Establishment and development of a *Typha* paludiculture site – lessons learned from a 10-ha pilot site

Meline M. Brendel[1], Nora Köhn[1], Bernd Bobertz[2], Janina Deininger[1], Kerstin Haldan[1], Christina Hellmann[2], Hans Joosten[1], Matthias Lampe, Sebastian van der Linden[2], Josephine Neubert[1], Duc-Viet Nguyen[2], Silja Seiler[1], Sabine Wichmann[1], Juergen Kreyling[1]

[1]Institute of Botany and Landscape Ecology, University of Greifswald, Partner in the Greifswald Mire Centre, Germany
[2]Institute for Geography and Geology, University of Greifswald, Partner in the Greifswald Mire Centre, Germany

SUMMARY
Natural peatlands are carbon sinks, but drainage turns them into CO_2 sources. Rewetting minimises the CO_2 release and protects the stored carbon. Almost all temperate European fens have been drained for agriculture. Paludiculture is the productive use of rewetted peatlands, e.g., by growing *Typha* spp., with multiple usage options such as building or insulation material. However, data on establishment and development of sites with an economically meaningful size are lacking. In 2019, we established a 10-ha *Typha*-paludiculture pilot site in NE Germany to test practical feasibility at field scale. This study assessed site development over the first three years (September 2019–December 2022). Density categories based on *Typha* and accompanying species were defined in the first year and their development followed. Biomass productivity increased up to 7.43 ± 1.88 t ha^{-1} on average in December 2022 for density category "dense *Typha*" and was not affected by a mechanical harvest in December 2021. The development of the *Typha* stand and the accompanying species was analysed with permanent plots and main drivers were assessed using boosted regression tree analysis (BRT) and non-metric multidimensional scaling (NMDS). The number of *Typha* plants was strongly linked to accompanying species cover, the proportion of *T. angustifolia* to *T. latifolia* shoots and the mean annual water level. Despite an initially unsuccessful establishment after planting, *Typha* increased dominance over time after seeding by hand and drone, whereas accompanying species decreased over the first three years, as shown by remotely sensed imagery. Nutrient concentrations in both pore- and surface water showed no consistent pattern. For example, NH_4 increased over time at both the inlet and the potential outlet of the pilot site, whereas Fe became more available in porewater only at the potential outlet, but not at the inlet. Practical lessons learned for establishing large *Typha* sites are that (1) maintaining the water level above soil surface proved suitable for the successful establishment of *Typha* and may control accompanying vegetation cover at our site, and that (2) late planting in autumn was likely one important factor contributing to the low planting success. Seeding by hand or drone, however, provides promising measures to fill gaps in the *Typha* stand and should be further tested as exclusive method for whole stand establishment.

KEY WORDS
biomass – cattail – fen – nutrients – rewetting

INTRODUCTION
Natural peatlands store large amounts of carbon and nutrients such as nitrogen in their peat (United Nations Environment Programme 2022), as their waterlogged environments slow down decomposition (Joosten & Clarke 2002). Draining peatlands for agriculture or other purposes leads to peat mineralization surpassing carbon input, causing drained peatlands to emit approximately 2 Gt CO_2 per year, i.e., approximately 5 % of the global anthropogenic greenhouse gas (GHG) emissions (Joosten *et al.* 2016, Loisel *et al.* 2021). Rewetting drained peatlands can substantially lower these net GHG emissions. In Germany, rewetting drained peatlands used for crop production can decrease emissions by approximately 35 t CO_2eq ha^{-1} y^{-1} (implied emission factor of drained croplands on peat: 40.4 t CO_2eq ha^{-1} y^{-1}). For drained peatlands used as grasslands, reductions of up to 26 t CO_2eq ha^{-1} y^{-1} are possible (implied emission factor of drained grasslands: 31.7 t CO_2eq ha^{-1} y^{-1}). In comparison, rewetted peatlands emit approximately 5.5 t CO_2eq ha^{-1} y^{-1} according to national emission factors derived by Tiemeyer *et al*. (2020). Indeed, peatland rewetting is considered to be one of the most cost-efficient options for GHG mitigation (Röder *et al.* 2015). Rewetted peatlands that were agriculturally used in the past can still be used for wet agriculture or forestry (Wichtmann *et al.* 2016). This so-called paludiculture refers to the productive use of wet peatlands; growing crops adapted to waterlogged conditions, using adapted harvesting machinery and enabling many biomass utilisation options (Joosten *et al.* 2016, Wichtmann *et al.* 2016, Geurts & Fritz 2018, Tanneberger *et al.* 2020). Recent field studies indicate that fen paludiculture can

even function as a net GHG sink, with reported emission factors of −12 t CO_2eq ha^{-1} y^{-1} under rewetted conditions and mitigation potential of up to −52 t CO_2eq ha^{-1} y^{-1} (Bockermann *et al*. 2025).

Paludiculture biomass can be used for insulation and building material, as biorefinery material, as raw material for growing media, as well as for fodder or fuel (Giannini *et al.* 2016, Tanneberger *et al.* 2020, Krus *et al.* 2024, Wichmann & Nordt 2024). *Typha* is a paludiculture crop with broad utilisation options; due to the high proportion (up to 85 % of the plant mass) of air-filled tissue (aerenchyma), *Typha* is a promising insulation material with a low density and a high break resistance as well as load-bearing capacity (Georgiev *et al.* 2014, Krus *et al.* 2014, Haldan *et al.* 2022).

Research on *Typha* productivity and ecology has a long history, spanning natural stand observations, controlled experiments, small field trials and recent large-scale paludiculture pilots. Early work already demonstrated the remarkable biomass productivity of *Typha* species in natural wetlands; Dubbe *et al.* (1988) reported above-ground dry matter production of up to 21 t ha^{-1} for *T. angustifolia* in Minnesota. Studies by Grace and Wetzel (1981, 1982) highlighted that growth is strongly dependent on water level dynamics, as is competition, with *T. latifolia* outcompeting *T. angustifolia* in shallow water levels. In fen-restoration studies in north-east Germany, *Typha* successfully established both after planting and sowing (Roth *et al.* 1999). In Europe, the first attempts to utilise rewetted peatlands for *Typha* cultivation were made in a demonstration project that examined the sustainable cultivation of *Typha* as a building material in a rewetted fen in the "Donaumoos" valley of the Danube river (Pfadenhauer & Wild 2001, Wild *et al.* 2001). In three experimental sites that had a total *Typha*-stocked area of 6.2 ha, Pfadenhauer and Wild (2001) and Wild *et al.* (2001) tested the establishment and biomass production of *Typha* stands under managed water levels. Their work confirmed an initial influence of planting density which levelled out after three years and an effect of site hydrology on establishment and productivity and, furthermore, proved the potential of *Typha* for water purification.

Controlled and mesocosm studies further disentangled the environmental factors determining *Typha* performance. Productivity of *Typha* was found to be remarkably independent of low nutrient availability (Vroom *et al.* 2018, Ren *et al.* 2019). Yang *et al.* (2019) assessed the water level fluctuation requirements of *T. angustifolia*, conducting field investigations at lakes with sizes ranging from 2.0 km^2 to 2933.0 km^2 and simulation experiments. They found two phenotypic forms of *T. angustifolia*; one "tall" form (2.1–3.3 m) growing in lakes with intermittent or quasi-naturally fluctuating water levels and a "short" form (0.6–1.6 m) growing in lakes with reservoir-like (relatively small annual) fluctuations.

Within the CINDERELLA-project, several small-scale paludiculture pilots were investigated in the Netherlands (Geurts & Fritz 2018). Under controlled rewetting conditions, they examined the cultivation of *Typha* and *Phragmites* and identified that *Typha* performs best when nitrogen supply reaches at least 100 kg N ha^{-1} y^{-1} (Geurts & Fritz 2018, Geurts *et al.* 2020). Furthermore, stable water levels and appropriate soil preparation were identified as key prerequisites for high yields and successful *Typha* establishment. Eickenscheidt *et al.* (2023a) and Nordt *et al.* (2024) provide comprehensive guidelines for the establishment and management of paludiculture systems, covering site requirements, site preparation, sowing and planting techniques and subsequent site management.

Despite these advances, empirical knowledge on *Typha* establishment and stand development at scales relevant for agricultural application remains scarce. Most previous studies were conducted under controlled or pilot-scale conditions and covered limited spatial and temporal extents: mesocosm experiments typically encompassed only a few square meters and short cultivation periods of one to three years, while pilot projects such as the one in southern Germany (6.2 ha total; Pfadenhauer & Wild 2001, Wild *et al.* 2001) and the project in the Netherlands (largest plot with a size of approximately 0.7 ha; Geurts & Fritz 2018) provided valuable insights but these remain below the threshold of mechanised and economically feasible production. Consequently, little is known about the processes and constraints that determine the establishment and early development of *Typha* stands under real-world management conditions.

Consequently, more studies on the establishment and the potential drivers of stand development at scales large enough for usage of machinery and without border effects on nutrient supply are needed to gain data and draw conclusions on *Typha* paludiculture at scales relevant for practice. Therefore, in 2019, we installed a 10 ha-sized paludiculture pilot site on a formerly drained fen peatland in north-east Germany and established *Typha* on 8.5 ha.

This paper describes the establishment and development of a *Typha* paludiculture site over three years (2019–2022) with a focus on *Typha* shoot density and biomass production, as well as accompanying plant species and site conditions. We hypothesised that (1) the *Typha* stand increases in *Typha* shoot number over time with increasing water level above soil surface, that (2a) the species composition of the accompanying vegetation changes over time, shifting to more flooding-tolerant species and thus, (2b) the total species composition is mainly driven by the water level. Lastly, we hypothesized that (3) the pore- and surface water

nutrient contents decrease from the inlet towards the potential outlet, reflecting the filter function of the *Typha* stand.

MATERIAL AND METHODS

Pilot site

The pilot site is a 10 ha-sized field in the polder "Teichweide" near the town Neukalen in the German federal state of Mecklenburg-Vorpommern in north-eastern Germany (53°49'33.0"N 12°45'08.8"E). The thickness of the peat and intermingled limnic sediments ranges from about 2 m to 6 m at the site (Figure A1). The subjacent bed consists of glacio-fluvial sand and gravel with bands of silt and clay. The climate is temperate with a mean annual air temperature of 9.2 °C and mean precipitation of 583 mm in the 1991–2020 climate interval (Deutscher Wetter Dienst 2023).

The pilot site is part of a former fen mire located next to the river Teterower Peene. Above the glaciofluvial deposits lie silty sediments with increasing organic content, indicating that there existed a large body of still water in the valley basin before the formation of the fen. Following the recession of the lake, medium to very strongly decomposed sedge and reed peats with a high proportion of limnic sediments formed. The centre of the fen profile also contains a significant proportion of wood. The fen peatland has been drained since the 19th century, with intensive drainage and establishment of a polder at the end of the 1960s in the course of the GDR complex amelioration. The polder was used for fodder production and as a pasture for cattle and since 1996 as low-intensity grassland. The species composition of the grassland vegetation indicated fluctuating water levels, probably with dry periods in summer and wet conditions during winter and into spring. Therefore, species that can also grow under wet conditions occurred already before rewetting, especially *Phalaris arundinacea* as the dominant species. In 2019, this agriculturally-used grassland was rewetted within the project Paludi-PRIMA (funding code 22026017; Neubert *et al.* 2022). To do so, parallel dikes and ditches were built around the part of the polder to be rewetted, also to prevent influences of rewetting on the surrounding drained grassland. Dikes were 700 m in total at the northern and eastern side and were connected to the existing peat dike in the southern and western part of the pilot site. Dikes were made of peat extracted from the site (ditch excavation, sod removal) and had an overall height of 1.25 m above surface and a crown width of approximately 3 m during construction. Ditches were built outside of the dike in the northern part to collect seepage water (depth: approximately 1.5 m, width: approximately 4 m) and inside of the dike in the eastern part of the site for better distribution of the irrigation water (depth approximately 0.5 m, width approximately 1.8 m).

During construction, the site was levelled out with a caterpillar dozer by removing the sod on areas with an elevation higher than 60 cm above sea level. A drainage ditch was filled and underground drainage pipes were destroyed. The sward had been pushed away or severely weakened by the caterpillar in most parts of the site (approximately 6.1 ha, Figure A2). Therefore, additional large-scale soil preparation such as ploughing or tilling was unnecessary. Excluding the dikes and ditches, 25,000 *Typha angustifolia* and 25,000 *Typha latifolia* seedlings that were grown in seed trays (pot size 5 cm x 6 cm) were planted in two planting densities (approximately 1 plant m^{-2} and 0.5 plant m^{-2}, Figure A3) on 16–20 September 2019, using two single-row tree planting machines (RPKU type, Räumpflanzkombine Universal), one of them equipped with a furrow opener creating larger furrows and the other without such additional device resulting in planting the seedlings in small furrows. In the south-western part, both machines failed with planting into the remaining sod. Thus, particularly wide and deep furrows resulted from an additional furrows plough used prior to planting. Whereas the distance between the rows was planned to be two meters, the control after planting revealed an average distance of approximately 2.20 m. After planting, the water level was raised actively by pumping water into the site from the adjacent river until it was approximately at surface level on 26 September, 2019. Plant establishment was poor, probably due to the following reasons: planting near the end of the growing season, early frosts in October 2019 and late frosts until May 2020, washing out in flooded furrows during winter and disturbance by waterfowl (Neubert *et al.* 2022). Based on counting the new *Typha* plants in every second row on 4 May 2020, a planting success of approximately 16.9 % for *T. latifolia* and 6.7 % for *T. angustifolia* (approximately 13 % in total) was calculated. However, these values were likely underestimations since the emergence of shoots was still incomplete due to cold nights and partly inundation of the site. To improve stand establishment, seeds of both *Typha* species were additionally sown by hand (3.5 ha) and by drone (4 ha) in June 2020. Seeds were obtained by collecting spadices, which were frozen to kill off pests. Subsequently, seeds and fluff were separated and pellets were produced which were distributed on the site. Pellets for hand-seeding were made of clay and water, with a size of 20–30 mm, and pellets for drone-seeding were made of bentonite, starch and water with a size of approximately 2–4 mm. Both types of pellets dissolved in water within hours to few days, so that the seeds received sufficient light for germination. Furthermore, in the beginning of the vegetation period 2020, *Typha* germination from the seed bank was observed. In 2020, the

water level was fluctuating (approximately 30–90 cm a.s.l., mean site elevation 56 cm), due to the lack of permanently available water pumping equipment due to delivery delays during the COVID-19 pandemic and malfunctioning of the first emergency power generator. Therefore, rented equipment had to be used periodically in 2020. From the vegetation period 2021 on, a generator-driven water pump (waste water pump type Nautic 3.7 N-CD, pumping capacity 72 $m^3 h^{-1}$, 3.7 KW) was continuously available to adjust the water level as required and in June 2021, an additional solar water pump (Lorentz PS2 4000 deep well pump, type PS4000 C-SJ42-2, capacity max. 61 $m^3 h^{-1}$) was installed to react directly to daily peaks in evapotranspiration. In the following, the water pumps in the south-east of the site are denominated as the inlet. Two adjustable outlets were installed in the south-east and at the opposite end in the north-west, respectively (Figure 1). The weirs of the potential outlets were usually set so high that no water was observed to leave the site except for the instances of high water levels before machinery harvest, when the water level was actively lowered by lowering the weir from 01 November until 11 November 2021. In December 2021, a first mechanical trial harvest of a small area (approximately 1.3 ha) in the north-west of the field took place (Neubert *et al.* 2022, Figure A4). The harvest machine was a caterpillar-based Softrak 120 (Loglogic) with an ELHO double chopper as front attachment that produced chopped material.

