.14	
France	Transitional mesotrophic minerotrophic	Peatland drainage Drainage due to road construction and drainage ditch wear in 2009 increasing water loss	Closed static chamber	March 2013–February 2015	Drained site year 2013 WT: −18 cm in summer	10.83 ± 11.59	D'Angelo et al. (2021)
					Drained site year 2014 WT: −10.5 cm in summer	11.14 ± 10.53	
Estonia	Ombrotrophic	Peatland drainage. Drainage for forestry 40 years prior to the study	Closed static chamber	January to December 2009	Drained sites Site 1, WT: −13 cm Site 2, WT: −13 cm Site 3, WT: −15 cm	Site 1: 0.53 Site 2: 0.70 Site 3: 0.91	Salm et al. (2012)
Canada	Ombrotrophic	Peatland drainage	Closed static chamber	Growing season (May–August) 2005	Transect perpendicular to drainage ditch. Measurements at 1.5 m from the ditch. WT at 1.5 m from the ditch: −130 cm	3.6	Strack et al. (2009)
Canada	Ombrotrophic	Peatland drainage performed for peat extraction	Closed dynamic chamber	Growing season period 1999–2002	Average of 3 years of evaluation. Average WT: −66 cm	4.47	Strack et al. (2016)
Canada	Ombrotrophic	Peatland drainage performed for peat extraction	Eddy-covariance	July 2013–November 2016	Evaluations conducted 1 year and 15 years after the abandonment of the peatland	Year 1: 4.47 Year 15: 2.17	Nugent et al. (2018) and Nugent et al. (2019)
Canada	Cores obtained from an ombrotrophic peatland	Initial WT from −2 to −6 cm. After 60 days, WT was reduced to −36 cm. Between days 143 and 223, WT was between 0 and −2 cm and −30 and −33 cm, respectively	Closed static chamber in peat cores (mesocosm)	Not specified	Site 1: WT: −2 to −6 cm	2.07 ± 0.35	Blodau et al. (2004)
					Site 1: WT: −36 cm	3.39 ± 0.66	
					Site 2: WT: −2 to −6 cm	3.74 ± 0.66	
					Site 2: WT: −36 cm	4.18 ± 0.45	
Finland	Minerotrophic	Harvesting/Extraction of *Sphagnum* biomass	Closed dynamic chamber	Growing season (May–October) 2013 and 2014	Sites with commercial *Sphagnum* harvesting at a maximum depth of 30 cm. Average value for 2 years (2013 and 2014)	−5.81	Silvan et al. (2017)
Chile	Transitional mesotrophic minerotrophic	Harvesting/commercial extraction of *Sphagnum* biomass	Eddy-covariance	April 2015–October 2016	Site with commercial *Sphagnum* extraction	−0.09 ± 0.3	Valdés-Barrera et al. (2019)

TABLE 1 (Continued)

Country	Type of peatland	Type and details of anthropogenic disturbance	Method used to obtain CO_2 fluxes	Measurement period	Treatment/condition	CO_2 fluxes[a]	References
Germany	Ombrotrophic	Extraction of *Sphagnum* as donor plant material for use in peatland restoration	Closed static chamber	March 2017 to March 2018 (year 1) March 2018 to March 2019 (year 2)	Removal site of the top 5 cm of *Sphagnum*	Year 1: −0.40±0.20 Year 2: 0.20±0.60	Oestmann et al. (2021)

[a] Positive fluxes indicate CO_2 emissions to the atmosphere, and negative fluxes indicate CO_2 sequestration by the ecosystem.

harvested from a donor peatland, which are then spread on the site subject to restoration (Guêné-Nanchen et al., 2019; Murray et al., 2017). However, harvesting generates alterations in the donor peatland, either in site recovery or in CO_2 emissions due to decreased GPP and increased exposure of the readily oxidizable OM (McNeil & Waddington, 2003; Tuittila et al., 2004; Waddington et al., 2010). Despite a little documented history of these impacts (Murray et al., 2017; Oestmann et al., 2021), it has been suggested that disturbed sites recover relatively quickly, with moss covering ca. 70% after 11 and 30 years for peatlands located in Canada and Finland, respectively (Guêné-Nanchen et al., 2019; Silvan et al., 2017), returning to its original C balance naturally.

Studies that have addressed the impacts of this disturbance on CO_2 emissions at two donor sites, corresponding to a rich fen and a bog in Canada, report that both areas released more CO_2 during the growing season than adjacent undisturbed natural sites (Murray et al., 2017) (Figure 6; Table S3). In the case of the bog, sites with moss removal conducted 1 year prior to the study were a source of CO_2, with low GPP rates during the 2 years of records (2013 and 2014), and low R_{eco} rates during the second year, due to incipient vegetation recovery. Similar records showed sites with moss removal performed 6 years earlier, evidencing higher GPP rates in both seasons and higher R_{eco} records in 2014, due to higher moss recovery. Therefore, CO_2 emissions were not affected by the time elapsed since moss harvesting. In turn, both sites exhibited NEE values 203% higher than an adjacent undisturbed natural bog, a condition that could be related to the time required for adequate *Sphagnum* recolonization and the consequent increase in GPP. For its part, the donor fen was a 230% higher emitter of CO_2 than the bog sites, and the undisturbed natural fen was shown to be an important sink (Murray et al., 2017).

Higher or lower CO_2 release from *Sphagnum* donor sites will be associated with post-intervention WTs, which generally increase because moss removal causes a decrease in water-holding capacity (Murray et al., 2017; Oestmann et al., 2021; Silvan et al., 2017). Residual vascular plant cover and their productivity rates should also be considered, as the establishment of sedges correlates with high initial rates of C sequestration (Kuiper et al., 2014; Strack et al., 2014; Tuittila et al., 1999), due to a higher biomass volume and wider leaf area index in this plant type.

3 | THE IMPACT OF MANAGEMENT ACTIONS FAVORING THE REESTABLISHMENT OF *SPHAGNUM* MOSSES ON CO_2 EMISSIONS IN PEATLAND ECOSYSTEMS

In the previous section, the impact of *Sphagnum* extraction on CO_2 emissions in moss-donor peatlands was discussed. However, it is important to determine whether the assisted reintroduction and natural recolonization of *Sphagnum* in degraded peatlands can restore the carbon sink function to levels comparable with undisturbed natural peatlands.

Hydrological restoration of peatlands by blocking drainage ditches, together with the reintroduction of *Sphagnum* mosses (Beyer & Höper, 2015; Lamers et al., 2015; Quinty & Rochefort, 2003; Rochefort et al., 2003) have been promoted in Europe and North America as a method of land use management, being considered a climate mitigation tool and a means to decrease greenhouse gas (GHG) emissions (Bonn et al., 2016; IPCC, 2014; Järveoja et al., 2016; Mahmood & Strack, 2011).

3.1 | Assisted reintroduction of *Sphagnum* mosses in degraded peatlands and its impact on CO_2 fluxes

In general terms, the active reintroduction of *Sphagnum*, together with other rehabilitation activities, accelerates the reestablishment of the typical vegetation and restores the peatland to its CO_2 sequestration condition (Figure 6; Table 2; Table S4) (Brown et al., 2017; Huth et al., 2021; Murray et al., 2017; Tuittila et al., 2004). It has been shown that disturbed sites without a vegetative cover of mosses (bare peat), show seasonal CO_2 emissions 150% higher than sites restored with *Sphagnum* (McNeil & Waddington, 2003). In turn, Purre et al. (2019) state that the *Sphagnum* cover explained between 12% and 82% of CO_2 uptake and sequestration in a peatland restored by MLTT in Estonia, which has also been reported for peatlands restored with moss in northern Europe and North America (Brown et al., 2017; Järveoja et al., 2016; Laine et al., 2016).

It has been shown that restoration approaches based on removal of the residual peat surface layer at 30 and 60 cm depth, rewetting, and incorporation of *Sphagnum* fostered CO_2 sequestration in a transitional peatland in northwestern Germany (Huth et al., 2021), a condition that was also associated with the maintenance of a high and stable WT between 0 and −10 cm. Sites with peat removal at 30 cm and reintroduction of *Sphagnum* sequestered three times more C (−80.5 ± 11.4 g CO_2-C m^{-2} year^{-1}) than areas where only the WT was rehabilitated (−26.9 ± 11.9 g CO_2-C m^{-2} year^{-1}). In turn, sites with peat removal at 60 cm and moss introduction were the strongest C sink (−123.6 ± 10.2 g CO_2-C m^{-2} year^{-1}), in contrast to areas with hydrological rehabilitation, which were a moderate source (30.6 ± 8.2 g CO_2-C m^{-2} year^{-1}). Similar conditions were reported for an abandoned poor fen in Canada, restored using MLTT, where sites with restored hydrology and *Sphagnum* reintroduction went from being a net emitter (132.9 ± 17.8 g CO_2-C m^{-2} season^{-1}) the first year of records (2010), to a net sink (−64.7 ± 33.2 g CO_2-C m^{-2} season $^{-1}$) the third year (2012); unlike areas with WT rehabilitation, where CO_2 emission decreased (from 184.6 ± 9.9 to 78.1 ± 8.0 g CO_2-C m^{-2} season^{-1}), but a net source condition was maintained after 3 years (Lazcano et al., 2018).

Contrary to the aforementioned, the incorporation of *Sphagnum* fragments in a transitional bog in Estonia maintained its CO_2 source status, although 60% lower than that of a surrounding site with bare peat (Järveoja et al., 2016). Similarly, Purre et al. (2019) indicate, for two bogs restored by MLTT in Estonia, that both sites were a source of CO_2 (site 1: 29 ± 77 g CO_2 m^{-2} season^{-1}; site 2: 72 ± 42 g CO_2 m^{-2} season^{-1}) due to a WT that declined to −45 cm induced by the drought, however, one of them became a sink (site 1: 13 ± 77 g CO_2 m^{-2} season^{-1}; site 2: −48 ± 42 g CO_2 m^{-2} season^{-1}) in the following season, product of a higher WT of −30 cm. Despite these results, it is possible to indicate that maintaining a near-surface WT is not a sufficient condition to ensure an efficient CO_2 sink, since, in general, the incorporation of moss brings fluxes closer to adjacent undisturbed sites than to those where only the WT was rehabilitated (Lazcano et al., 2018) within the first years after its reintroduction.

Several studies that have addressed the effects of restoration on the dynamics of CO_2 fluxes over the medium term (15 years) at the same site have generated relevant insights into biogeochemical functioning in these environments (Nugent et al., 2018, 2019; Strack & Zuback, 2013; Waddington et al., 2010). An example of this is represented by the Bois-des-Bel peatland (Quebec, Canada), restored by MLTT in 1999 (Strack & Zuback, 2013; Waddington et al., 2010), with records of CO_2 emissions in years after the restoration that has allowed to establish a chronology of its condition as a sink/emitter. From this, Waddington et al. (2010) report that immediately after restoration, the NEE was similar between unrestored sites and those restored with *Sphagnum*. Three years after restoration, the restored area was a consistent sink for CO_2, showing a 220% increase in seasonal NEE (Waddington et al., 2010). However, after that, a CO_2 sequestration gradient differentiated by vegetation cover type is established, where areas with herbaceous plants > *Sphagnum* > shrubs. For their part, Strack and Zuback (2013) report that after 10 years, three sites studied, corresponding to the restored area, a degraded area without restoration, and a natural area, were a net source of CO_2, although CO_2 emissions from the restored site were 58% and 82% lower than in the natural and non-restored site, respectively. Finally, Nugent et al. (2019) indicate that the peatland returned to being a CO_2 sink 14 years after restoration, with a significantly higher NEE at the site with *Sphagnum* reintroduction compared to the natural reference area (Table S4).

Favorable results are reported for a restored bog with diverse *Sphagnum* species on the east coast of Canada, where mean CO_2 uptake doubled from the first to the second year (2013: −2.85 ± 0.26 g CO_2 m^{-2} day^{-1}; 2014: −5.60 ± 0.42 g CO_2 m^{-2} day^{-1}), due to increased moss cover, ranging from 12% to 65% during the first year to 12.4%–82.5% during the second year (Brown et al., 2017). However, the use of chronosequences of greenhouse gas emissions performed by Nugent et al. (2019) in a peatland with MLTT restoration in Canada, indicates that after 1 year the CO_2 emissions of the restored site were similar to adjacent non-restored areas, due to the decomposition of the straw mulch used in this technique to maintain moisture, a condition that is also indicated by Brown et al. (2017) when stating that mulch decomposition accounts for half of the seasonal respiration in a peatland restored with MLTT in Canada. Considering this situation, it has been proposed that after 4 years following restoration, reductions in annual CO_2 emissions associated with a decrease in mulch decomposition (Brown et al., 2017; Nugent et al., 2018) and increased moss GPP could be observed, obtaining a net CO_2 sink condition after 14 years (Nugent et al., 2019; Table S4).

TABLE 2 Effects of favorable management actions for the reestablishment of *Sphagnum* mosses in peatlands on the average CO_2 fluxes ($g\,CO_2\,m^{-2}\,day^{-1}$) reported in the literature.

Country	Peatland type	Type and details of the favorable anthropogenic action	Method used to obtain CO_2 fluxes	Measurement period	Treatment/condition	CO_2 fluxes[a]	References
Canada	Ombrotrophic	Bog with peat extraction and subsequent restoration by MLTT technique	Closed dynamic chamber	July to August 2014 May to August 2015	Wet site, stable WT / WT < 15 cm (average of two periods)	−0.58 ± 0.43	Brown et al. (2017)
					Wet site, unstable WT / WT < 15 cm (average of two periods)	1.78 ± 0.23	
					Dry site, stable WT / WT between 15 and −25 cm (average of two periods)	−1.07 ± 0.22	
					Dry site, unstable WT / WT between 15 and 25 cm (average of two periods)	0.66 ± 0.18	
Canada	Ombrotrophic	Bog with peat extraction and subsequent restoration by MLTT technique.	Closed dynamic chamber	Growing season (May–October 2010) Non-growing season (January, February, March) 2010	Restored site with 88% *Sphagnum* cover, 20% vascular plants WT = −26.5 cm	4.5 ± 0.31	Strack and Zuback (2013)
Finland	Minerotrophic	Fen with peat extraction in blocks with subsequent abandonment (transition peatland). Restoration by MLTT technique.	Closed static chamber	June to November 1995 June to October 1996 June to September 1997 May to October 1998	Site with WT > −12 cm physiological optimum (dry site) (average of measurement periods)	1.95	Tuittila et al. (2004)
					Site with WT = −12 physiological optimum (optimum site) (average of measurement periods)	−0.80	
					Site with WT < −12 cm physiological optimum (wet site) (average of measurement periods)	−0.32	
Canada	Ombrotrophic and minerotrophic	Three peatlands on the east coast and three on the west coast disturbed by peat extraction and mineral soil removal. Restoration by MLTT technique	Closed dynamic chamber	Growing season period 1999–2002	Peatlands restored by MLTT technique	−2.9	Strack et al. (2016)
Estonia	Ombrotrophic	Peat extraction for horticultural purposes. Restoration by modified MLTT technique	Closed dynamic chamber	May to December 2014	Site with WT = −24 cm (wet). *Sphagnum* cover 31 to 91% Vascular plant cover: 2 to 9%. Shrub cover: 0 to 7%	1.11	Järveoja et al. (2016)
					Site with WT = −31 cm (dry) *Sphagnum* cover: 12 to 70% Vascular plant cover: 5–22%. Shrub cover: 5–22%	1.03	
Canada	Ombrotrophic	Transition peatlands restored by MLTT technique	Eddy-covariance	November 2013 to October 2016	Disturbed site with MLTT restoration. (90% cover of *Sphagnum* mosses and 33% cover of vascular plants)	Year 1: −0.95 ± 0.1 Year 2: −1.06 ± 0.07 Year 3: −0.70 ± 0.07	Nugent et al. (2018)
Canada	Ombrotrophic	Transition peatlands restored by MLTT technique	Closed dynamic chamber	Growing season (July–September) 2011 Growing season (May to July) 2012	Restored site	−23.4 to 2.2	Strack et al. (2014)
					Neighboring unrestored site	3.8 to 11.6	
Canada	Ombrotrophic	Transition peatlands restored by MLTT technique with long-term evaluation	Eddy-covariance	July 2013 to November 2016	Site restored 1 year after abandonment	5.06	Nugent et al. (2019)
					Site restored 4 years after abandonment	1.46	
					Site restored 15 years after abandonment	−0.90	
					Site restored 30 years after abandonment	−0.73	

TABLE 2 (Continued)

Country	Peatland type	Type and details of the favorable anthropogenic action	Method used to obtain CO_2 fluxes	Measurement period	Treatment/condition	CO_2 fluxes[a]	References
Canada	Ombrotrophic	Transition peatland with natural recolonization of *Sphagnum*	Closed dynamic chamber	Growing season (May–August) 2000	Dry site 1: WT = −31.5 cm	3.1 ± 1.6	McNeil and Waddington (2003)
					Dry site 2: WT = −31.3 cm	2.0 ± 1.4	
					Wet site 1: WT = −17.7 cm	−1.3 ± 1.1	
					Wet site 2: WT = −21 cm	2.6 ± 2.6	
Ireland	Ombrotrophic	Transition peatland with hydrological restoration and natural recolonization of peatland species	Closed static chamber	November 2008 to December 2011	Site dominated by *J. effusus* – *S. cuspidatum* WT = −5.6 cm	Year 1: −1.43 Year 2: −0.43 Year 3: −2.12	Wilson et al. (2013)
					Site dominated by *S. cuspidatum* WT = −12.5 cm	Year 1: −1.07 Year 2: −0.14 Year 3: −1.49	
Canada	Ombrotrophic	Transition peatland with hydrological restoration and natural recolonization of peatland species	Eddy-covariance	June 2015–June 2016	Natural recolonization of *Sphagnum* and sedges subsequent to rewetting	−1.79 ± 0.26	Lee et al. (2017)
Ireland	Ombrotrophic	Transition peatland with hydrological restoration and natural recolonization of peatland species	Closed dynamic chamber	November 2008–December 2013	Site 1 dominated by *J. effusus* and *S. cuspidatum*. Average WT = − 6.06 cm. (average annual value of CO_2 fluxes in 5 years)	−0.75	Wilson et al. (2016)
					Site 2 dominated by *S. cuspidatum* and *E. angustifolium* Average WT = −12.7 cm. (average annual value of CO_2 fluxes in 5 years)	−0.84	
Germany	Ombrotrophic	*Sphagnum* paludiculture	Closed dynamic chamber	September 2011 to August 2013	Sowing of fragments of *S. palustre*	Year 1: −1.72 ± 0.52 Year 2: −1.50 ± 0.25	Günther et al. (2017)
					Sowing of fragments of *S. papillosum*	Year 1: −2.46 ± 0.54 Year 2: −2.40 ± 0.27	
Germany	Ombrotrophic	*Sphagnum* paludiculture	Closed static chamber	March 2017–March 2018 (year 1) March 2018–March 2019 (year 2)	Cultivation of *S. papillosum* and *S. palustre* by MLTT technique, irrigation ditches	Year 1: 1.71 ± 0.13 Year 2: 2.21 ± 0.11	Oestmann et al. (2021)
					Cultivation of *S. papillosum* and *S. palustre* by MLTT technique, drip irrigation	Year 1: 0.91 ± 0.10 Year 2: 0.70 ± 0.10	
Germany	Ombrotrophic/polder	*Sphagnum* paludiculture	Closed static chamber	March 2017 to March 2018 (year 1) March 2018–March 2019 (year 2)	Cultivation of *S. papillosum* by MLTT technique and irrigation ditches through surrounding polders	Year 1: −0.60 ± 0.2 Year 2: 0.10 ± 0.10	Oestmann et al. (2021)
					Cultivation of *S. palustre* by MLTT technique and irrigation ditches through surrounding polders	Year 1: −0.00 ± 0.10 Year 2: 0.91 ± 0.20	
					Cultivation with a mixture of *Sphagnum* species of hummocks using MLTT technique and irrigation ditches through surrounding polders	Year 1: 0.10 ± 0.2 Year 2: 2.21 ± 0.2	

[a] Positive fluxes indicate CO_2 emissions to the atmosphere, and negative fluxes indicate CO_2 sequestration by the ecosystem.

Although there is a limited database that allows to consistently compare the dynamics of CO_2 emissions in sites with moss removal for restoration purposes with those where assisted reintroduction of *Sphagnum* has been performed, it is possible to preliminarily indicate that the incorporation of moss fragments together with an adequate management of WT can recover the CO_2 sink function of disturbed sites in the short and medium term (Figure 6). However, long-term studies and others are required to increase the reliability of the basis of comparison, for example, by contrasting peatlands of the same type in similar geographic locations and with a history of equivalent management or previous use.

3.2 | Natural recolonization of *Sphagnum* mosses in transitional peatlands and its impact on CO_2 emissions

Studies that have addressed the spontaneous reestablishment of *Sphagnum* in natural (Campbell & Corson, 2014), drained (Price et al., 2016), and rewetted (Tomassen et al., 2004) peatlands indicate that the moss presents serious difficulties in generating successful recolonization, taking even close to 30 years to reach 10% cover in a hydrologically restored bog in Canada (Price & Whitehead, 2001). Despite this, natural reintroduction of *Sphagnum* generates a net CO_2 sink condition (Lee et al., 2017; Renou-Wilson et al., 2019; Swenson et al., 2019; Wilson et al., 2016) (Figure 6; Table 2; Table S5), with possible modifications due to WT decline (Swenson et al., 2019; Wilson et al., 2016). *Sphagnum* reestablishment, even subsequent to that of vascular plants, maintains a high WT 90% of the time due to its water-holding capacity, favoring CO_2 sequestration (Bengtsson et al., 2020; Taylor & Price, 2015; Waddington & Price, 2000; Wilson et al., 2016).