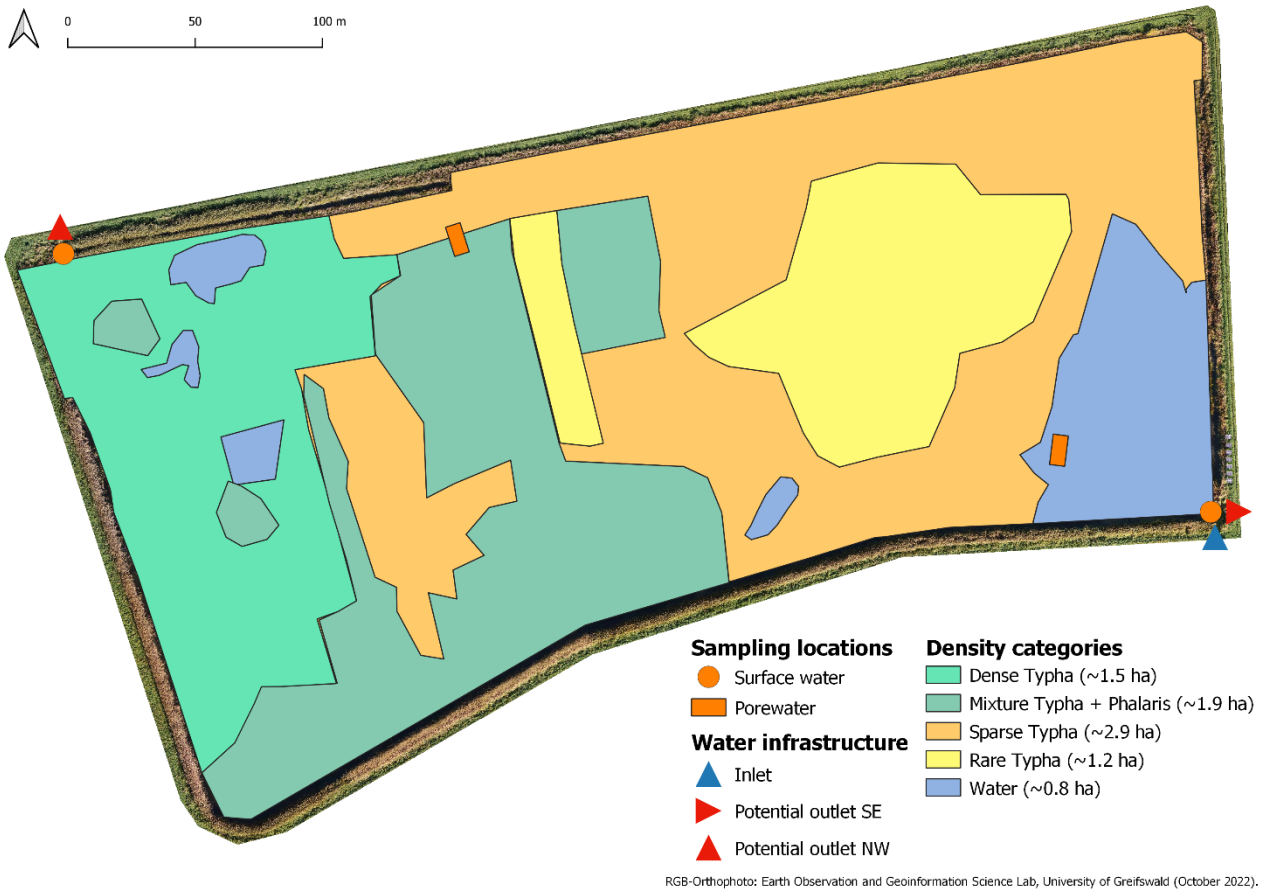


Figure 1. Density categories of *Typha* vegetation at the pilot site. The five density categories "dense *Typha*", "mixture *Typha* + *Phalaris*", "sparse *Typha*", "rare *Typha*" and "water" were defined in 2021 using visual field observations and drone imagery. Note that outlet SE is kept above outlet NW during times with active pumping through the inlet.

Vegetation monitoring

Starting with the first vegetation period after planting (2020), vegetation surveys were carried out every year in July to depict the plant community at the peak biomass production of *Typha*, which is reported to be in early summer (Fiala 1971). To do so, 20 randomly located plots within each planting area, yielding 80 plots in total, were installed and repeatedly monitored each year (Figure A3). Although establishment after planting was poor, the 80 plots remained evenly distributed across the pilot site; therefore, we maintained the original plot design. During the vegetation survey, *Typha* shoots were counted, the *Typha* cover, the cover of the total accompanying vegetation and of each individual plant species and the cover of open water were estimated.

Sum of cover of *Typha*, accompanying vegetation and open water were set to 100 %. The coverage of each species was estimated as percentage integer numbers.

In June 2021, we defined four density categories of *Typha*, distinguished by visual impression and drone pictures. A fifth category was assigned to areas with open water surfaces without *Typha*. The five categories are namely, "dense *Typha*", "mixture *Typha* + *Phalaris*" (*Typha* stand was also dense in this category, yet mixed with *Phalaris*), "sparse *Typha*", "rare *Typha*" and "water" (Figure 1).

Due to the overall limited establishment by planting and the subsequent seeding as well as an observed germination from the seed bank, the original planting design could no longer be treated as a valid experimental contrast. This limits the use of inferential statistics and the causal interpretation of treatment effects. Therefore, all subsequent analyses were conducted within an explicitly exploratory framework. Nevertheless, potential residual effects of the initial planting treatments could not be ruled out and were retained as explanatory variables in the statistics to assess whether such potential effects were still present. Species composition and the underlying environmental drivers were assessed with a non-metric multidimensional scaling (NDMS) using the package *vegan* in R (Oksanen 2019). Since the stress was > 0.2 when calculating a two-dimensional NMDS, we decided for a three-dimensional NMDS (stress 0.15) to be sure that the distribution of the species and sites was adequately depicted by the NMDS (Kruskal 1964, Clarke 1993). NMDS axes 1 and 3 were chosen for visualization, as the significant explanatory variables were mostly distributed along these axes. The assessed environmental variables were year and average water level as numerical explanatory variables, and planting density (two levels: 0.5 plants per m^2 / 1 plant per m^2), "mown December 2021" (two levels: yes / no; 14 of 80 plots were mown), sod presence (two levels: sod removed / sod presence), size of the planting furrows (five levels: no furrows (planting machine without furrow opener) / small furrows (planting machines with furrow opener) / partially no furrows + partially small furrows / large furrows (site preparation by furrow plough) / partially no furrows + partially large furrows) and density category (five levels: water / dense *Typha* / mixture *Typha* + *Phalaris* / sparse T*ypha* / rare *Typha*), as factorial explanatory variables. Plots were assigned to categories of sod presence and furrows based on the records taken during planting and a drone picture from October 2020 after planting. If less than 10 % of a plot was covered with sod, it counted as sod removed. Subsequently, the correlation of the environmental drivers with the species composition was analysed in the ordination space using the *envfit*-function of package vegan version 2.6-4 (Oksanen 2019). Finally, an Analysis of Similarities (ANOSIM) was calculated to detect the differences in accompanying species between the groups of the respective environmental variables using the *anosim*-function of the package vegan (Oksanen 2019). These explorative methods were used because the initial planting densities were potentially rendered obsolete due to additional germination from the seed bank and after the sowing actions, yet their influence could also not be ruled out. Similarly, the furrows and the sod removal did not occur in a properly replicated and interspersed experimental design. Inferential statistics were therefore deemed inappropriate.

A boosted regression tree analysis (BRT) was used to investigate the relationship between the development of the *Typha* stand, expressed as number of *Typha* shoots per plot, and environmental variables potentially affecting this development. BRT was chosen for being able to run it on different levels of measurements (numerical and factorial in the same model) and its known ability to cope well with correlated and potentially interacting predictors (Perez *et al.* 2022), even though our dataset was relatively small. We used the package *gbm* in R and published guidelines on how to fit the models (Ridgeway 2007, Elith *et al.* 2008). A tree complexity of 3, learning rate of 0.08 and a bag fraction of 0.8 were chosen based on trials to minimise the estimated cross-validated deviance (Elith *et al.* 2008).

Several proxies describing the water condition of the plots were considered as environmental variables in the BRT, including mean water level in July, annual mean, maximum and minimum water level values (from July of the previous year to July of the subsequent year), and number of days with water level below soil surface. Moreover, the effect of the cover of accompanying species and the species composition, depicted by the NMDS axes values were assessed as explanatory variables for the number of *Typha* shoots per plot. To reduce the number of predictors, univariate GAMs (generalised additive model) were fitted. The GAMs for the water and the vegetation proxies were compared, and for both groups the proxy with the highest r^2 value was chosen as a representative for the respective group. For the accompanying species this was NMDS1, for the water level characteristics this was annual mean water level. Although both *Typha* species were planted in different parts of the site (Figure A3), *T. latifolia* seemed to dominate on the whole site based on visual inspections in 2020, presumably mainly by germination from the seed bank. Later, though, both species occurred in mixture, with *T. angustifolia* becoming more frequent over time. To account for this development in the statistical analyses, we recorded the presence of *T. angustifolia* in each plot in 2023, even though regular sampling had already ended. A quantitative distinction between the two species (based on their shoot abundance per plot) was then carried out in 2024. Therefore, we report all results without differentiating

between the two species but explore potential differences between them by using the share of *T. angustifolia* per plot as additional predictor in the BRT-model. Consequently, predictors for the BRT-model were average water level during the sampling times, NMDS scores of axis 1, year, cover of the accompanying vegetation, proportion of *Typha angustifolia* in 2024, sod presence, furrows and if the plots were mown in December 2021.

Due to a lack of data on water level/soil elevation and *Typha* fitness traits such as plant height/ diameter in 2020, the NMDS and BRT analysis on the main drivers for the stand development were done only for the years 2021 and 2022.

To assess the relation between the accompanying vegetation and the water levels, linear models were applied for the single years 2020 through 2022, respectively.

All statistical analyses were conducted using the software R, version 4.4.0 (R Core Team 2024).

Water levels

Soil surface elevation was used as a reference for water level, since direct water level values were not available for all plots in all years. We assumed that the water level was the same across the site and that the water level could therefore be determined from the elevation of any specific point. The water level in the pilot site was measured with an ElliTrack water logger (ElliTrack-D water level loggers, Leiderdorp Instruments BV), which was positioned in the centre of the pilot site. The logger registered the water level hourly from 17 June 2020.

In 2020, the elevations of 50 out of 80 plots were measured with a differential GPS (GPS device EMLID Reach Rs+, a single-band RTK GNSS receiver with centimetre precision). In autumn 2022, the elevation of each plot was measured the same way (using the Leica GNSS-Rover ATX1230+RX1250 with SAPOS-correction data). Yet, to cope with the lacking elevation values for the missing 30 plots in 2020 and for all plots in 2021, the difference in elevation for the 50 plots that had values for 2020 and 2022 was calculated. We used a linear model to calculate the model-based difference between the plot elevations in 2020 and 2022 which was then subtracted from the plot elevations in 2022. For 2021, we assumed that the soil elevation increased linearly, and subtracted half of the calculated difference-value based on the linear model from the soil elevation values from 2022 (Figure A5). Since there was no plot elevation data for 2021 and only some data for 2020, it was decided to extrapolate the elevation and water level data for all plots for all three years based on the linear model for the plot elevations.

In 2022, the water levels were measured manually with a ruler in every plot from the soil surface. Subsequently, we plotted the measured water level values of 2022 versus the extrapolated values based on the elevation of 2022 to check if the relationship is linear (Figure A6). To depict the water level in the pilot site during the study period, we averaged the water level of the time from one sampling period to the next one, i.e., for sampling in July 2021 (water level data from 10 July 2020 until 13 July 2021) and for July 2022 (water level data from 14 July 2021 until 7 July 2022). Yet, plants are not only influenced by the average water level. To take this into account, we considered further water level characteristics i.e., the number of days with the water level being above or below soil surface as well as minimum and maximum values reached during the year. We calculated these variables per day based on the hourly measurements of an entire sampling year (7 July 2020 until 13 July 2021 and 14 July 2021 until 7 July 2022).

Development of *Typha* biomass on the pilot site

From 2021 on, we assessed the development of the *Typha* biomass on the pilot site every year in July and in December. Due to the delayed establishment of the *Typha* plants, initial biomass sampling to assess yield was postponed to the second vegetation period. For each density category, ten 1-m^2 plots were randomly chosen in the geoinformation system QGIS. A minimum distance of 15 m between plots and at least 3 m to the edges of the site was set. For each sampling period, new, randomly selected plots were sampled. In December 2021, a mechanical trial harvest took place in parts of density categories “dense *Typha*” and “mixture *Typha* + *Phalaris*”. Thus, for July and December 2022, we differentiated between the initial four density categories as well as harvested density categories “dense *Typha*” and “mixture *Typha* + *Phalaris*”. *Typha* biomass of each plot was harvested by cutting the plants by hand using knives at an average height of approximately 10–15 cm above soil surface, and dried at 70 °C until constant dry weight, before being weighed to a precision of 0.01 g (KERN PCB 1000-2 precision scale) or to a precision of 1 g (digital scale, Weis, model 10531 ECO pro). Going forward, all values were rounded to the nearest gram. No differentiation was made between *T. latifolia* and *T. angustifolia* in these samplings, as *T. latifolia* quickly became dominant after planting and seeding, and a reliable distinction between the two species was not possible by the harvesting staff.

Porewater nutrient content

We analysed the porewater nutrient content using rhizons (Rhizon MOM soil moisture sampler with 10 cm porous part, reinforced with glass fibre wire; Rhizosphere Research Products, Wageningen, The Netherlands), at two locations: close to the inlet in the south-east and close to the potential outlet in the north-west of the pilot site. On 15 July 2020, at each location 20 rhizons were installed approximately 1 m apart from each other in two rows. The upper end of the porous part was 5–10 cm deep in the soil (in the root horizon). We sampled on seven dates (17 July 2020, 6 October 2020, 27 November 2020, 14 April 2021, 13 July 2021, 13 October 2021, 5 April 2022), taking mixed samples of all rhizons of one location and analysed pH (DIN EN ISO 10523 (04/2012)), ortho-phosphate (P, DIN EN ISO 15681-1 (05/2005)), total nitrogen (N, DIN EN ISO 11905-1 (08/1998)), ammonium-N (NH_4-N, DIN EN ISO 11732 (05/2005)), nitrate-N (NO_3-N, DIN EN ISO 10304-1 (07/2009)), calcium (Ca), magnesium (Mg) and potassium (K) (all DIN EN ISO 11885 (09/2009)) and iron (Fe, DIN EN ISO 17294-2 (01/2017)). O_2-concentration (Fibox 4, PreSens Precision Sensing GmbH, Regensburg, Germany) and electric conductivity (Conductivity meter LF 318 with Standard-conductivity cell TetraCon 925, Wissenschaftlich-Technische Werkstätten GmbH, Weilheim, Germany) were measured in each sample directly after sampling in the field.

The nutrient content in the porewater at the inlet versus potential outlet of the pilot site was assessed with a paired-samples t-test in R (R Core Team 2024).

Surface water nutrient content

We examined the nutrient concentrations in the surface water at two locations of the pilot field: at the inlet in the south-east and at the potential outlet in the north-west. The sampling started in September 2019 and took place once a month until August 2022. The water sample was taken from the surface of the open ditch ending in a small pond at the closed potential outlet. In case of biomass litter or floating aquatic plant species, we moved them carefully aside or changed the sampling location to an uncovered spot.

Temperature, O_2-concentration (Fibox 4, PreSens Precision Sensing GmbH, Regensburg, Germany) and electric conductivity (Conductivity meter LF 318 with Standard-conductivity cell TetraCon 925, Wissenschaftlich-Technische Werkstätten GmbH, Weilheim, Germany) were measured in each sample directly after sampling in the field. Parts of each sample were chemically fixed with H_2SO_4 for NH_4-N and HNO_3 for Fe in separate sampling tubes. Subsequently, the samples were transported in an insulated box to the lab for further analyses. For surface water nutrient content, the same variables as for porewater nutrient content were analysed.

The nutrient content in the surface water at the inlet and the potential outlet in the north-west of the pilot site was compared with a paired-samples t-test in R (R Core Team 2024).