Hydrological restoration of a peatland can promote recolonization of both *Sphagnum* and vascular plants, in which case Wilson et al. (2013) indicate that, although moss-dominated areas generate a CO_2 sink (−5 to −140 g C m^{-2} $year^{-1}$), sequestration rates are lower than adjacent sites covered by a mixed *Juncus effuses* L./*Sphagnum* community (−35 to −204 g C m^{-2} $year^{-1}$), and than areas recolonized by *Eriophorum angustifolium* Honck. (−146 to −583 g C m^{-2} $year^{-1}$). A similar condition is reported by Purre et al. (2019) for a bog with hydrological rehabilitation in Estonia, where areas with a high WT (−9 cm) dominated by sedges were a larger sink for CO_2 than sites recolonized by *Sphagnum*. Despite this, vascular plant cover facilitates *Sphagnum* establishment (Kuiper et al., 2014; McNeil & Waddington, 2003; Pouliot et al., 2011), due to a protective effect against elevated solar radiation and desiccation (Heijmans et al., 2004; Kuiper et al., 2014; Pouliot et al., 2011).

Several studies have reported favorable results for CO_2 sequestration associated with natural *Sphagnum* reestablishment. For example, Lee et al. (2017) indicate a CO_2 sink condition (−179 ± 26.2 g CO_2-C m^{-2} $year^{-1}$) for a bog located on the west coast of Canada with 25% of the rewetted area recolonized by *Sphagnum*, with increases in NEE early in the growing season due to increases in GPP of moss and the vascular species *Rhynchospora alba* (L.) Vahl. Similar results are reported for a bog in Ireland rewetted in 2012 and where the reestablishment of *Sphagnum* and ericoid plants drove a moderate CO_2 sink during the 2 years of study (mean NEE: −49 ± 68 g C m^{-2} $year^{-1}$), in contrast to a surrounding site with the absence of *Sphagnum* that recorded significant CO_2 emissions (mean NEE: 137 ± 24 g C m^{-2} $year^{-1}$) (Renou-Wilson et al., 2019). Meanwhile, Swenson et al. (2019) report that sites recolonized by *Sphagnum* in a bog located in Finland were a CO_2 sink in both years of study (2016: −47 ± 43 g C m^{-2} $year^{-1}$; 2017: −53 ± 42 g C m^{-2} $year^{-1}$), unlike areas with *Calluna vulgaris* (L.) Hull reestablishment, which were a strong emitter in both years (2016: 219 ± 50 g C m^{-2} $year^{-1}$; 2017: 156 ± 61 g C m^{-2} $year^{-1}$), and another site dominated by *Eriophorum angustifolium* which showed a neutral condition in the first year, and a sink condition, although less than *Sphagnum*, during the second year (2016: 3 ± 61 g C m^{-2} $year^{-1}$; 2017: −22 ± 76 g C m^{-2} $year^{-1}$). Similarly, Kivimäki et al. (2008) report for a recolonized transitional peatland in southern Finland that sites covered by *Eriophorum vaginatum* L. and *Carex rostrata* Stokes created sinks of −23 to −114 g CO_2-C m^{-2} $season^{-1}$, while mixtures of sedges and *Sphagnum* mosses were larger sinks of −75 to −186 g CO_2-C m^{-2} $season^{-1}$.

A condition opposite to those already noted is reported by Vanselow-Algan et al. (2015) for a bog located in northern Germany, where three sites were differentiated based on the species that dominated natural recolonization 30 years after rewetting, in which case all areas were CO_2 sources, however, the *Sphagnum*-dominated site (99% cover) showed emissions four and five times lower (59.5 ± 142.5 g CO_2 m^{-2} $year^{-1}$) than areas dominated by *Molinia caerulea* (L.) Moench (67% cover; 247.3 ± 330.2 g CO_2 m^{-2} $year^{-1}$) and *Erica tetralix* L. (31% cover; 308 ± 386 g CO_2 m^{-2} $year^{-1}$), respectively, due to a deeper WT in sites with vascular vegetation and differentiated photosynthesis levels during the winter period between perennial (*Sphagnum* and *Erica tetralix*) and deciduous (*Molinia caerulea*) plants. Similar results were obtained by Samaritani et al. (2010) for a bog located in Switzerland, with NEE assessments 29, 42, and 51 years after *Sphagnum* recolonization, where the youngest site was a net source of CO_2 during the growing season (40 g CO_2-C m^{-2} $season^{-1}$), while the remaining areas were a net CO_2 sink (site 42 years: −222 g CO_2 m^{-2} $season^{-1}$; site 51 years: −209 g CO_2 m^{-2} $season^{-1}$), a condition associated with higher R_{eco} rates at the younger site during summer, indicating that the early stages of moss regeneration may be more sensitive to warmer weather.

4 | *SPHAGNUM* PALUDICULTURE AND ITS EFFECTS ON CO_2 EMISSIONS

Sphagnum cultivation corresponds to a land use strategy whose objective is to harvest live moss biomass in a renewable way (Gaudig et al., 2017). It can be oriented toward commercial purposes, ecological objectives, or a combination of both; and it has been proposed as a sustainable use alternative for transitional peatlands (Brown et al., 2017; Brust et al., 2017; Munster et al., 2015; Oestmann et al., 2019; Pouliot et al., 2014).

Sphagnum cultivation has been associated with the concept of peatland paludiculture (Gaudig et al., 2013, 2017; Oestmann et al., 2021; Wichmann et al., 2017), where various species of *Sphagnum* mosses are grown under controlled WT and harvested in 5-year cycles (Beyer & Höper, 2015) to obtain a slightly humified peat ("white" peat) (Gaudig et al., 2013). Cultivation experiences have been generated mainly in Germany and Canada, with some small-scale trials in Chile, Ireland, Finland, South Korea, New Zealand, and Japan in recent years (Pouliot et al., 2014). Most moss cultivations have been established following the MLTT technique, managing to generate an average moss cover ca. 90% after 4 years (Gaudig et al., 2017) and greater than 90% after 7 years (Daun et al., 2023; Pouliot et al., 2014), however, this depends on key environmental factors and the productivity of the moss species to be established.

Experiences with *Sphagnum* cultivation have allowed conducting field studies aimed at quantifying the CO_2 emissions generated at these sites, and, in some cases, it has been reported that the development of this practice offers considerable mitigation of emissions compared to the establishment of crops and grasslands on transitional peatlands (Günther et al., 2017; Oestmann et al., 2021; Pouliot et al., 2014), because the moss production strips show a CO_2 sink condition during the *Sphagnum* establishment phase, which can be reduced by 20% when considering a productive cycle of 7 years (Daun et al., 2023) (Figure 6; Table 2; Table S6).

The literature reports contradictory results related to CO_2 emissions, since, on the one hand, sites irrigated by ditches exhibit a CO_2 sink condition with a 2-year (2010/2011) average record of −98.7 g CO_2-C m^{-2} $year^{-1}$ (Beyer & Höper, 2015), and, in other cases, a net CO_2 source condition is indicated, with 2-year (2017/2018) average records of 195 g CO_2 m^{-2} $year^{-1}$ (Oestmann et al., 2021). It should be noted that both studies were developed in the same geographic region (northwestern Germany), and that the differences could be related to the influence of the WT on *Sphagnum* productivity and respiration rates (Brown et al., 2017; McNeil & Waddington, 2003; Newman et al., 2018). For example, the second year of records, during the Oestmann et al. (2021) study, was unusually dry and with high temperatures, generating fluctuations in WT and increases in R_{eco} rates. For their part, Günther et al. (2017) indicate that the production strips of two *Sphagnum* species (*Sphagnum palustre* L. and *Sphagnum papillosum* Lindb.) in northwestern Germany were small annual CO_2 sinks during the establishment phase with a NEE of -0.64 ± 0.04 g CO_2 m^{-2} h^{-1} for both species during the growing season (Table 2, Table S6). Despite these favorable results, Günther et al. (2017) state that to assess whether crop fields are net sinks or sources of CO_2 during a productive cycle, the harvested biomass of moss should be included in the balances, a situation addressed by Daun et al. (2023) in the same culture site 7 years after its establishment, indicating a homogenization in the composition of the cultivated *Sphagnum* species with a dominance of *S. fallax*, which is why the results CO_2 emissions were addressed together. From this, Daun et al. (2023), report that the production strips were a CO_2 sink throughout the production cycle (687 ± 405 g CO_2 m^{-2} a^{-1}), with a lower sequestration rate in the seventh year of cultivation compared to years 1 and 2 (Table S6), confirming that the NPP of the moss can remain high for several years. Despite this, at the end of the 7-year production cycle, and considering the partial harvest of *Sphagnum*, the site was a source of CO_2 (954 ± 454 g CO_2 m^{-2} a^{-1}), due to substantial contributions of CO_2 emissions generated by ditches and causeway established in the production area (ditches: 904 ± 790 g CO2 m-2 a-1; causeway: 2763 ± 984 g CO_2 m-2 a-1) (Daun et al., 2023).

An important variable to consider in order to foster CO_2 sequestration in cultivate sites is the maintenance of a high and stable WT, as fluctuations will affect NEE, especially when mosses are exposed to periodic desiccation (Brown et al., 2017; Oestmann et al., 2021; Price et al., 2003). However, *Sphagnum* presents water level limits due to the blockage of CO_2 diffusion in highly humid films surrounding the phyllids, potentially reducing gas exchange, photosynthetic rates, and CO_2 capture (Newman et al., 2018; Weston et al., 2015; Williams & Flanagan, 1998). The establishment and productivity of vascular plants in cultivation sites is another relevant factor to consider in CO_2 emissions, as they can alter GPP and autotrophic respiration levels (Ward et al., 2009). In general, increased vascular plant cover is accompanied by increases in GPP and R_{eco} rates at cultivate sites (Oestmann et al., 2021), which, in the case of R_{eco}, could contribute 40% to ecosystem CO_2 emissions (Peichl et al., 2018; Walker et al., 2016). In turn, the presence of a certain vascular vegetation cover, especially during the summer, can increase R_{eco} levels, and sometimes these rates can be higher than those of GPP, generating increases in CO_2 emissions (Oestmann et al., 2021). However, vascular vegetation can promote *Sphagnum* growth by providing microhabitats with higher shade and moisture levels (McNeil & Waddington, 2003; Oestmann et al., 2021; Pouliot et al., 2014).

5 | FINAL CONSIDERATIONS AND FUTURE RESEARCH NEEDS

Peatlands are complex ecosystems where a series of biogeochemical, ecophysiological, and hydrological processes interact under various mechanisms to generate CO_2 sequestration or emissions (Belyea & Baird, 2006; Blodau, 2002; Limpens et al., 2008; Vitt, 2006). When considering vegetation composition, *Sphagnum* mosses are recognized for their pivotal role in peat formation and, thus, in long-term C sequestration (Jassey & Signarbieux, 2019; Kasimir et al., 2021; Piatkowski et al., 2021), an essential process for nature-based climate change mitigation (Humpenöder et al., 2020). However, the lack of knowledge of their contributions to the ecosystem services provided by peatlands threatens their permanence through continuous anthropogenic disturbances, estimating that by 2050, 26% of these environments will show serious signs of degradation and will produce ca. 10% of global anthropogenic CO_2 emissions (Loisel & Gallego-Sala, 2022; Urák et al., 2017). The search for strategies and practical alternatives that promote the restoration, sustainable use, or conservation of these environments in the long term is urgent, as they are the most cost-effective measures to achieve the ultimate objective of zero net C emissions from the land use management sector (Ostle et al., 2009).

Although there is currently an important scientific database that addresses the influence of *Sphagnum* mosses on greenhouse gas emissions in peatland ecosystems, there is an evident geographic bias in the research developed, with records focused on boreal and subarctic environments in Europe and North America. Temperate peatlands in the southern hemisphere, and especially those located in the Patagonian region, are underrepresented in the scientific literature, despite increasing threats associated with commercial extraction of peat and *Sphagnum* moss (Holl et al., 2019; León et al., 2021; Loisel & Yu, 2013; Veber et al., 2018). On the other hand, field and experimental studies have focused on the effects of drainage, peat removal, and restoration of degraded peatlands. New practices related to *Sphagnum* paludiculture are receiving increasing attention due to their potential to use transitional peatlands sustainably, minimizing impacts related to CO_2 emissions (Gaudig et al., 2017). Nevertheless, we consider it relevant to increase experimental studies based on the elimination of plants subjected to various environmental alterations to determine specific mechanisms and the effects that each functional type of plant, including *Sphagnum*, exerts on CO_2 emissions. Finally, it is necessary to advance in the knowledge related to the impacts caused by the extraction of *Sphagnum*, either for commercial or ecological purposes, on CO_2 emissions, considering that this type of anthropic disturbance is one of the most widespread in peatlands distributed in the southern hemisphere.

6 | CONCLUSIONS

- *Sphagnum* mosses play a fundamental role in CO_2 uptake and sequestration in peatland ecosystems by producing a litter rich in recalcitrant compounds that favor peat formation. A continuous and permanent vegetation cover of *Sphagnum* consolidates atmospheric C storage and contributes to climate change mitigation.
- Anthropogenic disturbances that directly affect *Sphagnum* have the potential to modify to varying degrees its ability to reduce CO_2 emissions, with drainage being the disturbance that generates the most significant detrimental impacts.
- Anthropic actions that favor the recovery of *Sphagnum* cover, such as its natural or assisted reintroduction in degraded peatlands, return these sites to their status as CO_2 sinks in the medium term.
- *Sphagnum* cultivation in transitional peatlands offers an attractive possibility to mitigate CO_2 emissions, and to realize a sustainable use in transitional peatlands.
- It is necessary to expand research on the specific role of *Sphagnum* in CO_2 sequestration or emissions to less represented geographic areas (Patagonian peatlands), and to consider the impact of new anthropogenic pressures on its ecological role in CO_2 fluxes.

AUTHOR CONTRIBUTIONS

Patricio Andrés Pacheco-Cancino: Conceptualization; formal analysis; investigation; writing – original draft; writing – review and editing. **Rubén Fernando Carrillo-López:** Writing – review and editing. **Armando Sepulveda-Jauregui:** Writing – review and editing. **Marcelo Arturo Somos-Valenzuela:** Writing – review and editing.

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CONFLICT OF INTEREST STATEMENT

The authors have no conflict of interest to disclose.

DATA AVAILABILITY STATEMENT

The data that support the findings of this study are openly available in Zenodo at https://doi.org/10.5281/zenodo.8377704.

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SUPPORTING INFORMATION

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CAPÍTULO 12

PROPAGACIÓN ARTIFICIAL DEL MUSGO *SPHAGNUM MAGELLANICUM* BRID. (BRYOPHYTA: SPHAGNACEAE), COMO MODELO PARA EL RESCATE DE ECOTIPOS RELEVANTES EN LA CONSERVACIÓN DE ECOSISTEMAS DE TURBERAS

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RESUMEN

La creciente demanda comercial por el musgo *Sphagnum magellanicum* Brid. en Chile, ha generado una intensa actividad extractiva, impactando fuertemente los ecosistemas naturales de pomponales y turberas naturales producto de su sobreexplotación y consecuente degradación. Esta situación hace necesario generar acciones que permitan promover la conservación y restauración de estos ecosistemas. Considerando ello, el presente trabajo tuvo como objetivo colectar diversos ecotipos del musgo *S. magellanicum* y seleccionar aquel que presentó las características morfológicas más apropiadas y demandadas en el ámbito de su comercialización para ser masificado *ex situ*. Para ello se realizó una prospección de áreas con presencia de *S. magellanicum* en turberas antropogénicas entre las regiones de Los Ríos y de Los Lagos, desde donde se colectaron muestras de cada subpoblación. El ecotipo seleccionado correspondió al ubicado en el sector de Quillaipe, en la provincia de Llanquihue, el cual se masificó artificialmente en 6 contenedores de fibra de vidrio de 14 m^3, instalados en un invernadero de 275 m^2 de estructura metálica y cubierta de policarbonato, provisto de un sistema hidráulico que permitió la circulación de agua entre la turbera y el invernadero. En cada contenedor se depositó fierrillo, turba y una capa semidescompuesta de musgo sobre la cual se sembraron segmentos de 10 a 15 cm de *Sphagnum*. Los resultados arrojaron un crecimiento promedio en la longitud del caulidio de 0,46 y 0,78 cm por mes para la primera y segunda temporada anual respectivamente, superando las tasas naturales de crecimiento registradas en turberas antropogénicas. Se espera que el desarrollo de esta experiencia permita, a futuro, sentar las bases para el rescate y la conservación de ecotipos de *Sphagnum magellanicum*, favoreciendo con ello la

conservación y la restauración de ecosistemas de turberas perturbados por la actividad extractiva del musgo.

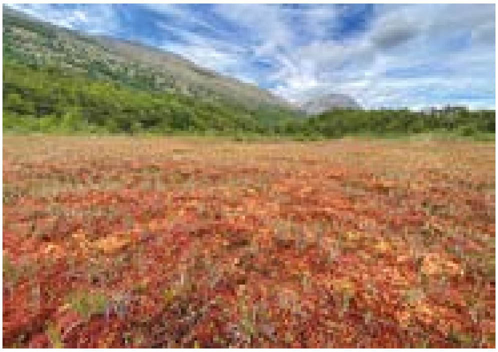
Palabras clave: producción artificial, ecotipos, *Sphagnum magellanicum*, conservación, restauración.

INTRODUCCIÓN

Las turberas corresponden a ecosistemas de humedales distribuidos en latitudes altas en ambos hemisferios, aunque también es posible encontrarlos en zonas tropicales (Díaz *et al.* 2015; Xu *et al.* 2018). Cubren cerca de 400 millones de ha, equivalente al 2,8% de la superficie terrestre mundial, y representan el 55% de los humedales del planeta (Secretaría de la Convención de Ramsar, 2004; Xu *et al.* 2018). Ellas también han sido reconocidas mundialmente por el suministro de bienes y servicios ecosistémicos que entregan a la sociedad, destacándose sus contribuciones en el abastecimiento, la regulación y la calidad del recurso hídrico, además de los servicios de regulación y soporte como la conservación de la biodiversidad y el secuestro de carbono, a lo que se suma la provisión de turba, musgos y fibras, entre otros recursos (Kimmel & Mander, 2010; De Groot *et al.* 2012; Grand-Clement *et al.* 2013; Martin-Ortega *et al.* 2014; Webster *et al.* 2018).

A pesar de sus importantes valores ecológicos y ambientales, las turberas han sido utilizadas en forma intensa principalmente en las zonas boreales, y desde hace algunas décadas atrás en zonas tropicales y patagónicas, causando su degradación debido al drenaje, la extracción indiscriminada de turba, los incendios, o el cambio de uso del suelo para la habilitación de tierras para la producción agrícola y forestal (Turestsky *et al.* 2015). Las turberas patagónicas presentes en Chile y Argentina han recibido menores perturbaciones en cuanto a intensidad y temporalidad, manteniéndose gran parte de su superficie en condiciones prístinas debido a las dificultades de acceso (Kleinebecker *et al.* 2010).

En Chile, los ecosistemas de turberas están circunscritos a la región de Tundra y, en sentido estricto, se extienden desde el golfo de Penas hasta el extremo sur del país (48°S - 56°S). Sin embargo, es posible observar incursiones septentrionales de dicha región hasta la zona de cordillera Pelada, al sur de la región de Los Ríos (Schlatter & Schlatter, 2004; León *et al.* 2012). Dentro de esta zona, y en particular en la región de Los Lagos, existen turberas originadas tras el retroceso de glaciares



(turberas naturales) y también otras áreas que corresponden a lugares anegados, y dominados por musgos del género *Sphagnum,* denominadas turberas antropogénicas, o localmente como pomponales, debido a que su origen se asocia a la quema o tala rasa de bosques nativos en sitios con drenaje pobre, por lo que son mucho más recientes que las turberas naturales del extremo sur del país (Zegers *et al.* 2006; Díaz *et al.* 2008; Díaz *et al.* 2012).

Durante las últimas décadas del siglo pasado las turberas de la región de Los Lagos comenzaron a ser intervenidas con el objetivo de extraer turba (Valenzuela & Schlatter, 2004; León *et al.* 2012). Junto a ello, a partir de mediados de la década de los 90, las turberas antropogénicas o pomponales comienzan a cobrar gran relevancia debido al interés por la cosecha y comercialización del musgo *Sphagnum magellanicum*, constituyéndose en una fuente de trabajo durante los meses de verano en las comunidades rurales. La creciente demanda por el musgo *Sphagnum* en la región de Los Lagos y de turba en la región de Magallanes ha traído como consecuencia una intensa actividad extractiva, lo que ha impactado fuertemente los ecosistemas naturales de los cuales forman parte, ello porque difícilmente vuelven a ser funcionales, debido a la lenta recuperación de las áreas, siendo finalmente abandonados (Díaz & Silva, 2018). El drástico deterioro de estos ecosistemas, junto a la fragmentación y degradación de importantes hábitats biológicos, han llevado a una enorme pérdida en la riqueza de especies y su diversidad genética. Las severas alteraciones de las condiciones ecohidrológicas y microclimáticas que han ocurrido en pomponales y turberas cosechadas intensamente, han dificultado la recolonización del musgo *Sphagnum*, disminuyendo con ello las superficies productivas de este recurso (León *et al.* 2014; Domínguez & Vega-Valdés, 2015; Díaz & Silva, 2018).

Esta problemática ha llevado a desarrollar una serie de iniciativas tendientes a disminuir el impacto sobre estos importantes ecosistemas, las cuales se han orientado al diseño y difusión de técnicas de cosecha sustentable para su implementación por parte de los recolectores (Domínguez, 2014; Oberpaur *et al.* 2018), siendo un importante aporte en la solución del problema; pero en la práctica, pocos recolectores aplican estas recomendaciones, principalmente por el nivel de degradación de algunos pomponales y la necesidad de alcanzar volúmenes determinados de producción.