Based on the mean water nutrient concentrations of the Peene river per vegetation period and year and the estimated pumping volumes, we calculated the nutrient inputs per year and hectare. Pumping volumes could only be estimated because water meters were installed in June 2021 for the generator-driven water pump and in September 2021 for the solar water pump. For 2020, we estimated the pumping volumes from the changes in water levels and the estimated displacement in the water level fluctuation range. For 2021, estimates for the generator-driven water pump were based on average pumping rates and the operating hours of the water meter. Estimates for the solar water pump were based on a regression (power function of the form a (sunshine-hours - 1)^b for sunshine-hours ≥ 1 and with $0 < b < 1$) of the daily pumping rates on the sunshine-hours of the nearby weather station (Deutscher Wetter Dienst 2023) for 2021 and for 2022 based on a linear regression of the water meter values minus the value at the beginning of the season over time. Data on nutrient concentrations in the Peene river for the vegetation period of each year (April to September) when potential pumping events took place were provided by the state agency of agriculture and environment of the Mecklenburg Lake Plateau (Staatliches Amt für Landwirtschaft und Umwelt Mecklenburgische Seenplatte (StALU MS) 2024).

$$nutrient\ input\ [kg\ ha^{-1}y^{-1}] = \frac{mean\ nutrient\ concentration\ [mgL^{-1}] * pumping\ volume\ per\ year\ [m^3\ y^{-1}] * 10^{-3}}{8.5\ ha}$$

To account for the standard errors of both variables, we applied the standard error propagation formula for the product of two independent variables. Specifically, we calculated the square root of the sum of the squared relative errors, and multiplied the result by the product.

Soil analysis

Soil characteristics of the pilot field were assessed on 16 July 2019 (before site construction), on 17 September 2019, after site construction, but before rewetting, and on 8 September 2020 (one year after site construction/rewetting). Eight disturbed soil samples were taken from the upper 10–15 cm of the soil layer (samples before site construction were taken underneath the sod for comparability), along two transects from

north to south (four samples/transect). The samples were taken at the same fixed points, whereas the accuracy of the GPS was approximately 10–15 m. The following soil properties were analysed in an external lab: fresh and dry matter bulk density (BGK II / A 4), dry residue after drying (DIN EN 14346 (03/2007)), organic matter content (DIN EN 15935 (11/2012)), pH-value (in $CaCl_2$; DIN EN 15933 (11/2012)), electric conductivity (EC, DIN EN 15933 / DIN EN 27888), effective cation exchange capacity (ECEC) (DIN ISO 11260 (11/2018)), total C (DIN EN 13137 (12/2001)), total-N (DIN ISO 11261 (05/1997)), P, K, Mg (DIN EN ISO 11885 (09/2009)), iron (Fe) and aluminium (Al) (DIN EN ISO 17294-2 (01/2017)). Additionally, for September 2019, ortho-phosphate (P) (DIN EN ISO 15681-1 (05/2005)), ammonium-N (DIN EN ISO 11732 (05/2005)/FIA), nitrate-N (DIN EN ISO 10304-1 (07/2009)) were analysed.

We tested for changes in the soil characteristics induced by rewetting with linear mixed models using *lmerTest* package in R with the transects as random effect (Kuznetsova *et al.* 2017, R Core Team 2024). Changes in soil elevation of the sampling plots from 2020 to 2022 (data available for all 80 plots in 2022 and for 40 plots in 2020) were tested with a paired t-test.

Remote sensing

The study site was regularly monitored with multispectral image data from uncrewed aerial vehicles (UAVs). Monitoring started in July 2020 and included multiple observations per year (Table 1). Image data were acquired with a DJI Phantom 4 Multispectral drone encompassing five spectral bands: blue (450 nm), green (560 nm), red (650 nm), red edge (730 nm) and near infrared (840 nm). The ground resolution differed depending on the flight altitude (Table 1). In order to use the imageries, they were combined to image mosaics and digital surface models (DSM) by structure-from-motion in Agisoft Metashape Professional (v1.8.4) and to derive the normalised difference vegetation index (NDVI; Tucker 1979).

For each observation year, classifications were undertaken to map species distribution. For comparability, a late summer image mosaic of excellent data quality was selected for each year of observation, i.e., 17 September 2020, 7 September 2021, and 31 August 2022. To include textural information into subsequent analyses, the original pixels of 1.96–2.64 cm ground resolution (see Table 1) were aggregated into 0.5 m x 0.5 m raster cells and the median, 25 %- and 75 %-percentile values calculated for each spectral band, the NDVI and DSM. The resulting 21 layers for each year were stacked into three annual data sets.

Following data pre-processing, we conducted supervised classifications to map the occurrence of *Typha* spp., *Phragmites australis*, *Phalaris arundinacea*, *Juncus* spp., as well as *Water* and *Algae* for the three years 2020, 2021 and 2022. An additional class *Other Wetland Vegetation* was required, especially in 2020, to describe areas where none of the target species was dominating. Representative species for this class were, e.g., *Carex* spp., *Alisma plantago-aquatica*, *Deschampsia cespitosa*, *Agrostis stolonifera*, and others. We used a Random Forest classifier (Breiman 2001) that we trained with approximately 20–40 reference pixels per class and year. Reference pixels were selected based on observations in the field. First tests showed that a higher number of reference pixels was needed for *Phalaris*, i.e., up to 81. 80 % of the reference data was used for training and 20 % were held out for validation.

Independent from the present study, the UAV data from three dates in 2021 was used by Hellmann *et al.* (2024) to model *Typha* biomass. They created a *Typha* map from the multispectral information and derived DSMs from structure-from-motion, which are in combination used to predict *Typha* biomass. The good quality of their results underline the quality of the UAV data also used in the present study, e.g., to delineate vegetation types.

Table 1. Overview of UAV data acquisitions at the pilot site collected with a DJI Phantom 4 Multispectral drone. Data quality, as a result of inhomogeneous cloud cover, is indicated for each entry. Acquisition dates used in further analysis of this paper are highlighted in bold.

Acquisition date	**Flight altitude [m above ground]**	**Ground resolution [cm]**	**Data quality** (1-excellent; 2/3-mostly/partly usable; 4-insufficient)
14 Jul 2020	90 and 50	4.76 and 2.64	4
17 Sep 2020	37	1.96	1
13 Oct 2020	37	1.96	2
28 Oct 2020	37	1.96	2
19 May 2021	50	2.64	4
15 Jun 2021	50	2.64	1
05 Jul 2021	50	2.64	4
03 Aug 2021	50	2.64	2
07 Sep 2021	50	2.64	1
06 Oct 2021	50	2.64	3
16 Nov 2021	50	2.64	3
10 Mar 2022	50	2.64	1
29 April 2022	50	2.64	1
16 Jun 2022	50	2.64	2
27 Jul 2022	50	2.64	1
31 Aug 2022	50	2.64	1
20 Oct 2022	50	2.64	3
23 May 2023	50	2.64	2
07 Jul 2023	50	2.64	3
25 Jun 2024	50	2.64	1

RESULTS

Development of the *Typha* biomass

In July 2021, total yield of the aboveground *Typha* biomass was 1.13 t ha^{-1} on average for the entire pilot site. Over December 2021 and July 2022, total productivity of the aboveground *Typha* biomass increased up to a mean dry matter of 5.00 ± 3.51 t ha^{-1} for the entire pilot site in December 2022. Considering each density category separately, density category "dense *Typha*" increased from a mean of 2.69 ± 2.46 t ha^{-1} in July 2021 to 7.43 ± 1.88 t ha^{-1} in December 2022, density category "mixture *Typha* + *Phalaris*" increased in mean dry *Typha* biomass from 1.37 ± 1.42 t ha^{-1} in July 2021 to 4.62 ± 4.02 t ha^{-1} in December 2022 (Figure 2). Density category "sparse T*ypha*" developed from 0.42 ± 0.29 t ha^{-1} on average in July 2021 to 5.17 ± 2.23 t ha^{-1} in December 2022. Density category "rare *Typha*" increased from 0.04 ± 0.10 t ha^{-1} in July 2021 to 0.74 ± 0.74 t ha^{-1} in December 2022. Density categories "dense *Typha*" and "mixture *Typha* + *Phalaris*" were partially mown in December 2021. Yet, mowing did not alter the aboveground biomass production significantly in these density categories (6.25 ± 2.59 t ha^{-1} in July 2022 and 7.44 ± 2.30 t ha^{-1} in December 2022 for mown plots in "dense *Typha*" and 3.28 ± 3.84 t ha^{-1} in July 2022 and 3.88 ± 3.79 t ha^{-1} in December 2022 for "mixture *Typha* + *Phalaris*").

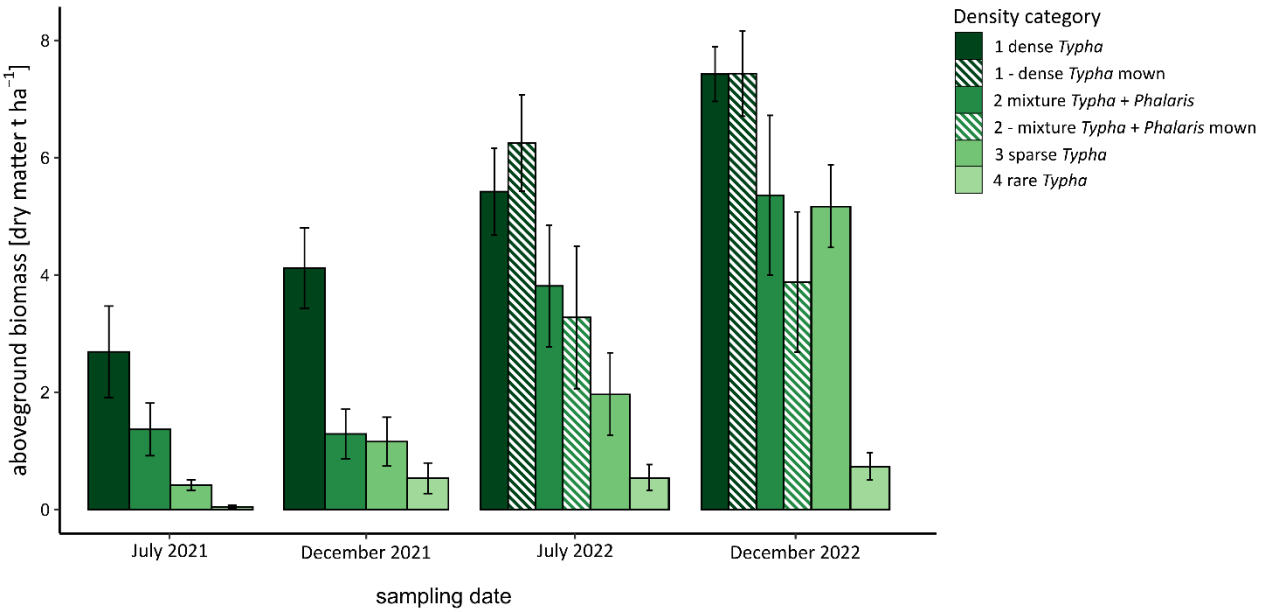


Figure 2. Development of *Typha* standing aboveground biomass [dry matter t ha^{-1}] from 2021 to 2022. Plots that were mown within the mechanical trial harvest in December 2021 were considered separately in 2022. Mean values and standard errors of the mean of ten plots (1 m²) per category (n = 40 in 2021 and n = 60 in 2022) are shown.

Water levels in the pilot site

The average water level was +24 ± 5 cm above soil surface in the sampling period in July 2021 (5–13 July) and +13 ± 6 cm in the sampling period in July 2022 (2–7 July). The mean long-term water level from 10 July 2020 to 13 July 2021 was +10 ± 5 cm, whereas it was +18 ± 6 cm from 13 July 2021 to 7 July 2022. Estimated pumping volumes were 9412 ± 1176 m^3 ha^{-1} in 2020, 7300 ± 600 m^3 ha^{-1} in 2021 and 8600 ± 600 m^3 ha^{-1} in 2022. Water level fluctuated from 18 June 2020 (first day of water level measurements) to 9 July 2020 within the 10 %- and 90 %-percentiles from –14 cm to +16 cm. From 10 July 2020 to 13 July 2021, water level fluctuations ranged between +11 cm and +27 cm, whereas the water level fluctuated between –4 cm to +34 cm 13 July 2021 to 7 July 2022 in the same percentiles.

Drivers for development of the *Typha* stand

Numbers of *Typha* shoots per plot were highly correlated to the observed environmental variables (BRT cross-validated correlation = 0.77, Figure 3). The strongest relation was found with cover of accompanying species (32 % of the explained variation). The number of *Typha* shoots decreased rapidly at a coverage of approximately > 35 % of accompanying species. Maximum number of *Typha* shoots was reached at approximately 30 % cover of accompanying species. Proportion of *T. angustifolia* in 2024 accounted for 28 % of the explained variance in the number of *Typha* shoots. In July 2023, 36 out of 80 sampling plots contained at least one *T. angustifolia* plant and in 2024, 46 of 80 plots contained *T. angustifolia.* A high proportion of *T. angustifolia* (> 60 %) correlated with a lower number of total *Typha* per plot. In contrast, a low proportion of *T. angustifolia* (approximately 10–15 %) correlated with a low proportion of total *Typha*, whereas a medium share (20–60 %) of *T. angustifolia* positively affected the total number of *Typha* shoots per plot. Annual mean water level accounted for 24 % of the explained variation in *Typha* shoot number. Water levels higher than +10 cm positively influenced the *Typha* stand, whereas water levels between 0 and +10 cm were negatively correlated with the number of *Typha* shoots.

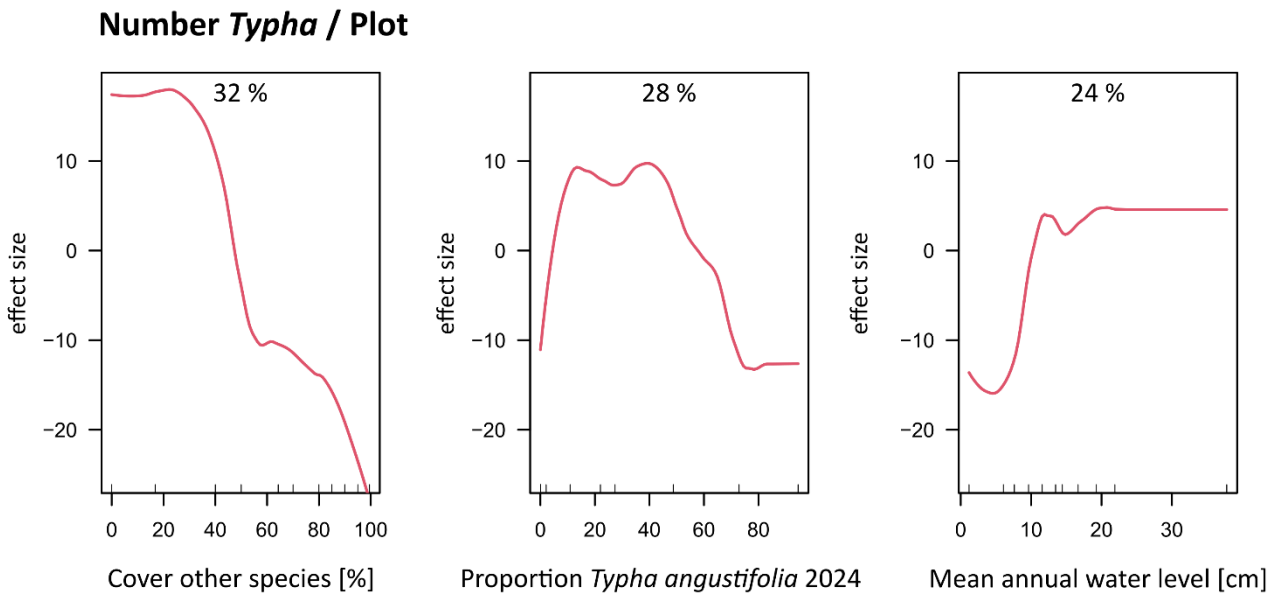


Figure 3. *Typha* stand development, expressed as number of *Typha* individuals per plot shown in relation to the explanatory variables as indicated by an BRT analysis. Positive effect sizes indicate more *Typha* shoots per plot than the average, negative effect sizes indicate shoot numbers below the average. Cross-validation correlation was 0.77. Upward ticks along the x-axis indicate data distribution in steps of 10 %. Only variables accounting for > 10 % of the variance in the response variable explained by the model are depicted, relations with other variables are shown in Figure A7.

Drivers of the species composition

The accompanying species composition of the paludiculture site changed only marginally from the second to the third year of the study (ANOSIM R = 0.05; Figure 4 d). It was mainly linked to water level (correlation values NMDS1 –0.87, NMDS2 –0.48, NMDS3 –0.08, $p < 0.001$). An NMDS depicting the species names can be found in the attachment (Figure A8).