Por otra parte, no se han identificado ni masificado aquellos ecotipos del musgo que, desde el punto de vista comercial, generen mejores oportunidades de mercado considerando las características de calidad demandadas por los exportadores, así como el rol ecológico que éstos cumplen y que deben ser resguardados para futuras acciones de restauración. En este sentido, algunos países europeos (Alemania y Finlandia) están desarrollando opciones alternativas de uso de la tierra para turberas, las que se basan en el cultivo de musgos *Sphagnum* asociados al concepto de paludicultura de turberas, permitiendo la producción de biomasa de musgos al tiempo que se evita el drenaje y la extracción de turba, proporcionando una alternativa de uso sostenible para estos ambientes (Wichmann *et al.* 2017; Günther *et al.* 2018). Junto a ello, la implementación de actividades de restauración de turberas dominadas por *Sphagnum*, conocida como técnica de transferencia de capa de musgo (Quinty & Rochefort, 2003), necesariamente requiere de sitios donantes donde el musgo *Sphagnum* es extraído desde turberas naturales, provocando alteraciones adicionales ante la imposibilidad de contar con una producción *ex situ,* como ocurre con la viverización de plantas vasculares (Murray *et al.* 2017). En la actualidad no existen experiencias descritas que den cuenta de una propagación artificial *ex situ* del musgo *Sphagnum magellanicum* desde un punto de vista productivo, junto a la selección de características comerciales de este recurso en base a la prospección, selección, recuperación y propagación artificial de ecotipos superiores. Por tanto, de manera inédita, se generó un sistema artificial *ex situ* de producción del musgo que permitió resguardar la mantención de la calidad del recurso, y promover un volumen para apoyar el resguardo ecológico de ecosistemas de pomponales y turberas de *Sphagnum*, los cuales por su extensión en nuestro país juegan un rol fundamental en los flujos de gases de efecto invernadero (CO_2, CH_4, N_2O) y en la regulación del régimen hídrico de los sectores donde se emplazan (Carrillo & Pacheco, 2017).

El presente trabajo tuvo como objetivo colectar ecotipos del musgo de turbera *Sphagnum magellanicum* Brid. y seleccionar aquel que presentó las mejores características morfológicas en cuanto a la calidad exigida para el producto a comercializar, para luego ser masificado *ex situ* considerando para ello las condiciones ambientales que favorecen su adecuado establecimiento y reproducción.

Esta investigación se desarrolló en el marco del Proyecto FIA (Fundación para la Innovación Agraria) "Plan Piloto de Masificación del Musgo *Sphagnum*" PYT-0087-2012 y ha sido registrada su propiedad intelectual con la inscripción N°277.195.

METODOLOGÍA

Colecta y selección de ecotipo

La colecta de ecotipos se realizó en base a los antecedentes bibliográficos existentes, la información aportada por servicios públicos (INDAP, ODEPA) y por comercializadores del musgo, logrando seleccionar un área geográfica comprendida entre las provincias de Valdivia (región de Los Ríos) y Chiloé (región de Los Lagos). Dentro de esta área geográfica se seleccionaron 5 sectores para la colecta del musgo, los cuales se detallan en el Tabla 1 y Fig. 1. En cada uno de estos lugares se extrajeron tres muestras compuestas de musgos, equidistantes 1 m a un punto central seleccionado como representativo del área para el cual se registró su coordenada UTM mediante GPS. El musgo se extrajo manualmente y se almacenó directamente en bolsas Ziploc rotuladas. Las muestras se almacenaron en un cooler con gel refrigerante para luego ser transportadas al Laboratorio de Biología Vegetal (LBV) de la Universidad de La Frontera, donde fueron refrigeradas a una temperatura de 2° C. En cada sector de muestreo se midieron las características químicas del agua y se caracterizó la vegetación circundante, aspectos ecológicos que condicionan junto al clima, el crecimiento y desarrollo del musgo, los cuales son de interés para la evaluación de su óptimo crecimiento. La identificación taxonómica se realizó mediante metodologías clásicas apoyadas con técnicas de biología molecular. Para ello, las muestras fueron analizadas en la sección de Botánica del Museo Nacional de Historia Natural, en donde se corroboró que todos los individuos colectados correspondían a la especie de briófito *Sphagnum magellanicum* Brid.

Tabla 1. Sectores de colecta de ecotipos del musgo de turbera *Sphagnum magellanicum.*

Sector	Región	Provincia	Coordenadas UTM S	Coordenadas UTM E
Los Ulmos	Los Ríos	Valdivia	5573413 S	659361 E
Quillaipe	Los Lagos	Llanquihue	5395124 S	690510 E
Aucar	Los Lagos	Chiloé	5329901 S	622477 E
El Palomar	Los Lagos	Chiloé	5341332 S	598626 E
Quellón	Los Lagos	Chiloé	5241874 S	608714 E

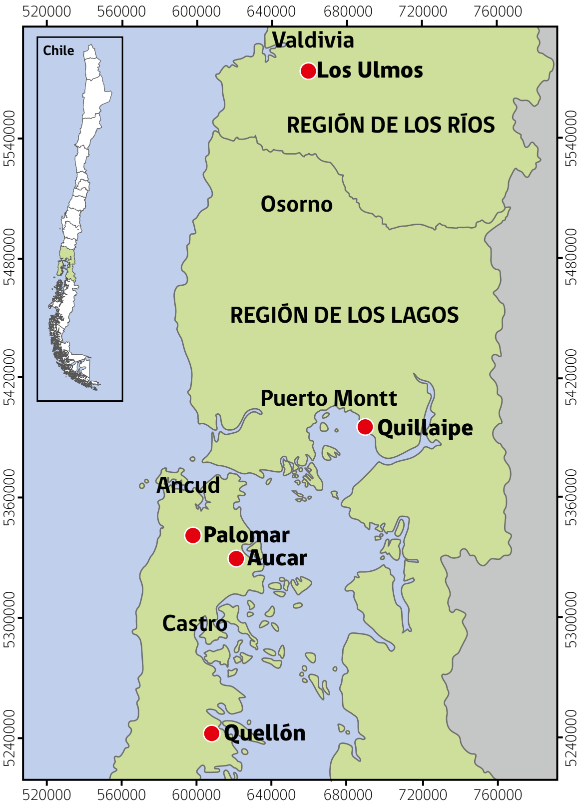


Fig. 1. Sectores de colecta de ecotipos del musgo *S. magellanicum.*

En el LBV se seleccionaron aquellos ecotipos superiores, es decir, que cumplían con características fenotípicas de interés comercial. Para ello también se consultó la opinión especializada de pequeños productores y de exportadores del musgo pompón. Los parámetros de calidad que son demandados por parte de compradores nacionales y exportadores para el musgo *Sphagnum* se focalizan principalmente en 2 aspectos, los que corresponden al grosor y la longitud de la fibra, además de que el producto no presente impurezas. En algunos casos también es posible que se soliciten características basadas especialmente en el color del musgo. Considerando lo indicado, la descripción morfológica fue realizada en base a parámetros cuantitativos y cualitativos para cada uno de los ecotipos muestreados. Dentro de los análisis cuantitativos se realizaron mediciones del largo de fascículos (conjunto de ramas que nacen de una inserción en el caulidio), además del largo y ancho de filidios. Se consideraron estos parámetros ya que al tener una mayor longitud de agrupaciones y mayores anchos de caulidio se asume un mayor poder de absorción del musgo. En cuanto a los parámetros cualitativos se evaluaron la forma, coloración y concavidad de filidios en agrupaciones y aquellos que salen directamente del caulidio.

Proceso de propagación artificial

El proceso de propagación artificial consideró el desarrollo de infraestructura, la cual fue instalada en un terreno particular perteneciente a uno de los socios de la Agrupación Gremial de Pequeños Agricultores de Musgo Pompón, ubicado en la localidad de Quillaipe, distante aproximadamente 28 km al sureste de la ciudad de Puerto Montt por la ruta 7 (Fig. 2).

Fig. 2. Localización infraestructura para la producción artificial del musgo *S. magellanicum.* Referencia respecto a la ciudad de Puerto Montt.

una superficie de 275 m^2 (25 m de largo por 11 m de ancho) construido en estructura metálica y recubierto completamente con policarbonato alveolar. Se confeccionaron 6 contenedores de fibra de vidrio de un volumen de 14 m3 cada uno y cuyas medidas fueron de 10 m de largo, 2 m de ancho y 0,7 m de altura, de color gris para impedir la proliferación de algas en el fondo y los bordes del contenedor. La disponibilidad permanente de agua en la producción artificial del musgo fue obtenida directamente desde el humedal (turbera) hacia los contenedores. Es así como se desarrolló un sistema hidráulico que permitió captar el agua de la turbera, transportarla hacia los contenedores y después tener la posibilidad de devolverla al humedal. Para ello se construyó un acumulador de agua en el humedal a 70 m en línea recta del invernadero. El sistema hidráulico consideró la instalación de una motobomba en el punto de salida del acumulador y cañerías de PVC para el transporte del agua hacia el invernadero (Fig. 3).

Fig. 3. Infraestructura generada para el proceso de masificación artificial. **A)** Invernadero recubierto en policarbonato, **B)** Contenedores de fibra de vidrio, **C)** Caseta motobomba sistema hidráulico y **D)** Acumulador de agua en la turbera.

En el invernadero se instalaron tuberías subterráneas de PVC y una llave de paso de agua para cada contenedor. Para la extracción del agua desde los contenedores hacia la turbera se instaló una motobomba al interior del invernadero con un montaje de filtros para reducir el transporte de sedimentos. El sistema de recarga y descarga de agua fue ejecutado a través de un panel de control dispuesto dentro del invernadero, el cual además disponía de todos los bloqueos de seguridad necesarios para no dañar las motobombas en caso de que no se produjera la succión y el movimiento del agua (Fig. 4). Se instalaron 6 ventanas abatibles dispuestas en las paredes, junto a dos extractores industriales ubicados en la cara frontal de la nave para realizar la circulación de aire y controlar la temperatura dentro del invernadero. Los extractores estaban sincronizados con 4 persianas instaladas en la cara posterior del invernadero para permitir la extracción del aire cálido acumulado al interior de la nave. Este sistema permitió reducir la temperatura ambiental de manera rápida y eficiente. Para el funcionamiento automático de los extractores se instaló un timer digital conectado a un sensor de temperatura ambiental ubicado en el interior del invernadero. Durante el período estival se consideró necesaria la instalación de una malla raschel de 80% de sombreado sobre el techo del invernadero, y sobre cada uno de los contenedores para disminuir la temperatura y la radiación solar excesiva, favoreciendo el crecimiento del musgo (Fig. 4).

Fig. 4. Sistema hidráulico dispuesto en el interior del invernadero. **A)** Motobomba de extracción de agua desde contenedores, **B)** Tablero de control, **C)** Llaves de paso dispuestas en cada contenedor, **D)** Incorporación de agua a contenedores, **E)** Extractores de aire industriales y **F)** Persianas sincronizadas a extractores para la circulación del aire.

El diseño de masificación artificial consideró un sistema que replicara las condiciones naturales de crecimiento del musgo en turberas antropogénicas presentes en el sector de Quillaipe, las cuales se caracterizan por presentar un reducido estrato de turba, el cual se acumula sobre un suelo mineral caracterizado por la presencia de una capa cementante de fierrillo (óxidos de fierro y aluminio), el cual provoca el anegamiento permanente o temporal de la turbera, y proporciona condiciones químicas particulares al agua presente en este ambiente.

Considerando ello, se recolectó una proporción de fierrillo, obtenida del mismo sector de emplazamiento de la infraestructura, el que fue esparcido homogéneamente en cada contenedor formando una capa de 10 cm de alto. Posteriormente se realizó un procedimiento similar con la turba, la cual fue obtenida desde cortes transversales en los bordes del humedal. La turba fue dispuesta de manera homogénea sobre la capa de fierrillo estableciendo un estrato de 10 cm de alto. Luego los contenedores fueron cargados parcialmente con agua para cubrir los estratos de fierrillo y turba. Sobre las dos capas anteriores se incorporaron homogéneamente 5 cm de musgo *Sphagnum* semidescompuesto sin capacidad de desarrollo, a modo de sustrato orgánico, el cual fue obtenido directamente desde la turbera. Finalmente se sembraron hebras de 10 a 15 cm de largo del ecotipo de musgo seleccionado para su masificación artificial (Campeau & Rochefort, 1996; Quinty & Rochefort, 2003), las cuales fueron esparcidas homogéneamente sobre el estrato de musgo parcialmente descompuesto. El material vegetal a propagar se obtuvo de la cosecha de ~ 70 kg de musgo húmedo desde la turbera aledaña, el cual se limpió y permaneció en dicho lugar hasta que perdió el 50% de su humedad. De esta manera se trató de afectar en menor medida el ciclo hidrológico del humedal (Domínguez, 2014). Luego de ello el musgo fue trasladado al invernadero para ser pesado y utilizado en la siembra de los contenedores. Se usaron ~ 25 kg de musgo para ser destinados en la siembra de cada contenedor, valor que se utilizó para establecer una relación de peso ganado al momento de la cosecha de los contenedores. Inmediatamente luego de la siembra, todos los contenedores fueron irrigados considerando como referencia un nivel freático muy próximo a la superficie (~ 2 cm de profundidad) en el cual el contacto de las hebras con el agua fue superficial, sin que éstas quedaran sumergidas como material sobrenadante.

La primera siembra fue realizada a inicios del mes de junio del año 2014, luego de lo cual los únicos factores que se mantuvieron en constante vigilancia fueron el nivel de agua en los contenedores (todo el período) y los niveles de radiación y temperatura (en el período estival). La primera cosecha fue realizada en el mes de mayo del año 2015. El segundo período de crecimiento considera la siembra realizada en el mes de junio del año 2015 y la cosecha en el mes de junio de 2016.

La cosecha del musgo en los contenedores se realizó mediante un sistema de bloques alternados, el cual consistió en dividir los contenedores en 10 secciones de 1 m cada uno, generando un total de 10 bloques de similar superficie en cada contenedor (cada bloque tuvo una superficie de 2 m^2). Este patrón se mantuvo durante todo el contenedor, cosechando una superficie total de 10 m^2. La cosecha se realizó extrayendo manualmente el musgo en los bloques impares y dejando intactos los bloques pares, es decir, se cosechó un bloque por medio. Se optó por este sistema con el objetivo de contar con material vegetal para la recolonización de los bloques cosechados. Luego de la cosecha, el material fue transportado al tendal para su secado por un período aproximado de 15 a 20 días. Posterior al secado, el musgo fue transportado nuevamente al invernadero para ser pesado una vez más.

Durante cada período de evaluación de 1 año, se realizaron 3 mediciones intermedias y una medición final del crecimiento en longitud de las hebras de musgo sembradas. Para ello, cada contenedor fue dividido en 3 secciones de 3,3 m cada una, y dentro de cada sección se extrajeron al azar 10 hebras de musgo obtenidas desde los bordes y desde el centro del contenedor. Por tanto, se midió una muestra de 30 hebras de musgo por contenedor. Se registró el crecimiento en longitud de las hebras mediante un pie de metro digital a partir de la elongación de los caulidios, el cual es impulsado por el proceso de fotosíntesis realizado mayormente en los capítulos de cada segmento.

Análisis Estadístico

En el caso de los análisis realizados sobre los parámetros morfológicos que fueron evaluados para determinar el mejor ecotipo, los datos se sometieron a una prueba de Kruskal-Wallis para 5 grupos con el propósito de determinar si existían diferencias estadísticamente significativas entre las 5 áreas muestrales trabajadas. Esto debido a que los datos no presentaron una distribución normal. Posteriormente, se analizaron las áreas muestrales en pares, con la prueba de Mann-Whitney-Wilcoxon para dos grupos, con el fin de determinar exactamente cuáles eran los sectores que presentaron diferencias en la distribución de su varianza.

RESULTADOS Y DISCUSIÓN

Selección de ecotipo a masificar

La descripción morfológica de parámetros cuantitativos pretendió determinar la existencia de diferencias entre estos parámetros que puedan ser considerados estadísticamente significativos entre las distintas procedencias. Se debe destacar la necesidad de identificar parámetros morfológicos idénticos, para finalmente obtener resultados que sean comparables entre un área muestral y otra. Los resultados de los parámetros cuantitativos se presentan a continuación.

Parámetros cuantitativos evaluados

En la Tabla 2 se presentan los valores promedios y las desviaciones estándar para las muestras de cada procedencia que fueron evaluadas en cada uno de los parámetros morfológicos cuantitativos considerados en el análisis.

Tabla 2. Valores promedio y desviaciones estándar para las mediciones de parámetros cuantitativos evaluados para cada procedencia de musgo *Sphagnum*.

Parámetro cuantitativo evaluado	Sector / Procedencia Ecotipo	Media $\overline{x}$ (mm)	Desviación Estándar (mm)
Largo de fascículos	Los Ulmos	8,800	1,937
	EL Palomar	9,400	2,500
	Quellón	8,333	2,454
	Quillaipe	9,267	2,180
	Aucar	10,067	2,333
Largo de filidios	Los Ulmos	2,113	0,340
	El Palomar	2,497	0,206
	Quellón	2,117	0,305
	Quillaipe	2,040	0,298
	Aucar	2,340	0,244
Ancho de filidios	Los Ulmos	0,990	0,195
	El Palomar	1,050	0,170
	Quellón	1,033	0,267
	Quillaipe	1,093	0,198
	Aucar	1,030	0,168

En el caso del parámetro largo de fascículos, el valor *p* arrojado por la prueba estadística de Kruskal-Wallis fue de $p = 0{,}023$, por lo tanto, existieron diferencias estadísticamente significativas entre las muestras de largo de fascículos entre los distintos sectores de procedencia examinados. Al aplicar la prueba por pares de Mann-Whitney-Wilcoxon los resultados arrojaron que el área de muestreo de Quillaipe presenta diferencia estadísticamente significativa con las áreas muestrales de Los Ulmos y Quellón. Sin embargo, estas últimas no presentan diferencia significativa entre sí. Las áreas de muestreo correspondientes a El Palomar y Aucar presentaron los promedios más altos de largo de fascículos extendidos. A su vez las áreas muestrales de Quillaipe, Los Ulmos y Quellón presentaron los valores más bajos. Se observó en terreno que el nivel freático era más superficial en las áreas muestrales de Aucar y El Palomar que en Quillaipe, Los Ulmos y Quellón. Considerando esta situación es probable que la longitud de los fascículos extendidos esté influenciada por las condiciones de anegamiento y, por tanto, mayor disponibilidad de agua, en la que se encontraban los sectores cuando se extrajeron las muestras.

En el caso del parámetro largo de filidios, el valor *p* arrojado fue de $p = 0{,}011$, por lo tanto, existieron diferencias estadísticamente significativas entre las muestras de largo de filidios en los distintos sectores de procedencia examinados. La prueba por pares mostró que las áreas de muestreo de El Palomar y Aucar presentaron diferencia estadísticamente significativa con todos los demás sectores, incluyendo diferencias mutuas. Al igual que el parámetro de largo de fascículos, el referido al largo de filidios puede estar fuertemente influenciado por un nivel freático más superficial en ambos sectores y una menor proporción de cobertura de protección de plantas herbáceas y arbustivas sobre el musgo.

Finalmente, en el caso del parámetro ancho de filidios, el valor *p* arrojado fue de $p = 0{,}276$, por lo tanto, no existieron diferencias estadísticamente significativas entre las muestras de ancho de filidios en los distintos sectores de procedencia examinados.

En relación a los resultados obtenidos, y realizando un consolidado de los 3 parámetros cuantitativos evaluados, se puede mencionar que las procedencias de El Palomar y Quillaipe fueron las que presentaron las mejores características morfológicas. No obstante, las muestras de El Palomar son superiores a las de Quillaipe, principalmente en los parámetros largo de fascículos y largo de filidios. Sin embargo, el ancho de los filidios se considera uno de los parámetros más importantes desde el punto de vista de las características comerciales del musgo, ya que un mayor ancho de filidio indica una mayor concentración de células hialinas dentro de este componente vegetal, lo que favorecería positivamente la absorción de agua, siendo este valor más elevado en las muestras obtenidas desde Quillaipe (Tabla 2).

Parámetros cualitativos evaluados

En la descripción de filidios de caulidio se apreciaron diferencias significativas, principalmente en su forma, concavidad y la coloración. Las áreas muestrales de Quillaipe, Quellón y Aucar presentaron forma de filidio espatulada, lo que conlleva mayor concavidad y menor superficie basal. En cambio, el área de El Palomar presentó forma de filidio ovalada en el ápice, lo que presenta como consecuencia una mayor superficie basal. Caso intermedio resultó el área de Los Ulmos, que presentó una mezcla de ambos tipos de filidios, en mayor proporción ovalados y en menor proporción espatulados (base muy angosta). Su concavidad fue la más alta de los cinco sectores muestreados.

En cuanto a la coloración, principalmente se diferenciaron por el engrosamiento de sus bases y bordes. En las áreas de Quellón y Quillaipe se apreció un color marrón oscuro, a su vez en las áreas de Los Ulmos, El Palomar y Aucar se observaron bases y bordes de color rojo oscuro a naranjo. Cabe destacar que la coloración de los filidios va a depender en gran medida de la radiación solar a la cual se encuentra expuesta la planta. Es así como aquellas que se encuentran con una exposición plena al sol, es decir, sin un grado de cobertura de protección desarrollan un mecanismo que les permite decolorar los filidios del capítulo. Esta estrategia permite que la radiación solar que llega a las plantas se refleje por la mayor tonalidad pálida que presentan, permitiendo una menor desecación y con ello una mayor tasa de fotosíntesis en los capítulos (Van Gaalen *et al.* 2007; Fukuta *et al.* 2012), aunque también existen evidencias que indican que los cambios en la coloración del musgo obedecen a disminuciones del contenido de clorofila y de su capacidad fotosintética (Harley *et al.* 1989).

En la descripción de filidios de fascículos se apreciaron diferencias significativas, principalmente en forma y concavidad. Las áreas muestrales de Los Ulmos y Quellón presentaron filidios con forma ovada. El área de Aucar presentó los filidios con forma lanceolada. Las áreas muestrales de El Palomar y Quillaipe no tenían una forma definida, se encontraron individuos con ambos tipos de forma lanceolada a ovada. La concavidad fue alta en las áreas muestrales de Los Ulmos, Quillaipe y Quellón, y media a moderada en las áreas muestrales de El Palomar y Aucar.