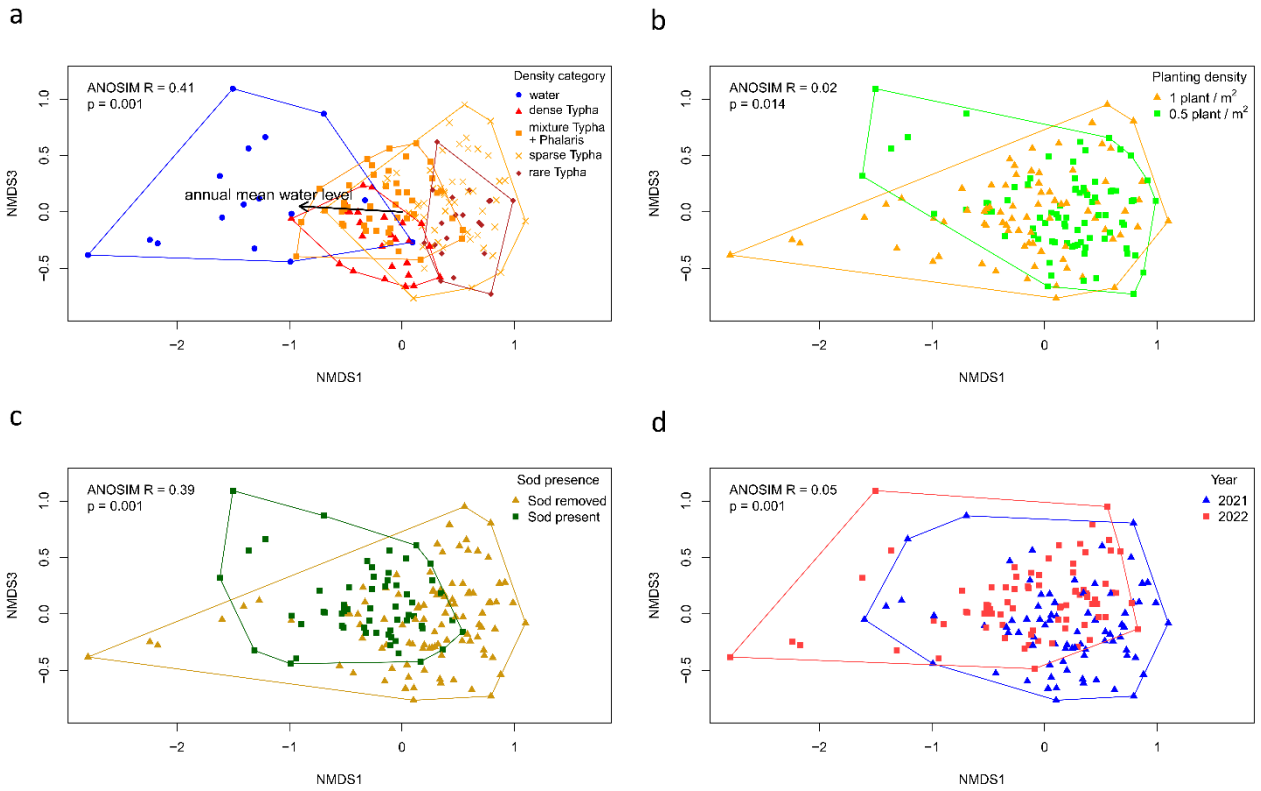


Figure 4. Non-metric Multidimensional Scaling (NMDS) depicting the species composition of the 80 sampling plots in 2021 and 2022. a) Plots classified by density category (ANOSIM R = 0.41, p = 0.001), including mean annual water level as significant environmental vector (p = 0.001); b) plots sorted by planting density (ANOSIM R = 0.02, p = 0.014); c) plots sorted by sod presence (ANOSIM R = 0.39, p = 0.001); d) plots sorted by year (ANOSIM R = 0.05, p = 0.001). For all plots, axes 1 and 3 of the three-dimensional NMDS are depicted. P-values in the figures are ANOSIM values for the respective explanatory variable.

Accompanying vegetation – water level

The accompanying vegetation was more abundant in plots with lower water levels (Figure 5). Yet, there were slight differences between the study years. In year 2020, accompanying species had relatively high cover (50 %) up to a water level of +15 cm and cover of accompanying species did not change significantly with mean water level ($p = 0.24$). In 2021, cover values of the accompanying species decreased significantly with increasing water level ($p < 0.01$). A strong decrease in cover of accompanying species occurred at water levels higher than approximately +15 cm. Moreover, the maximum cover values of the accompanying vegetation were higher in 2021 (approximately 60 %) than in 2020. In 2022, cover of accompanying vegetation was much higher than in the previous years with some cover values reaching 90–100 %. Yet, there was a strong, significant decline of the cover of accompanying species with increasing water level ($p < 0.001$). Already at a water level of approximately +5 to +10 cm, the cover of accompanying species decreased, and there was only a low cover of accompanying species at +20 cm.

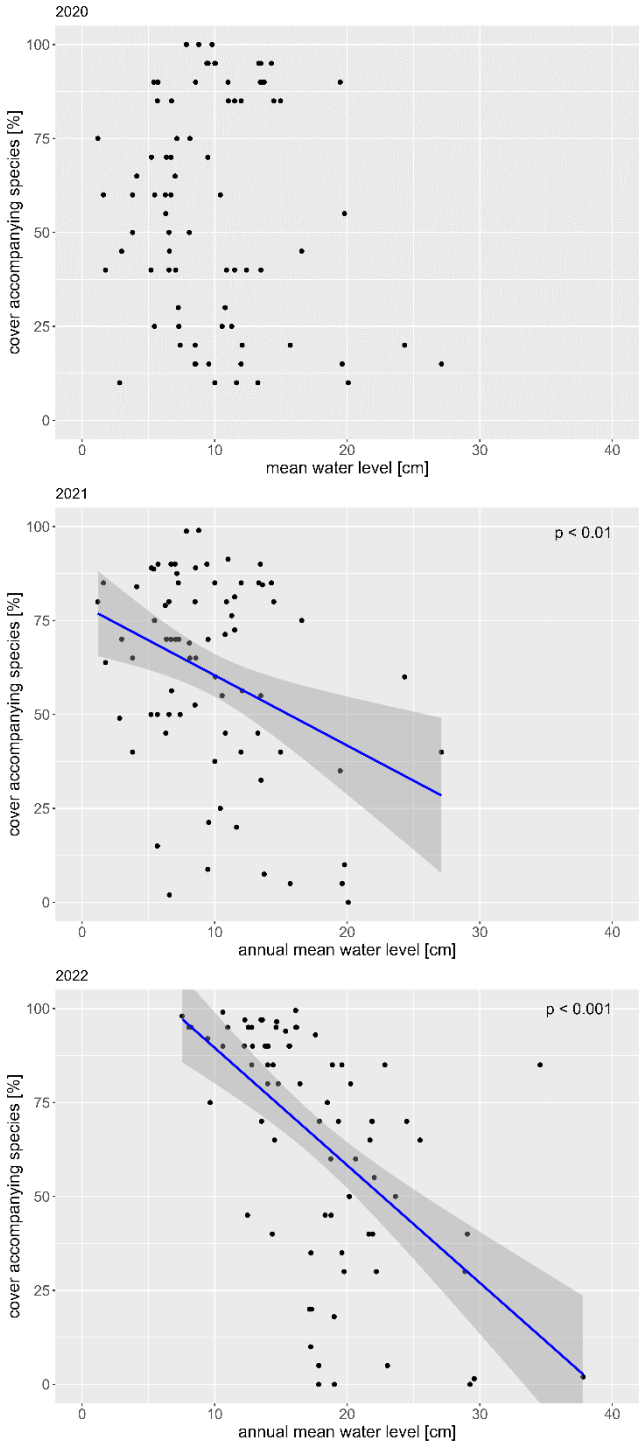
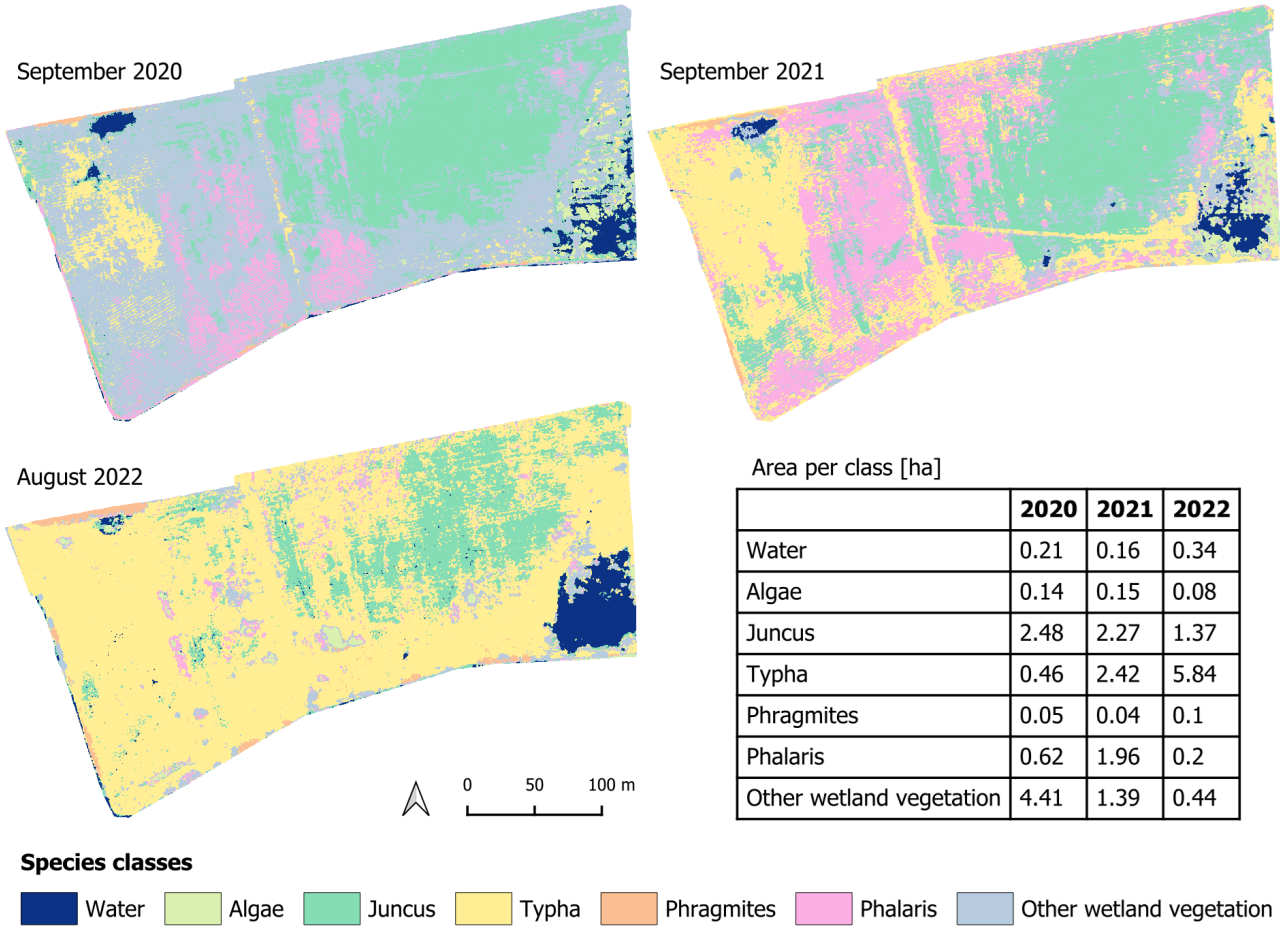


Figure 5. Relationship between annual mean water level and the cover of the accompanying species over the years 2020–2022. A linear regression model (blue line) with a 95% confidence interval (grey) was fitted to the data and is depicted for $p < 0.05$. Black dots indicate original values of accompanying species cover. Water level values are relative to the soil surface, with positive values indicating inundation. As water level monitoring started in June 2020, the 2020 values are based on measurements from 18 June to 9 July 2020.

Vegetation development assessed by remote sensing

Overall accuracies and class-wise F1 scores based on the UAV images were constantly very high and above 0.9 for all mapped classes. The classification for 2020, 2021 and 2022 generally highlighted the patterns of dominant vegetation observed in the field (Figure 6). In 2020, an area of 4.41 ha was classified as *Other Wetland Vegetation* indicating that none of the targeted species/genera was dominating within the aggregated 50-cm pixels and instead a mix including less abundant species was modelled. Although individual shoots were found in most of the area, *Typha* dominated only relatively small areas in the west and along the former central ditch which was filled during site preparation (0.46 ha in total). *Juncus* on the other hand dominated central to north-eastern areas and *Phalaris* dominated in the south and as patches within *Juncus* dominated parts (2.48 ha and 0.62 ha, respectively). *Phragmites* covered only small patches along the surrounding ditches. Water and Algae were found in the areas with lower elevation in the south-east and north-west. The situation changed considerably for 2021, when the areas without dominating species (i.e., *Other Wetland Vegetation* in 2020) decreased substantially to 1.39 ha and larger and more continuous patches of *Typha*, *Juncus* and *Phalaris* developed in the aforementioned parts of the study site. The other classes remained similar. In 2022, *Typha* dominated most of the area (5.84 ha). *Phalaris* was only detected on very small patches in the central parts (0.2 ha in total) and *Juncus* remained as dominating vegetation in the central to eastern parts (1.37 ha in total), but much sparser and together with *Typha*.



Area per class [ha]

	2020	2021	2022
Water	0.21	0.16	0.34
Algae	0.14	0.15	0.08
Juncus	2.48	2.27	1.37
Typha	0.46	2.42	5.84
Phragmites	0.05	0.04	0.1
Phalaris	0.62	1.96	0.2
Other wetland vegetation	4.41	1.39	0.44

Figure 6. UAS-based classification of the vegetation for 2020, 2021 and 2022.

Porewater nutrient content

The porewater nutrient analysis revealed significant differences between the inlet and the potential outlet in the north-west of the pilot site for NH_4 ($p = 0.01$), Ca ($p = 0.00$), EC ($p = 0.0$) and a minor difference for Fe ($p = 0.04$, Figure 7). NH_4 concentrations were higher at the inlet (mean = 5.11 ± 2.98 mg L^{-1}) than close to the potential outlet (mean = 1.07 ± 1.26 mg L^{-1}) throughout the entire study period. Ca concentrations were higher at the potential outlet for each sampling date. Similarly, electric conductivity was always higher at the outlet. Fe concentrations tended to be higher at the outlet, yet, there was an interesting pattern revealing a strong decrease in Fe concentrations for in- as well as potential outlet in April 2021. Ortho-P content did not alter

over time or differ between the sampling locations. Yet, there was an interesting pattern for ortho-P over time, as it increased at the inlet from 0.03 mg L^{-1} in November 2020 up to 8.00 mg L^{-1} in July 2021, then decreased again rapidly to 0.40 mg L^{-1} in October of the same year.

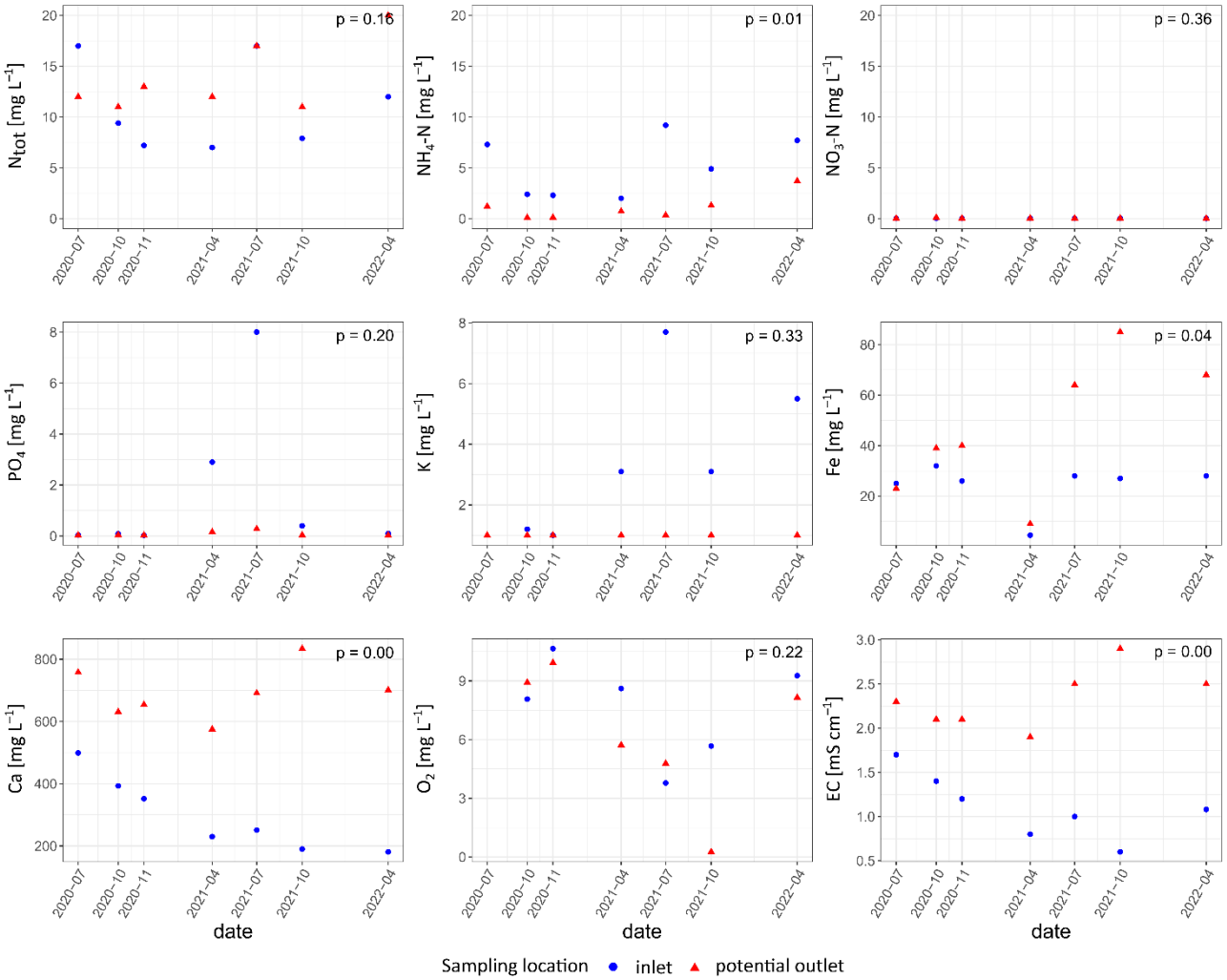


Figure 7. Porewater nutrients close to the inlet in the south-east (blue) and close to the potential outlet in the north-west (red) of the pilot site between 2020 and 2022. Note that the outlet was closed for the majority of the time to achieve high water tables, so that no outflow of water happened and the sampling location 'potential outlet' rather indicates the conditions furthest away from the inlet.

Surface water and soil analysis at inlet and potential outlet

K concentration in the surface water differed significantly between the inlet in the south-east and the potential outlet of the pilot site in the north-west ($p < 0.001$). While the water showed higher K contents at the inlet, K also tended to increase over time at both in- and potential outlet. Mg concentrations were slightly ($p = 0.05$) higher at the potential outlet than at the inlet. No significant differences between inlet and potential outlet were detected for the other nutrients (Figure 8).

Average values for the growing season (April–September) in the Peene river for N were 3.27 ± 0.62 mg L^{-1} in 2020, 1.67 ± 0.18 mg L^{-1} in 2021 and 2.7 ± 0.43 mg L^{-1} in 2022, respectively. For P, annual average values for 2020 were 0.21 ± 0.06 mg L^{-1}, 0.26 ± 0.13 mg L^{-1} in 2021 and 0.15 ± 0.05 mg L^{-1} in 2022, respectively. With the pumping volumes we calculated the nutrient input from the Peene river (Table 2).

Rewetting had a significant effect on the soil characteristics bulk density ($p < 0.01$), electric conductivity ($p < 0.01$) and Fe content ($p < 0.01$). There was a slightly significant effect on dry residue ($p < 0.05$, Table 3). The bulk density (fresh mass) increased by ~24 % from before the site construction and rewetting to one year after ($p < 0.01$). The electric conductivity increased by 92 % ($p < 0.01$), whereas Fe increased strongly from 0.001 % dry matter in July 2019 to 1.03 % dry matter in September 2020 ($p < 0.01$). Dry residue decreased

slightly from 58.4 ± 9.3 % in July 2019 to 46.9 ± 10.7 % in September 2020 ($p = 0.03$). No effect was detected on any of the other soil variables. Soil elevation of the sampling plots increased from 2020 to 2022 significantly by 10.00 ± 0.43 cm ($p < 0.01$, Figure A9).

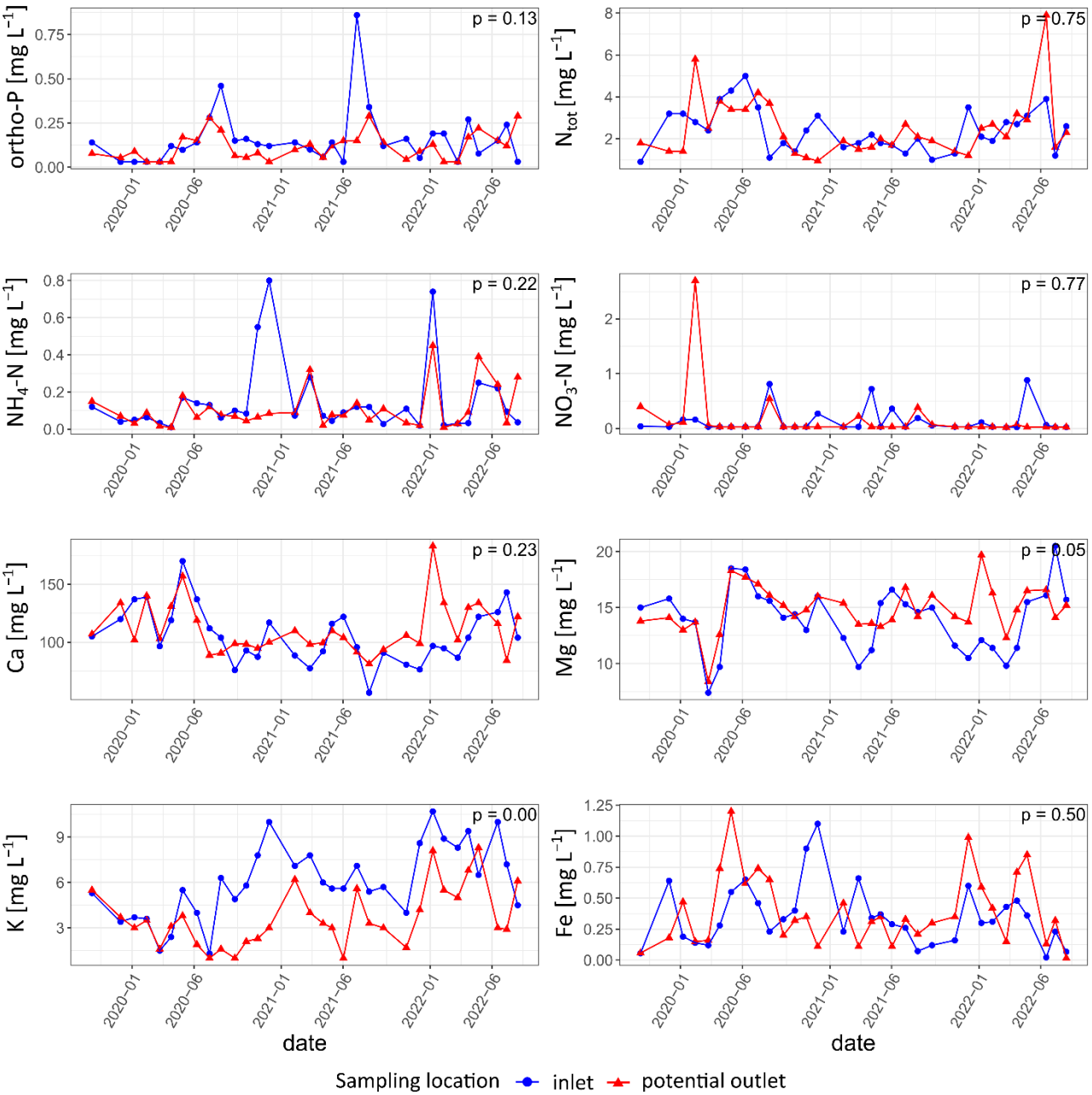


Figure 8. Nutrients in the surface water close to the inlet in the south-east (blue) and potential outlet in the north-west (red) of the pilot site (both locations inside the site). Note that the outlet was closed for the majority of the time to achieve high water tables, so that no outflow of water happened at the potential outlet and the sampling location 'potential outlet' rather indicates the conditions furthest away from the inlet.

Table 2. Nutrient input per hectare and year based on mean nutrient concentrations during the vegetation periods (April–September) in the Peene river and pumping volumes for the years 2020–2022.

Year	NH_4-N [kg ha^{-1} y^{-1}]	NO_3-N [kg ha^{-1} y^{-1}]	NO_2-N [kg ha^{-1} y^{-1}]	Total-N [kg ha^{-1} y^{-1}]	PO_4-P [kg ha^{-1} y^{-1}]
2020	2.51 ± 1.60	4.41 ± 1.76	0.32 ± 0.16	14.39 ± 3.72	1.21 ± 0.44
2021	0.49 ± 0.21	3.01 ± 1.29	0.11 ± 0.02	10.88 ± 2.66	1.19 ± 0.39
2022	0.78 ± 0.21	5.66 ± 3.40	0.16 ± 0.07	14.21 ± 5.53	0.85 ± 0.49

Table 3. Soil characteristics for July 2019 (16 July, before sod removal), September 2019 (17 September, after sod removal) and September 2020 (8 September, one year after sod removal and rewetting). All data are average values of 8 samples, including standard deviation (± SD). Soil sampling was conducted at a depth of 0–20 cm. Dry residue resulted from dry bulk density divided by (fresh) bulk density. Adjusted intraclass correlation coefficients (ICC) from mixed-effects models indicate the variance attributed to the random effect (transect). P-values refer to the fixed effect 'sampling date'.

Parameters	**before site construction**	**directly after site construction**	**1 year after rewetting**	**adjusted ICC**	**p-value**
bulk density [g L^{-1} fresh mass]	1012.5 ± 76.1	1083.8 ± 147.8	1253.8 ± 113.6	0.17	**< 0.01**
dry bulk density [g L^{-1} dry matter]	597.5 ± 132.5	537.5 ± 156.5	598.8 ± 179.2	0.41	0.58
dry residue [%]	58.4 ± 9.3	49.4 ± 10.5	46.9 ± 10.7	0.45	**0.03**
loss of ignition [% dry matter]	25.2 ± 10.3	29.9 ± 16.1	31.7 ± 18.6	0.54	0.59
pH-value (in $CaCl_2$ suspension)	7.2 ± 0.4	7.4 ± 0.3	7.4 ± 0.2	0.27	0.26
electric conductivity [µS cm^{-1}]	137.9 ± 30.6	192.6 ± 50.6	264.6 ± 59.8	0.30	**< 0.01**
cation-exchange capacity [cmol+ kg^{-1}]	51.9 ± 29.7	71.5 ± 44.1	80.9 ± 49.2	0.52	0.22
total C [% dry matter]	14.3 ± 4.5	19.1 ± 9.6	16.8 ± 9.2	0.12	0.49
total N [% dry matter]	1.13 ± 0.5	1.3 ± 0.7	1.1 ± 0.63	0.54	0.66
P [% dry matter]	0.1 ± 0.0	0.1 ± 0.0	0.1 ± 0.0	0.54	0.60
K [% dry matter]	0.1 ± 0.0	0.1 ± 0.0	0.1 ± 0.0	0.65	0.42
Mg [% dry matter]	0.1 ± 0.0	0.1 ± 0.0	0.1 ± 0.0	0.39	0.08
Fe [% dry matter]	0.0 ± 0.0	0.0 ± 0.0	1.0 ± 0.4	0.68	**< 0.01**
Al [% dry matter]	0.4 ± 0.1	0.3 ± 0.1	0.3 ± 0.1	0.72	0.38

DISCUSSION

Development of *Typha* stand and biomass

Typha biomass productivity on our pilot site increased steadily during the first three years after site establishment. Total *Typha* aboveground biomass continued to increase from July to December in both years (from 1.13 ± 1.72 t ha^{-1} in July 2021 to 3.55 ± 3.22 t ha^{-1} in December 2021 and from 1.78 ± 2.00 t ha^{-1} in July 2022 to 5.00 ± 3.51 t ha^{-1} in December 2022). In December 2022, three years after site establishment and planting, a dry biomass yield of 5.00 ± 3.51 t ha^{-1} over the whole site and of 7.43 ± 1.88 t ha^{-1} for the dense *Typha* parts of the site was still relatively low in comparison to the potential productivity of *Typha* stands.

Comparable to our results, dry matter yields of up to 8.83 t ha^{-1} y^{-1} were reported for *T. angustifolia* at a spacing of 0.50 x 0.32 m on a 1-ha site three years after site establishment (Eickenscheidt *et al.* 2023b). Pijlman *et al.* (2019) report dry matter yields of 9.81–10.89 t ha^{-1} from single harvests of *T. latifolia* already in the second year after site establishment on a trial site of 60 m² (consisting of several plots of either 0.5 m x 0.5 m with a planting density of 15 plants m^{-2} or 1.6 m x 1.6 m plots with a planting density of 3.5 plants m^{-2}). Across rewetted but otherwise unmanaged sites in Europe, *T. latifolia* produced dry matter yields of 12.4 ± 6.6 t ha^{-1}

y^{-1} (Geurts *et al.* 2020). Dry matter yields of 20.15 t ha^{-1} were reported by Pfadenhauer and Wild (2001) for an experimental site of 2 ha three years after site establishment. Dubbe *et al.* (1988) even report *Typha* productivity of up to 21.1 t ha^{-1} (*T. angustifolia*) for natural stands in Minnesota. Our pilot site features a heterogeneity that is not present in smaller study sites, as it is many times larger. This heterogeneity, which is depicted by the four density categories identified for our pilot site, probably decreased the total productivity, as areas with a sparse *Typha* stand also account for the total dry biomass. The heterogeneous biomass distribution and hence relatively low average productivity is confirmed by the UAS-based study by Hellmann *et al.* (2024), who used the UAS data described above also for biomass estimates and found a total biomass of 1.16 t ha^{-1} in 2021. Furthermore, stand age is a factor that can strongly influence *Typha* stand productivity (Pfadenhauer & Wild 2001, Geurts *et al.* 2020: stands younger than three years were found to produce two to three times less dry biomass (6.1 t ha^{-1} per year, similar to the value on our site in 2022) than stands older than five years (14.1 t ha^{-1} on average)). So, while the yield on our site in 2022 does not reach possible maximum yields reported in literature, the already strong increase in yield observed between 2021 and 2022 is a promising indication that higher yields are possible as the stand matures. Although *Typha* is highly productive, its long-term biomass production depends on sustained nutrient availability; high nutrient uptake in dense *Typha* stands, together with evidence that prolonged summer biomass removal can reduce nutrient pools, suggests that nutrient limitation may eventually constrain *Typha* growth and dominance (Bansal *et al.* 2019, Geurts *et al.* 2020). Three years after establishment, *Typha* biomass on our pilot site was still increasing, whereas the accompanying species composition had remained constant from 2021 to 2022 (Figure 4d). In our pilot site, irrigation water stems from the Peene river, whose largely agricultural catchment area has a consistent, low nutrient load, making complete nutrient depletion unlikely in the near future. This highlights the need for further research to examine whether such nutrient-driven declines will also emerge in large, managed stands over longer timescales.