De acuerdo a los resultados obtenidos, el ecotipo de El Palomar es el que presenta la mayor superficie basal. A su vez Quillaipe presentó filidios espatulados y con mayor concavidad. La mayor concavidad de un filidio es una buena característica comercial en el sentido de que existe una mayor superficie de absorción. No obstante, sería adecuado conocer en qué sección del filidio se produce la mayor concentración de humedad al colocar el musgo en contacto con el agua. De esta manera se definiría con base científica cuál de las dos procedencias es la más adecuada.

Desde el punto de vista de la coloración, el color marrón anaranjado presente en el musgo de El Palomar lo descalificó al compararlo con la procedencia de Quillaipe, ya que este último presenta un tipo de coloración marrón, la cual es más demandada en los comercios internacionales.

Consulta a expertos

Los resultados que arrojó la consulta a expertos indicaron una clara preferencia hacia los ecotipos de Quillaipe y de Aucar. De acuerdo a sus apreciaciones, el ecotipo de Quillaipe presenta un adecuado largo y grosor de fibra, además de una adecuada densidad de filidios por hebra, lo que le entrega mayor estabilidad a la fibra. A su vez, el resumen de comentarios hacia el ecotipo de Aucar indica que el largo de la fibra es muy adecuado, no obstante, presenta deficiencias en el grosor de la fibra a lo que se suma una baja densidad de filidios por hebra, disminuyendo de esta manera la resistencia de la fibra.

Al consultarles por una asignación de valor en base a ponderaciones que entregarían a los parámetros de largo de hebra, ancho de fibra y capacidad de absorción de agua para determinar cuál de los ecotipos presenta mejores características para su comercialización, los resultados señalan que el mayor peso lo tiene el largo de la hebra, seguido por el grosor o ancho de fibra y finalmente la capacidad de absorción de agua. Desde el punto de vista cuantitativo los resultados estandarizados de ponderaciones entregados por los expertos consultados dan por conclusión que el parámetro largo de fibra tendría un valor de ponderación entre el 40 a 45 %, el ancho de fibra entre un 30 a 35% y la capacidad de absorción de agua valores de entre un 10 a 15%. De esta manera se estableció una ponderación estandarizada final para determinar, en base a los valores de los parámetros señalados, cuál de los ecotipos sería el más apropiado para comercialización. Los valores indican una ponderación final de un 45% para el largo de fibra, 35% para el ancho de fibra y un 20% para la capacidad de absorción.

Al aplicar estas ponderaciones sobre los valores medios obtenidos para cada parámetro, los resultados arrojan que el ecotipo Aucar sería el más adecuado con un valor de 14,6; seguido de Quillaipe con un valor de 12 y más atrás El Palomar con 7,1; Los Ulmos con 7 y Quellón con 4,5. No obstante, al añadir a esta evaluación las características cualitativas asociadas principalmente al color, el ecotipo seleccionado para el proceso de masificación artificial correspondió al de Quillaipe.

Evaluación del proceso de masificación artificial

La evaluación del proceso de masificación artificial consideró tres mediciones intermedias y una al final de cada período de un año. La primera medición intermedia del primer período se realizó al quinto mes de efectuada la siembra, coincidente con el inicio en el aumento de la radiación solar y la temperatura. Los resultados obtenidos mostraron un crecimiento promedio de 2,5 cm por fibra, equivalente a un crecimiento promedio de 0,63 cm por mes, crecimiento mayor a los descritos en la literatura para *Sphagnum magellanicum* en condiciones naturales. Es así como en ensayos de restauración de turberas en la Patagonia con siembras de hebras de este musgo en humedales explotados, el crecimiento fluctuó entre 2,5 a 5,0 mm por año (Domínguez & Larraín, 2013; Domínguez, 2014), proceso extremadamente lento para alcanzar la recuperación de la turbera. Para turberas secundarias cercanas a Puerto Montt (región de Los Lagos), la evaluación del crecimiento y productividad de este musgo realizado en nueve turberas, donde se consideraron dos situaciones microtopográficas distintas, los valores de crecimiento reportados fueron de 0,22 cm/mes para los montículos y de 0,43 cm/mes para sectores planos (Díaz *et al*. 2012). De acuerdo a Schofield (2001), la tasa de crecimiento de *Sphagnum* varía ampliamente. El crecimiento es predominantemente apical e indeterminado y disminuye a medida que la altitud aumenta.

La evaluación del crecimiento en condiciones controladas se considera exitosa, pues el registro incluye gran parte del período invernal, ya que en condiciones naturales este período no es favorable para el desarrollo del musgo, debido al menor nivel de radiación solar. Para Tapia (2008), la mayor tasa de crecimiento en turberas secundarias de la provincia de Llanquihue ocurrió entre los meses de enero a marzo.

Antes de realizar la primera cosecha, se registró una nueva medición del crecimiento de las fibras con el fin de conocer el comportamiento de la producción luego del período estival. Los resultados del crecimiento fueron mucho mejores que los obtenidos inicialmente, corroborando de esta manera lo señalado por Tapia (2008). Se apreció un notable desarrollo de los capítulos del musgo con una mayor elongación de caulidios y filidios, mostrándose más alargados y engrosados, lo que fortalece la fibra. El color verde intenso del musgo en los meses de enero y febrero contrasta con la tonalidad marrón que presenta en el mes de marzo, debido al efecto de la malla sombreadora.

Los registros de desarrollo para esta medición indican un crecimiento acumulado promedio de 4,5 cm en un período de 9 meses, con valores de hasta 5 cm en este período. Estos registros indican un crecimiento promedio mensual desde 0,5 a 0,55 cm/mes, lo que reafirma las mediciones anteriores y dan cuenta de una muy buena tasa de crecimiento en condiciones controladas del musgo *S. magellanicum*.

Los resultados del crecimiento obtenidos en el primer período productivo de 1 año se presentan en las Figs. 5 y 6.

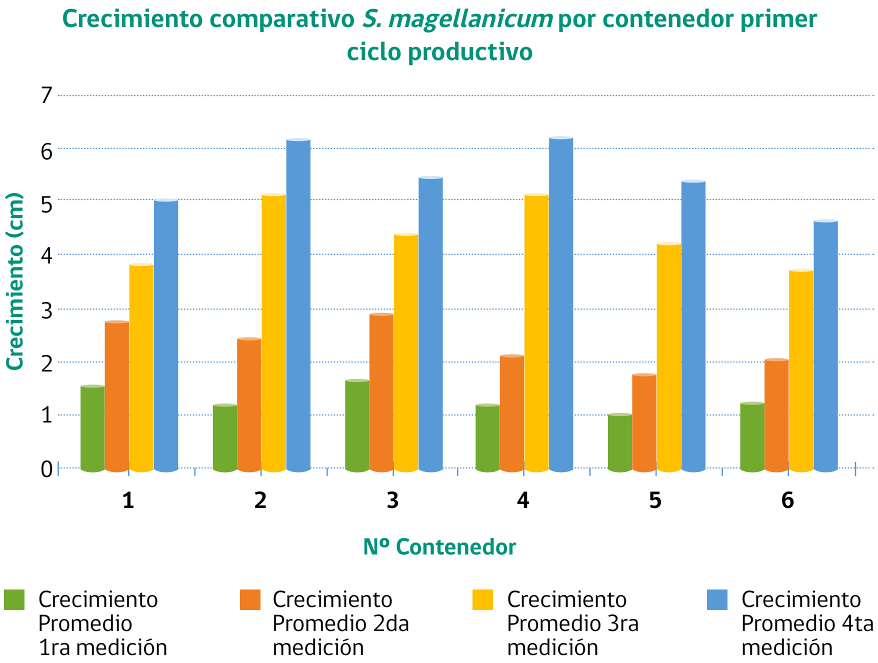


Fig. 5. Gráfico de barras con los niveles de crecimiento en longitud de la fibra del musgo *S. magellanicum* en condiciones de crecimiento artificial durante el primer ciclo productivo.

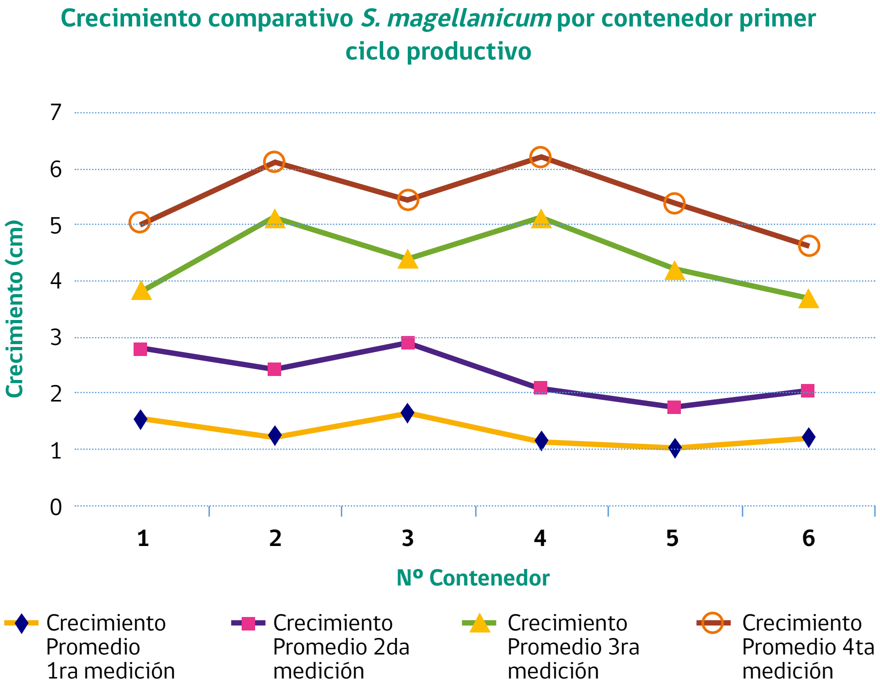


Fig. 6. Gráfico de tendencia con los niveles de crecimiento en longitud de la fibra del musgo *S. magellanicum* en condiciones de crecimiento artificial durante el primer ciclo productivo.

En ambas figuras se observa que los niveles de crecimiento del musgo son muy positivos, los que alcanzaron un promedio de 5,46 cm al final del primer ciclo productivo de 1 año, equivalente a un crecimiento promedio mensual de 0,46 cm. Este fue levemente superior al documentado para turberas antropogénicas en la región de Los Lagos, donde se indican niveles de crecimiento de 5,16 cm/año (Díaz *et al.* 2012), y muy superior a los registrados para turberas naturales de la región de Magallanes, donde los valores promedio de crecimiento anual son de 0,5 cm (Domínguez, 2014).

Se cosecharon 50 kg en peso fresco del musgo lo que consideró sólo la mitad de la superficie de cada contenedor de acuerdo al sistema de cosecha indicado en la metodología. Luego del secado en tendal el musgo cosechado fue pesado alcanzando 4 kg. Este rendimiento en biomasa es bastante auspicioso al compararlo con turberas naturales donde se ha descrito que para un volumen de 50 kg de musgo cosechado se obtienen sólo 2 kg de musgo seco (Domínguez, 2014).

El segundo ciclo productivo culminó con la cosecha realizada a fines del mes de junio de 2016. Antes de realizar la cosecha se registraron crecimientos de las hebras cercanos a los 10 cm promedio durante el año que se mantuvo el musgo dentro de los contenedores, es decir, un crecimiento de 0,83 cm/mes (Fig. 7).