We hypothesised that the *Typha* stand increased in shoot number over time with increasing water level above soil surface. Given the exploratory nature of our statistical analyses, the relationships reported below should be interpreted as patterns rather than causal effects. Yet, the BRT analysis revealed that the cover by other species was the explanatory variable which was most strongly related to *Typha* shoot numbers, contributing to 32 % of the explained variance, whereas mean annual water level reached only 24 %. Thus, hypothesis 1 was only partially supported, as water level was an important but not the strongest predictor of *Typha* shoot number. We furthermore detected a strong correlation between the proportion of *T. angustifolia* to *T. latifolia* (measured only after the fact in 2024) in our study plots with the total number of *Typha* shoots (Figure 3, 28 % of the explained variance according to BRT). The number of total *Typha* shoots decreased with increasing proportion of *T. angustifolia*, indicating that *T. angustifolia* tended to occur in parts of our pilot site where the stand was rather sparse. In contrast, there appeared to be fewer *Typha* plants overall when the proportion of *T. angustifolia* was low. Indeed, *T. angustifolia* is known to grow in denser stands than *T. latifolia* (Heinz 2012). Thus, we hypothesise that *T. angustifolia* fills up the *Typha* stand if there is up to 50 % *T. latifolia* present, causing an overall high number of *Typha* shoots. Yet, if there is only a small number of *Typha* shoots at all, probably due to other reasons like competition with accompanying vegetation or water levels that are not suitable, stands consist mainly of *T. angustifolia* and are sparse.

Moreover, our *Typha* stand seemed to benefit from annual mean water levels exceeding +10 cm. This is in line with hypothesis 1, which predicted a positive relationship between water level and *Typha* shoot number (Figure 3). Haldan *et al.* (2022) found in a mesocosm experiment that *Typha* shoot number was not affected by water levels (gradient between –40 cm to +40 cm) for *T. latifolia*, whereas *T. angustifolia* produced less shoots at waterlogged conditions. Yet, these experiments were conducted in mesocosms with young plants in their first year of growth. Heinz (2012) pointed out that the water level requirements of *Typha* change between germination, recruitment and establishment phases. While *Typha* seedlings benefit from moderately flooded conditions, water level fluctuations with prolonged inundation even enhance the dominance of *Typha* x *glauca* plants in wetlands in North America (Boers *et al.* 2007, Boers & Zedler 2008, Heinz 2012). In near-natural stands, both species have a greater tolerance of higher water levels of at least up to +50 cm (Grace & Wetzel 1981, 1982).

Besides fluctuating water levels, microrelief and a heterogeneous elevation resulted in higher and lower relative water levels which is reflected by the species composition across the site and even by the definition of the density categories (Figure 1). There is a dense *Typha* stand in the north-western part of the pilot site with slightly lower soil elevation, which provided more suitable germination conditions for *Typha* seeds in the first growing season. This dense *Typha* stand continued to extend until 2022, as can be seen in the UAS maps (Figure 6).

Phragmites australis is known to cope well with water levels > 35 cm in established, natural stands (in fact, *P. australis* has a hydrological amplitude of –6 to 2.3 m; Packer *et al.* 2017). Thus, it might be that *Typha* is outcompeted by *P. australis* in the long term at higher water levels as well as at lower water levels where *Typha* seed germination and thus the establishment of new stands may be less favourable. This indicates that stable water levels between 0 and +15 cm on average are to be preferred for a *Typha* paludiculture stand, as *Typha* benefits from water levels slightly above soil surface and competition with *P. australis* may be less pronounced than at higher or more fluctuating water levels. Boers and Zedler (2008) found for *Typha* x *glauca*, that sustained inundation enhanced the spreading vegetatively. Similarly, experiments with *T. domingensis* revealed that fluctuating water levels hindered the plants from spreading. *Typha* is more productive at waterlogged soil conditions (Li *et al.* 2004, Geurts & Fritz 2018). Furthermore, *T. latifolia* is known to outcompete *T. angustifolia* at water levels below 50 cm (Grace & Wetzel 1981, 1982). On our pilot site, *T. latifolia* is the dominant species, even though both *Typha* species were planted in the same numbers (Figure A3; Neubert *et al.* 2022). This might be due to mean water levels on the site of about +10 to +20 cm being in favour of *T. latifolia* rather than *T. angustifolia*.

The cover of accompanying species correlated negatively with the number of *Typha* shoots per plot (32 % of the explained variance in the BRT model). Yet, we cannot infer causality with the data at hand. It might be that more accompanying vegetation in a plot causes the *Typha* number to decrease, but it is also possible that *Typha* outcompetes the accompanying species. Both *Typha* shoot numbers and cover of the accompanying vegetation might furthermore be driven by a third variable, e.g., water level. As seen in Figure 5, the accompanying vegetation decreased with higher water levels, whereas *Typha* benefitted from higher water levels (Figure 3). However, this might not be fully representative, since the water levels did not exceed +30 cm at the top end of the water level range and 0 cm at the lower end of the water level range. Since the accompanying species mostly occurred on plots with lower water levels (Figure 5), it might be that *Typha* outcompeted the accompanying species at higher water levels of at least +20 cm to 30 cm. Thus, there seems to be less cover of accompanying species and more *Typha* shoots at higher water levels whereas there was more accompanying vegetation and less *Typha* in lower water levels. Accordingly, weed control on a *Typha* field (e.g., for perennial grasses such as *Phalaris arundinacea* L.) can be maintained by setting an adequate water level of at least +20 cm (Dubbe *et al.* 1988). Especially during early growing season, flooded field conditions can control weeds in an established stand (Dubbe *et al.* 1988). In addition, a continuously flooded site also favours biomass production of *Typha* (Li *et al.* 2004). Once *Typha* seedlings have established, they might grow faster than accompanying species and consequently outcompete them, as *Typha* is known to colonise the surface rapidly by clonal growth via rhizomes (Heinz 2012).

A trial harvest of a part (~1.3 ha) of the pilot field was conducted in December 2021, i.e., two years after site establishment. This did not affect the *Typha* stand development, nor the species composition based on the BRT analysis and the NMDS. *Typha* indeed regrows fast and builds a dense stand after harvest due to large rhizome systems (Bansal *et al.* 2019). The UAS-based maps (Figure 6) confirm the dense regrowth in 2022.The species composition of the accompanying vegetation on the *Typha* pilot site did not change between years two and three after establishment, which falsifies hypothesis 2a. Still, plant species composition showed considerable differentiation across the pilot site, which, according to the ordination, was mainly driven by the water level. In line with hypothesis 2b, species composition was strongly affected by water level, implying that there are species which mainly occur at high water levels, but there are other species occurring at low water levels. Overall, stable plant communities developed surprisingly fast (Figure 4).

While accompanying species composition did not change between the second and third year of this study, dominance patterns did change. The north-eastern part of the site was dominated by accompanying vegetation, especially *Juncus*, which grows at lower water levels, and less *Typha* (Figures 3, 4 & 6). In these sections *Typha* increased but did not reach a dominant level for much of the area (see Figure 6). Overall, dominance of non-*Typha*-species decreased over time.

Potential filter function of *Typha* paludiculture

Typha is known to perform well under a wide range of nutrient levels, reaching highest productivity at high nutrient levels, e.g., young *Typha* plants thrive at N loads up to 250 kg ha^{-1} y^{-1} (Geurts & Fritz 2018), whereas other studies found that *Typha* benefits from total-N additions up to 300 kg N ha^{-1} y^{-1} under controlled conditions (Ren *et al.* 2019).

Due to its high nutrient uptake capacity and its proven application in constructed wetlands, *Typha* can act as a biological filter that contributes to nutrient retention and water purification (Saydeger *et al.* 2004, Ciria *et al.* 2005, Heinz 2012). In the pilot site, the nutrient concentration patterns between the inlet and the potential outlet might indicate such filter function, despite the lack of a continuous flowing regime in the site. NH_4

concentrations in the porewater were higher at the inlet than at the potential outlet of the pilot site throughout the sampling period (July 2020–April 2022). Overall, there was a slight tendency of NH_4 concentrations to increase in the porewater in our pilot site, increasing from 2.2 ± 2.4 mg L^{-1} in 2020 (averaged for the pilot site) via 3.1 ± 3.1 mg L^{-1} in 2021 to 5.7 ± 2.0 mg L^{-1} in 2022. This reflects the typical post-rewetting increase caused by reducing conditions and enhanced mineralization of organic matter (Van Dijk *et al.* 2004, Zak *et al.* 2004, Zak & Gelbrecht 2007, Emsens *et al.* 2016, Lundin *et al.* 2017, Zak *et al.* 2018).

Elements such as K and Mg showed higher concentrations in the surface water at the inlet than at the potential outlet of the pilot site, consistent with the expected nutrient removal by plant uptake (Figure 8). Especially *T. latifolia* is known to be able to take up high proportions of nutrients like K, which can subsequently be removed from the system by aboveground biomass harvest (Geurts *et al.* 2020). This reduction in nutrients, whereby plants absorb nutrients that are then removed from the system through harvesting, is one of the important ecosystem services provided by paludiculture (Vroom *et al.* 2018).

Fe and Ca showed increased values in the porewater at the potential outlet of the pilot site (but no alterations in the surface water). The Fe content might have increased within the pilot site due to biogeochemical processes occurring after rewetting; Emsens *et al.* (2016) as well as Zak and Gelbrecht (2007) explained this by the fact that the insoluble Fe(III) is reduced to soluble Fe(II) by rewetting and especially by fluctuating water levels. The soil analysis supports these findings, showing strong increases in Fe concentration after rewetting (from 0.001 % to 1.03 % dry matter) and higher electrical conductivity. These changes indicate intensified redox dynamics, which may contribute to nutrient transformations and internal nutrient cycling rather than external retention. Ca concentrations did not only differ between in- and potential outlet with significantly higher concentrations at the potential outlet, but it seemed also that Ca concentrations tended to increase at the potential outlet and decrease at the inlet (Figure 7). Highly decomposed peat exhibits high contents of nutrients such as P, Fe, Al, and Ca (Zak *et al.* 2010). Yet, in comparison to the study sites assessed by Zak *et al.* (2010), Ca concentrations of 6060 ± 1540 mg L^{-1} are reported for sites with highly decomposed peat, 2740 ± 17 mg L^{-1} for moderately decomposed peat and 460 ± 280 mg L^{-1} for slightly decomposed peat. Thus, Ca concentrations of our pilot site were as low as in slightly decomposed sites (mean = 299.43 ± 118.68 mg L^{-1} for the inlet and 692.00 ± 85.10 mg L^{-1} for the potential outlet). Pfadenhauer and Wild (2001) compared nutrient concentrations in different parts of the *Typha* plant and in the soil; they found Ca concentrations to be highest in the *Typha* roots and comparatively low Ca contents in the soil. The intention of the surface water and porewater analysis at the in- and outlet of the pilot site was to assess the incoming nutrient loads and the extent of nutrient uptake by the vegetation when the water flows through the pilot site. Yet, it turned out that we did not have a flowing regime in the site. We rather pumped water into the pilot site from the Peene river, but did not create a clear water flow pattern where water passes through the potential outlet. The elevated EC as well as Ca and Fe in the porewater at the potential outlet of the pilot site compared to the inlet indicates a reduction in water volume, which is likely to lead to a concentration effect as nutrients are less diluted. This may indicate selective losses of water through processes such as evaporation or seepage, while nutrients remain, resulting in higher concentrations. The outlet was set high to pump only the minimum amount required in the field and the water remained in the *Typha* field, seeped into deeper soil layers and to the drained surroundings, or was released into the atmosphere by evapotranspiration, increasing the nutrient concentrations in the porewater. Therefore, nutrient removal by the site cannot be quantified simply by looking at the surface and porewater nutrients and the inlet and potential outlet of the pilot site. Moreover, the surface water samples were taken in the still water where dead parts of the plants, algae and the like were present, which might have falsified the results, especially for the potential outlet. Consequently, detailed studies of water flows and nutrient fluxes are currently under way.

In comparison to the nutrient values of the Peene river, the surface water nutrient values for total-N and P (averaged for inlet and potential outlet) developed differently; for the vegetation period (April to September) 2020 the Peene river had an average total-N load of 1.53 mg L^{-1} versus 3.35 mg L^{-1} ± 0.33 in the pilot site and an average P load of 0.13 mg L^{-1} of the river versus 0.18 ± 0.03 mg L^{-1} in the pilot site in 2020. In 2021, the N-load between the pilot site and the river did not differ as much; the annual average of total-N was lower in the river, with 1.49 mg L^{-1}, versus 1.83 ± 0.13 mg L^{-1} in the pilot site. P-loads were 0.16 mg L^{-1} in the river versus 0.20 ± 0.06 mg L^{-1} in the pilot site. In 2022, the annual average nutrient load of total-N was 1.66 mg L^{-1} in the river and 3.14 ± 0.58 mg L^{-1} in the pilot site and 0.10 mg L^{-1} P were measured in the river versus 0.17 ± 0.03 mg L^{-1} in the pilot site (Staatliches Amt für Landwirtschaft und Umwelt Mecklenburgische Seenplatte (StALU MS), 2024). N-loads seemed to increase from the river to the pilot site, going in line with our findings on NH_4 concentrations in the porewater which tended to increase over time after rewetting due to a high supply of organic matter as electron donor in combination with oxidizing substances (Zak *et al.* 2004, Zak & Gelbrecht 2007).

The P concentrations in the pilot site increased from vegetation period 2020 to 2021, but decreased again in 2022. As already described by Zak and Gelbrecht (2007) the P loads in rewetted fens are expected to increase. Highly decomposed peat, the result of ages of drainage and mineralization, lead to high P loads that may be mobilised by rewetting as the Fe(III) that bound P, is reduced to Fe(II) causing the release of previously bound inorganic P into the porewater (Zak & Gelbrecht 2007, Zak *et al.* 2010). Whereas other studies have found P rates to increase after restoration (Koskinen *et al.* 2017, Menberu *et al.* 2017), in our study the P values did not show such clear pattern. The rapid in- and decrease of P in the porewater near the inlet as well as averaged for in- and outlet might be reflecting short-time changes due to irrigation events using river water. Furthermore, the P content in our pilot site was comparatively low with values between 0.03 mg L^{-1} and 0.30 mg L^{-1}, whereas porewater concentrations of P between 3–14 mg L^{-1} are common in eutrophic wetlands (Zak *et al.* 2004). Van Dijk *et al.* (2004) revealed that, in degraded minerotrophic peatlands, a raise of the soil pH to about 7 causes the increase of P. In our pilot site, the soil pH was higher than 7 before rewetting (7.2 in July 2019, Table 2), and increased up to 8 in April 2022, but still, this did not provoke the increase of P in our pilot site. The zero point of charge for Fe(III) is known to be at pH 7.9 to 8.1, meaning that until then, P is bound to Fe(III) (Zak & Gelbrecht 2007).

In summary, the observed nutrient dynamics between inlet and potential outlet indicate that *Typha* stands may contribute to nutrient transformation and partial retention. However, the interpretation of nutrient dynamics is limited due to the absence of a flowing regime, a drained control site, and by the relatively small number of sampling dates (seven in total) over the three sampling years. Thus, the expected decrease in porewater and surface water nutrient concentrations from inlet to outlet could not be consistently demonstrated and the results should be regarded as indicative trends rather than robust quantifications of rewetting effects. Consequently, hypothesis 3, assuming a clear decrease in nutrient concentrations, reflecting the filter function of *Typha* plants, can only partly be supported.

Rewetting not only affects nutrient concentrations but also modifies soil physical properties; bulk density increased from July 2019 (before rewetting) to September 2020 (one year after rewetting). It is known that the bulk density of peat increases after drainage and mineralization of peat (Wells & Williams 1996, Minkkinen & Laine 1998, Emsens *et al.* 2020). After rewetting, these properties change again, leading to lower bulk densities since the peat "swells up again" and new "proto-peat" accumulates (Ahmad *et al.* 2020, Mrotzek *et al.* 2020). This increase of soil elevation was also observed in the pilot site. Indeed, the soil surface elevations of our sampling plots increased significantly by 10 cm from 2020 to 2022 and formation of up to several centimetres of dead plant residues was observed in our site (unpublished data). However, the cited studies were conducted in fens 20 years after rewetting. The significant increase in bulk density within one year after rewetting in our study was surprising and might not reveal a pattern but might be a variation which will be levelled out within the following years. Another explanation for the increase within one year after rewetting could be that the increase in bulk density is due to the site establishment, as heavy machinery might have compacted the top soil (e.g., during construction work or planting). However, as the bulk density directly after rewetting in September 2019 was only marginally higher than before rewetting, soil compaction as the main reason for the increased bulk density seems unlikely. As a result of ages of drainage and therefore decomposition of the peat, the top 10–15 cm of the soil contained a relatively small proportion of organic matter (25.2 ± 10.3 % loss of ignition in July 2019) in comparison to what Joosten and Clarke (2002) describe as peat (> 30 % organic matter). It might be that higher proportions of mineral components due to the oxidation of organic matter led to a compaction of the top soil layer after rewetting (Mrotzek *et al.* 2020).

Lessons learned and implications for site establishment

The establishment of the 10 ha *Typha* paludiculture pilot site was the first one of this size. We gained valuable experience regarding the following aspects: Levelling the uneven surface with a caterpillar dozer to enable efficient water management disturbed the sward of previous grassland vegetation at large scale. No additional soil cultivation was required for planting and seeding. It should be noted, however, that sod removal took place only at small scale in the elevated parts, at shallow depth and the majority of the material remained on the site. Top-soil removal would instead remove substantial amounts of nutrients and is therefore not generally recommended for fen paludicultures (Eickenscheidt *et al.* 2023a, Nordt *et al.* 2024). Alternative site-preparation methods such as shallow soil tillage and seedbed preparation may be more appropriate and should be further tested in future. In our case, sod removal was applied primarily for levelling the uneven surface to enable efficient water management. Thus, our observations represent practical experience rather than an experimentally tested treatment. Our site-specific recommendations are largely consistent with the two existing guidelines on paludiculture establishment (Eickenscheidt *et al.* 2023a, Nordt *et al.* 2024). Both guidelines identify reliable water management as the most important factor for establishment, emphasising that sowing

or planting should only begin once a fully functional water management is in place to enable rapid and sustained rewetting. Likewise, both sources recommend the removal of the old grass sward to suppress competition from accompanying vegetation. Furthermore, both guidelines highlight spring to early summer as the optimal window for establishment, matching our finding that late planting in autumn is risky due to early frosts and limited time for allocating sufficient resources for regrowth in spring.

For the planting, we observed a low survival rate of seedlings which was likely related to the combination of several detrimental factors including the late planting date in September 2019, early frost events, washout in flooded furrows during winter and disturbance by waterfowl. Waterfowl damaged the stand in the first winter 2019/2020 by pulling out seedlings, but could be successfully deterred by large colourful inflatable pool toys in the following years. A controlled stand establishment was impeded by fluctuating water levels during the first vegetation period 2020. With permanently installed irrigation infrastructure, the site was constantly waterlogged in 2021 and 2022. Despite the very low planting density (approximately 0.5–1 plant per m^2) compared to other pilots, large distance between planting rows (planned: 2 m, realised average: 2.20 m) and detrimental factors during the sensitive establishment phase, *Typha* has successfully spread across the whole site. Reasons were quick clonal growth of the surviving seedlings as well as germination from the seed bank and after seeding. Since *Typha* germinates optimally with a water level close to surface, there is a trade-off with stand management, especially weed management. The legacy of former grassland species adapted to fluctuating water levels required especially high water levels to suppress accompanying vegetation by inundation (> +15 cm). In addition, the microrelief across such a large site prevented the establishment of a single optimal water level. On the other hand, any water level higher than 10 cm was beneficial for *Typha*. A representative number of water level loggers throughout the entire pilot site would have been particularly important in 2020 when water levels dropped below soil surface. Especially with water levels below soil surface, data extrapolation should be treated with caution, as water dynamics in the soil might differ. For example, areas with the sod still present might have different bulk densities and porosities than lower peat layers (Liu *et al.* 2020, Menberu *et al.* 2021).

For future site implementations, water management remains to be the key factor for the successful establishment of *Typha* paludiculture, as consistently highlighted by Geurts and Fritz (2018), Eickenscheidt *et al.* (2023a) and Nordt *et al.* (2024). Sites should be equipped for stable, adjustable water levels before planting or seeding, ideally allowing gradual water level increases during the establishment phase to balance seedling growth or germination success and weed suppression. Planting should preferably take place in early summer, after the risk of late frosts has passed, to maximise seedling survival and allow for clonal growth and sufficient resource allocation to the root zone before the first winter. Planting densities of 0.5 or 1 plant m^{-2} could be tested again with earlier planting time, as established plants showed promising signs of quick clonal spread. Densities should be considered to increase if the planting conditions are not expected to be optimal. Previous studies indicate that seeding can be a viable alternative to planting under suitable hydrological conditions. Eickenscheidt *et al.* (2023b) showed that *T. angustifolia* and *T. latifolia* germinate well under fully rewetted conditions when sown in early summer and can reach similar yields as planted stands, whereas autumn sowings and sowing under only partially rewetted or drained conditions largely failed. In addition, the regional climate needs to be considered accordingly for other sites. Eickenscheidt *et al.* (2023b) recommended seeding for *Typha* and further paludiculture species already for mid-April until mid-May due to the risk of the seeds or seedlings dying when the dark peat heats up with irradiation towards summer solstice, especially if rewetting does not take place immediately. Whereas *Typha* seed germination most commonly occurs on moist but not inundated, exposed soils, we observed successful germination at our pilot site even under shallow inundation. Similar observations were made for *T. latifolia* under shallow flooding of 4–15 cm (Heinz 2012, Meng *et al.* 2016). A factual report suggests that the optimal water level for the germination of *T. latifolia* lies between the soil surface and approximately +15 cm, with seeds sinking into shallow water where they germinate (Jansen 2021). Overall, these experiences demonstrate that *Typha* can successfully establish and expand under variable conditions once hydrological stability is achieved, highlighting the importance of early establishment and functioning of water management infrastructure, flexible water control, and adaptive management for large-scale paludiculture systems. Seeding should be further explored as an effective, practicable and cost-efficient method for stand establishment and replenishing gaps – either as alternative to planting or to enhance planting densities. Our tests with drone-based seeding in 2020 were to our knowledge the first application for paludiculture and demonstrated the general feasibility. Drones enable targeted seeding under water-logged soil conditions and are advantageous compared to seeding methods depending on dry conditions before rewetting.

While maintaining high and stable water levels was identified as crucial for successful *Typha* stand establishment, this also raises important considerations regarding climate impacts of these intended water levels. Average water levels several centimetres above the soil surface resulted in the pilot site resembling a

constructed wetland rather than a typical fen peatland. This outcome reflects the broader finding that rewetting does not return drained fens to their original ecological state (Kreyling *et al.* 2021). Although consistently high water levels prevent peat oxidization and thus CO_2 emissions, they may also increase CH_4 release due to the predominantly anoxic conditions (Franz *et al.* 2016, Wilson *et al.* 2016, Günther *et al.* 2020). Consequently, the overall GHG mitigation potential of this *Typha* paludiculture pilot site depends on the balance between CO_2 emission reduction through rewetting and the potential CH_4 increase under prolonged inundation. Paludicultures with deeper groundwater levels, e.g., up to –0.12 m below soil surface, were found to have greater GHG reduction potential than waterlogged paludicultures, as CH_4 emissions are known to increase at water levels < –0.2 m (Tiemeyer *et al.* 2020, Bockermann *et al.* 2025). In case of *Typha* paludiculture, a mature and dense *Typha* stand may sufficiently suppress accompanying vegetation by shading and allow for (temporally) lower water levels to limit CH_4 emission. Further research is required to elaborate balanced management recommendations for different stages of *Typha* cultivation.

However, Günther *et al.* (2020) showed with their radiative forcing model that despite potential CH_4 emissions, prompt rewetting is crucial for climate change mitigation, due to the durability of CO_2 as GHG in the atmosphere in comparison to CH_4, which is a stronger but short-lived greenhouse gas, as long as sufficient free OH radicals are present in the atmosphere. Future monitoring and paludiculture management should therefore include GHG flux measurements including CH_4 as well as carbon budget assessments to identify optimal water level regimes that maximise both stand establishment and productivity while at the same time supporting climate mitigation.

CONCLUSIONS

To put *Typha* paludiculture into practice, it is important to test the cultivation, establishment and long-term development at large scale. In 2019, we set up a 10-ha experimental *Typha* paludiculture pilot site in north-east Germany on a formerly drained fen peatland. Over the first three years after rewetting, *Typha* expanded across the site and the overall biomass production increased steadily. The initial planting success was low, but sowing and germination from the seed bank promoted gap closure within the heterogeneous stand. Accompanying vegetation cover remained below 40 % and did not change substantially between the second and third year, suggesting that water level is a key driver of species composition. *Typha* benefits from relatively stable water levels not dropping below soil surface or exceeding +35 cm above soil surface. With UAV data, plant species dominance development can be reliably mapped over time. After rewetting, certain nutrients like Fe or NH_4 are released from degraded peat, whereas K was taken up by plants. Long-term monitoring is important to assess future development of both the *Typha* stand and environmental drivers such as nutrient dynamics in soil and water. Lessons learned from this site can inform the establishment of new sites: We expect higher planting success if planting happens earlier in the growing season while considering local climate for planting time in order to avoid late spring frost or intense substrate heating into summer. When planting is expected to occur during non-optimal conditions regarding climate and substrate, higher planting densities could support successful establishment. Sowing by drones appears to be a promising measure to fill gaps in the stand and should be tested for exclusive measure of stand establishment. Water levels above ground throughout the year are crucial in the establishment phase.

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AUTHOR CONTRIBUTIONS

MB: analyses, writing (first draft), visualization, lead author; NK: data sampling, support with data analysis; BB: data sampling; JD: data sampling; KH: data sampling; CH: data sampling, data analysis remote sensing; ML: data sampling; SvdL: conceptualization and methodology remote sensing, resources; JN: data sampling, coordination of the pilot site; HJ: conceptualization, resources, head of project; D-VN: data analysis remote sensing; SS: data sampling; SW: conceptualization, data sampling; JK: conceptualization, methodology, resources, supervision; all authors: writing (review and editing).

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AUTHOR FOR CORRESPONDENCE
Meline Brendel, Institute of Botany and Landscape Ecology, University of Greifswald, Partner in the Greifswald Mire Centre, Soldmannstraße 15, 17489 Greifswald, Germany; phone: +49 3834 420 4226; e-mail: meline.brendel@uni-greifswald.de

APPENDIX

Figure A1. Map of peat thickness at the pilot site and its surroundings, provided by Matthias Lampe.

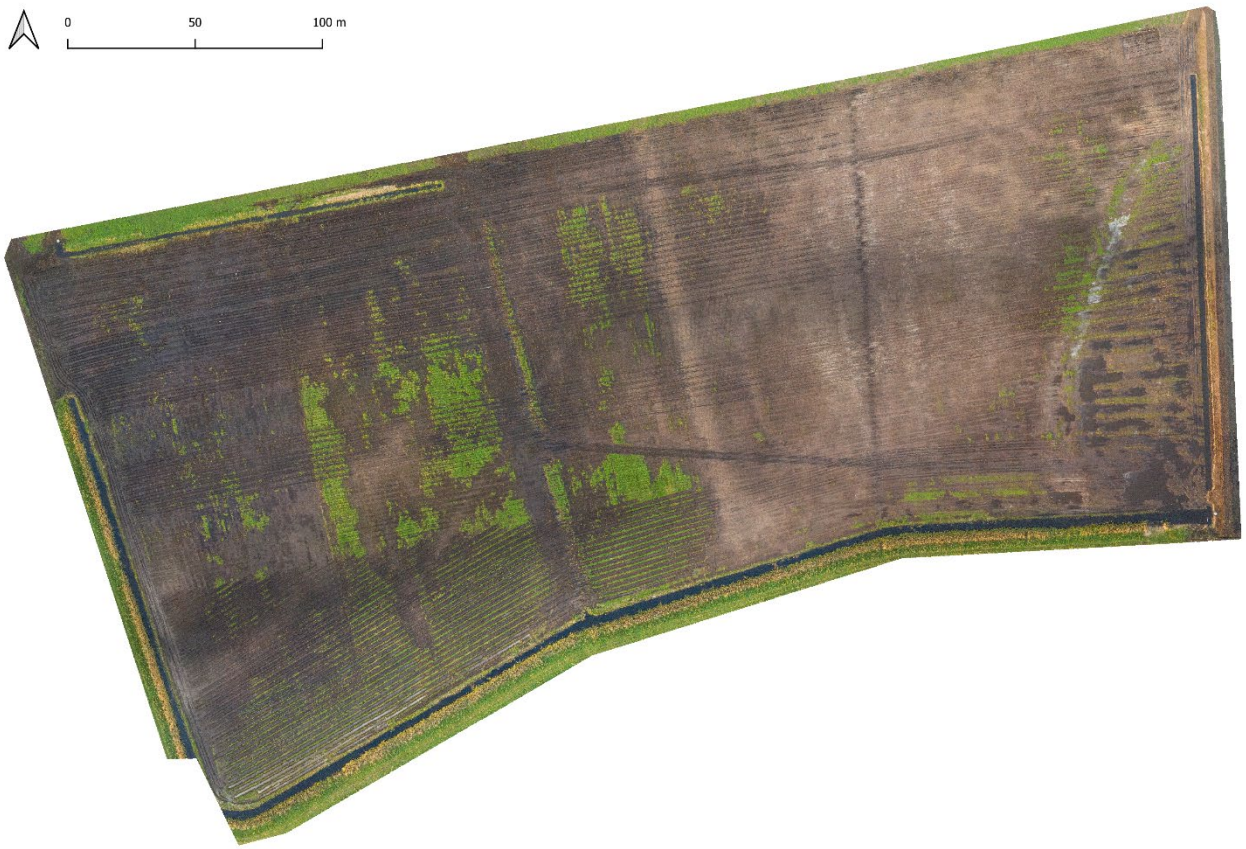

Figure A2. Aerial view of the pilot site after site construction and planting, which took place from 16–20 September 2019. The photo was taken on 15 October 2019 and illustrates the heterogeneous starting conditions for *Typha* establishment, particularly with respect to sod removal, soil disturbance, and furrow size.

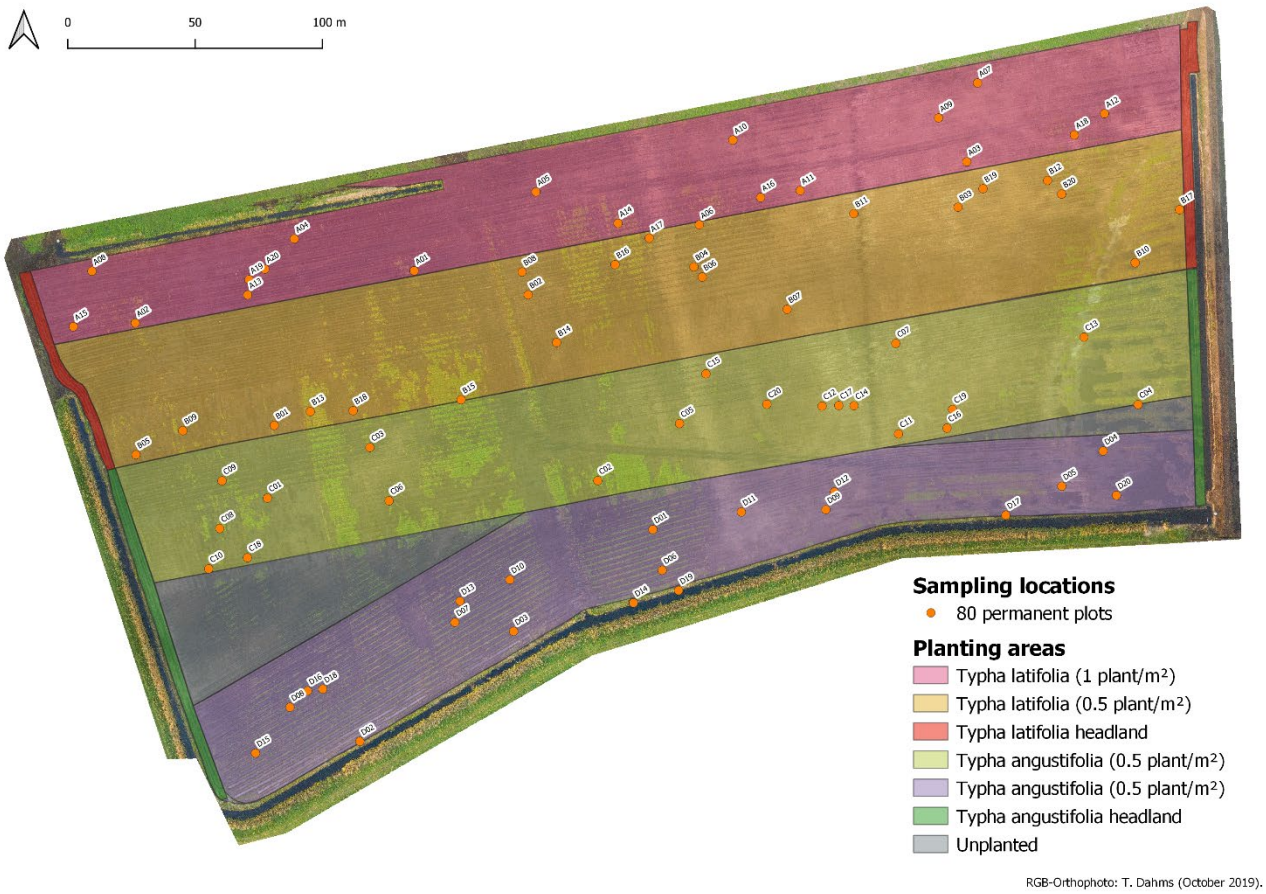

Figure A3. 80 randomly distributed permanent plots used for vegetation sampling within the planting areas as of September 2019. The two areas, *Typha angustifolia* headland and *Typha latifolia* headland, were not included in the random location of the plots.