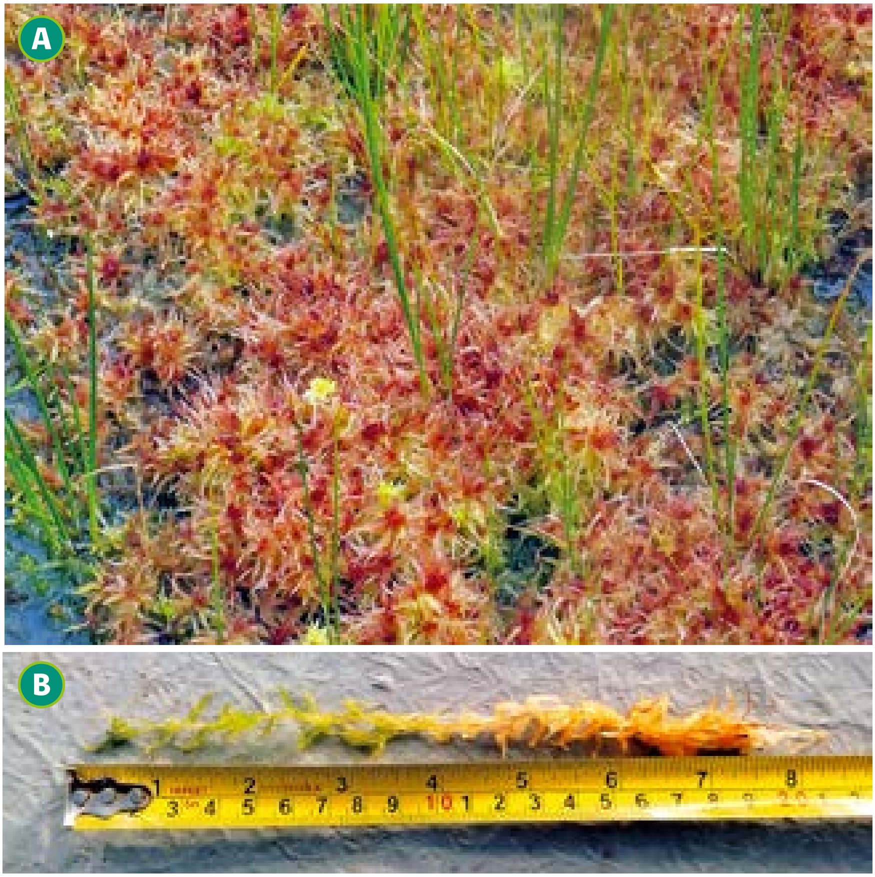


Fig. 7. **A)** Estado del musgo en los contenedores antes de la segunda cosecha y **B)** Crecimiento del musgo medido antes de la segunda cosecha.

Los resultados del crecimiento obtenidos en el segundo período productivo de 1 año se presentan en las Figs. 8 y 9.

En el segundo período productivo los crecimientos aumentaron, llegando a niveles promedio de 9,3 e incluso 9,8 cm de crecimiento en longitud de la fibra, lo que da cuenta de un aumento en la productividad del sistema artificial. Esta situación indica un crecimiento promedio de 0,78 cm/mes, lo que se considera muy favorable en comparación con los niveles de crecimiento naturales del musgo en turberas antropogénicas. Dentro de este período se realizaron las mismas mediciones en relación al tiempo de ejecución y metodología de toma de muestras que las realizadas en el primer ciclo.

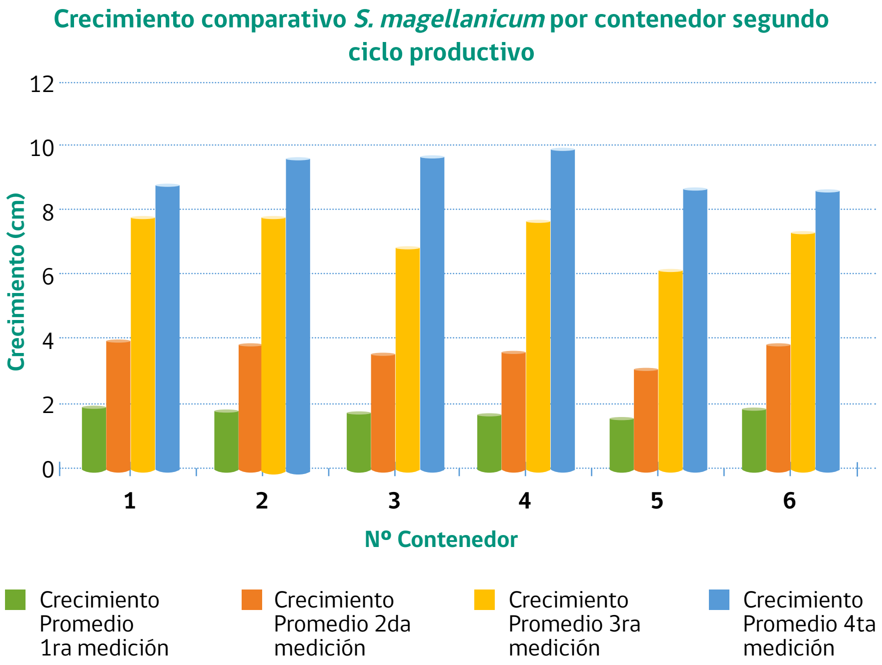


Fig. 8. Gráfico de barras con los niveles de crecimiento en longitud de la fibra del musgo *S. magellanicum* en condiciones de crecimiento artificial durante el segundo ciclo productivo.

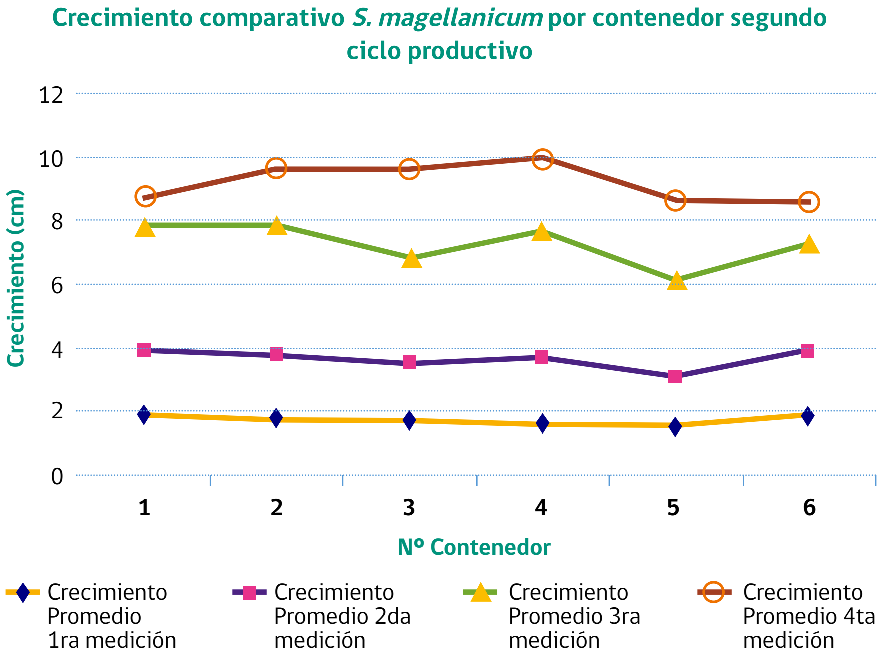


Fig. 9. Gráfico de tendencia con los niveles de crecimiento en longitud de la fibra del musgo *S. magellanicum* en condiciones de crecimiento artificial durante el segundo ciclo productivo.

Para la segunda cosecha el valor de peso fresco de musgo obtenido fue de 55 kg por contenedor en promedio, aumentando la productividad al comparar este parámetro con la primera cosecha. Al igual que en la cosecha inicial, el musgo fue trasladado al tendal para su secado. El peso promedio seco por contenedor fue de 4,5 kg, o sea, 0,5 kg más que el peso con respecto al primer ciclo productivo, lo que demuestra la posibilidad de generar un rendimiento sostenido durante períodos sucesivos de producción.

Sin duda la opción de masificar ecotipos morfológicamente superiores destinados a la producción comercial o a la restauración de turberas degradadas se transforma en una posibilidad real y comprobada para abordar estos desafíos. No obstante, es necesario generar alternativas en las cuales el proceso de masificación pueda realizarse sobre la base de menores costos asociados a la infraestructura a través de un sistema de producción artesanal.

Los resultados generados en esta experiencia representan un punto de partida que necesariamente debe ser complementado con mayores antecedentes obtenidos a partir de nuevas e innovadoras experiencias en el proceso de masificación, una mayor investigación *in situ* y a nivel de laboratorio para conocer qué otros factores tienen mayor influencia en un mayor desarrollo del musgo y en un menor tiempo de rotación.

CONCLUSIONES

Los resultados obtenidos durante la ejecución de esta iniciativa dan cuenta de una exitosa producción artificial del musgo de turbera *Sphagnum magellanicum* Brid. bajo condiciones controladas, lo que se manifiesta en que los niveles de crecimiento longitudinal de las fibras del musgo alcanzaron un promedio de 0,46 cm por mes para la primera temporada y de 0,78 cm por mes para la segunda temporada, valores que superan en un 6% y 80% respectivamente los antecedentes de crecimiento registrados en turberas naturales secundarias ubicadas en la región de Los Lagos. No obstante, este mayor crecimiento longitudinal no fue acompañado por un mayor desarrollo en el grosor de las fibras, lo que perjudicó la obtención de mayores volúmenes de biomasa y con ello el peso seco final de la producción por temporada.

Mediante los procedimientos de masificación artificial implementados en esta iniciativa fue posible demostrar que es viable generar una producción sostenida de este recurso en el tiempo, aumentando incluso las tasas de crecimiento en temporadas sucesivas y acortando los tiempos de rotación al compararlo con la respuesta del musgo en ambientes naturales.

La masificación sostenible del musgo *Sphagnum magellanicum* debe considerar un monitoreo permanente de los factores que condicionan su crecimiento. Dentro de éstos, la cantidad de agua disponible para el musgo y la mantención de rangos en ciertos parámetros físicos y químicos de este elemento tales como el pH, son claves en el desarrollo de *Sphagnum*. A ello se suman los niveles de radiación y de temperatura, especialmente en la época estival, los cuales son necesarios de monitorear permanentemente.

La creciente y sostenida demanda internacional por el musgo *Sphagnum magellanicum* Brid. ha traído como consecuencia una indiscriminada actividad extractiva, principalmente en la región de Los Lagos, situación que está generando significativos impactos ambientales y sociales, y un progresivo deterioro en la calidad del musgo que se comercializa. Por tanto, se hace necesario el resguardo y la generación de políticas de sustentabilidad de los ecosistemas de turbera y un reglamento que entregue las normativas y directrices necesarias para realizar una extracción sustentable del recurso, lo cual impactará positivamente en la mantención del rol ecológico que estas áreas cumplen a nivel global y la gran diversidad de servicios ecosistémicos que entregan a nivel local.

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