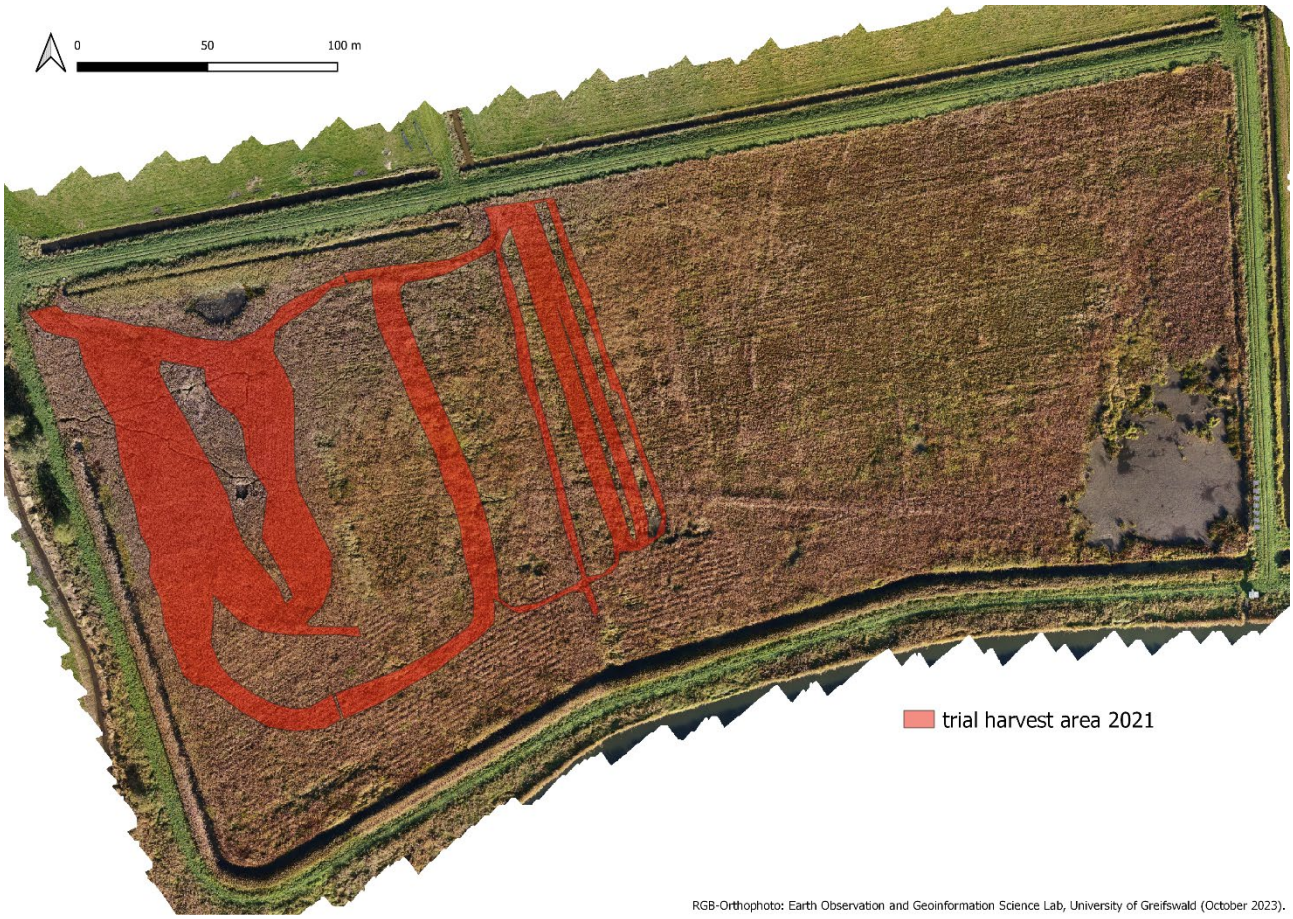

Figure A4. Area of the pilot site that was harvested during the trial harvest in December 2021.

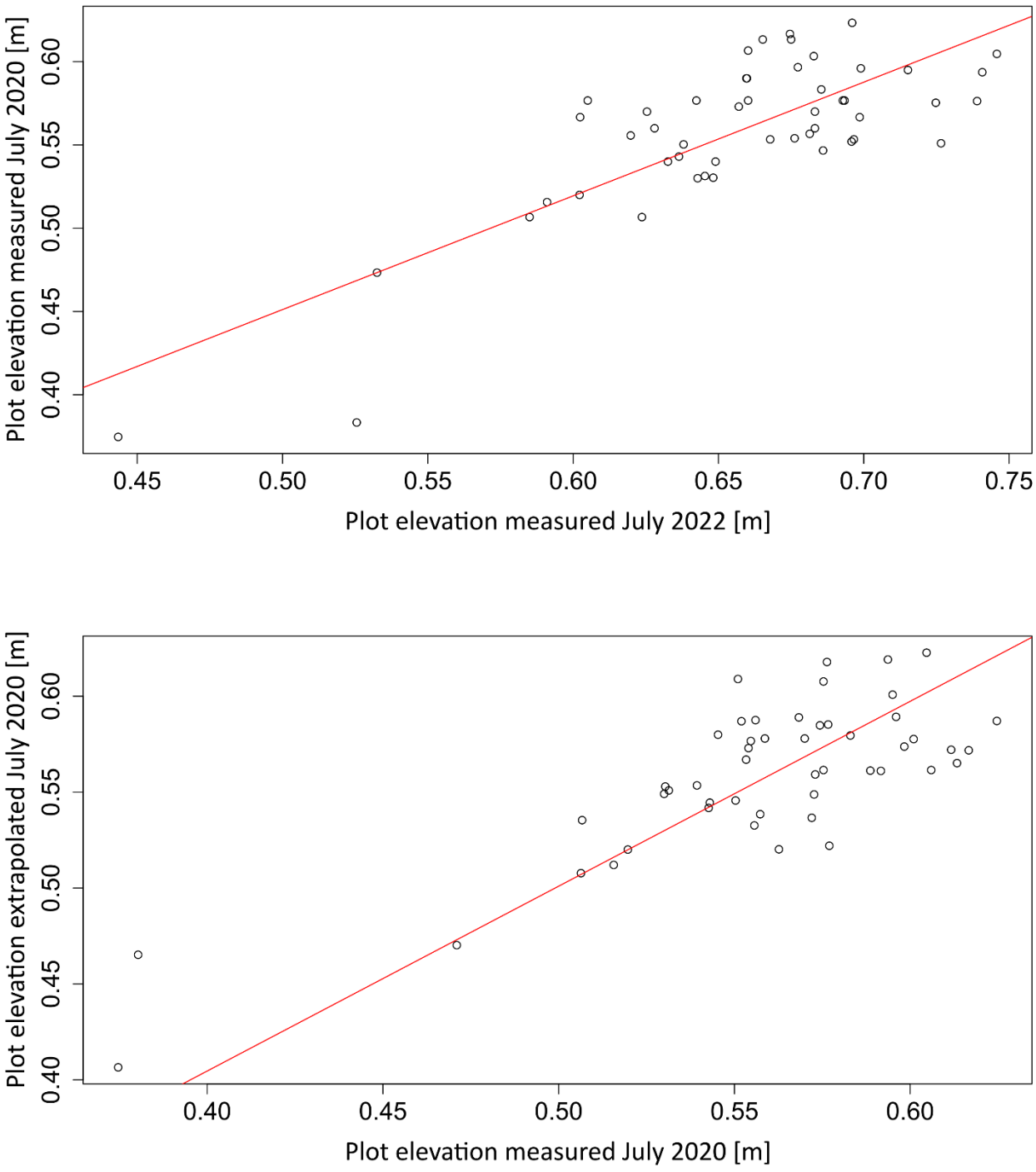

Figure A5. Scatter plots showing the relationship between measured plot elevations in 2020 vs. 2022 and between the measured and extrapolated plot elevations for 2020. Since plot elevation data were unavailable for 2021 and incomplete for 2020, elevation data were extrapolated for all plots across all three years. The linear regression for the 2020 data is described by $y = 0.65x + 0.19$ with $R^2 = 0.63$, and for the 2020 vs. 2022 data by $y = 0.68x + 0.11$ with $R^2 = 0.62$.

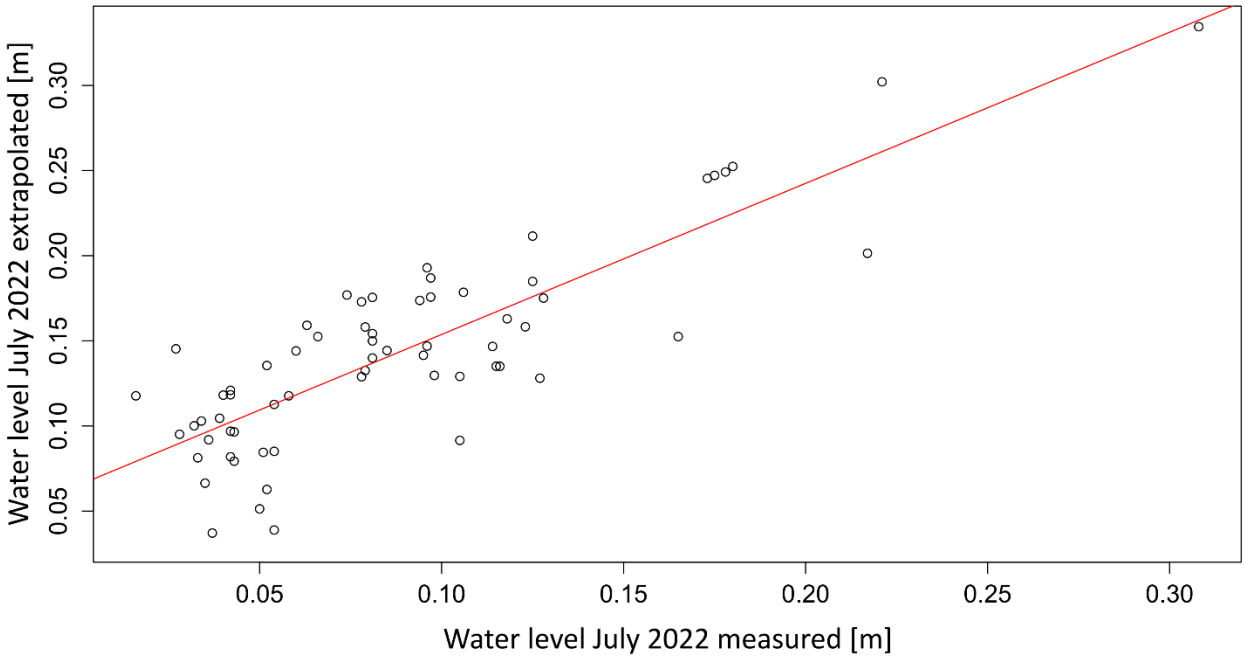

Figure A6. Scatter plot showing the relationship between the measured water levels and extrapolated water levels for the sampling period 2 July to 7 July, 2022. The linear regression is described by $y = 0.89x + 0.07$ with $R^2 = 0.71$.

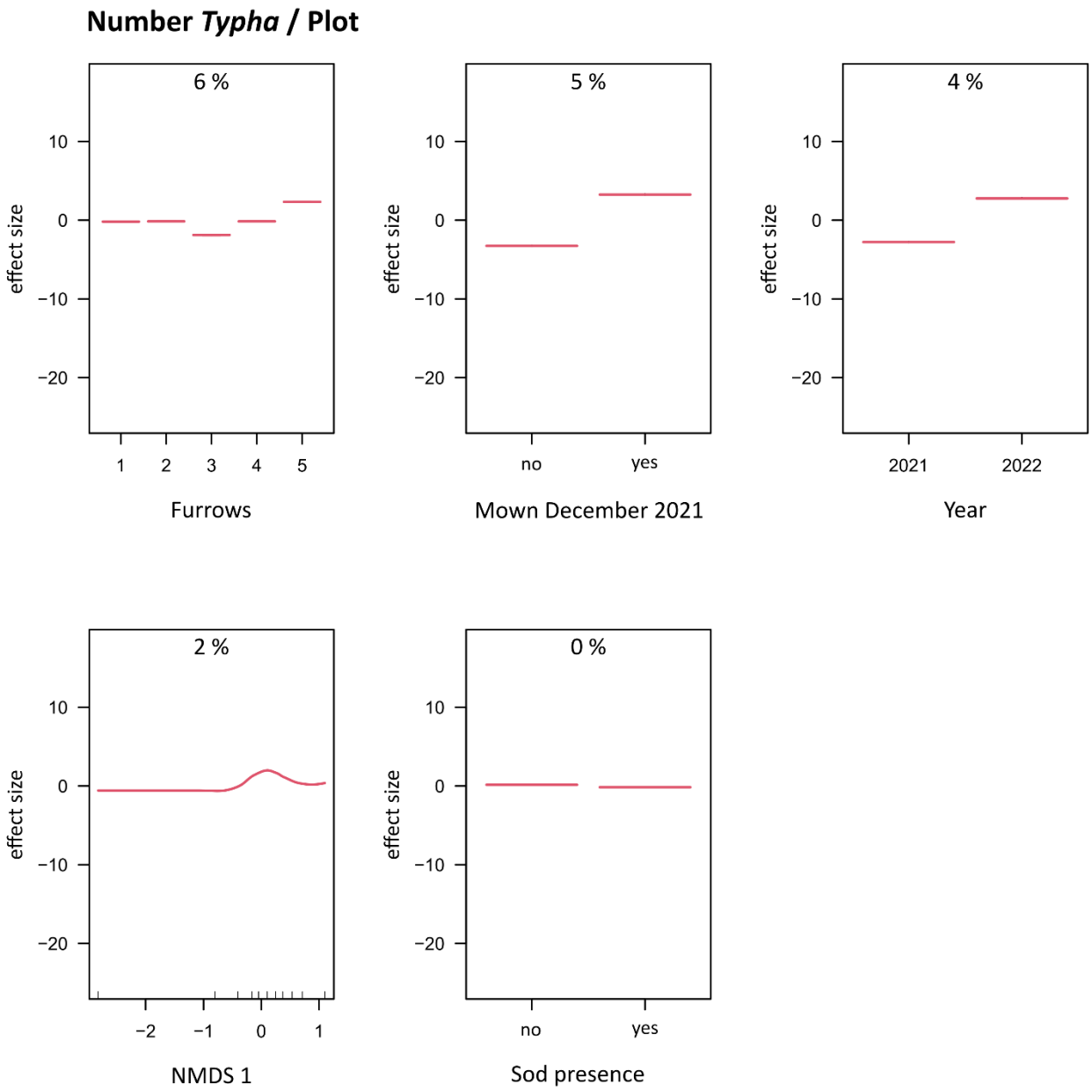

Figure A7. *Typha* stand development, expressed as the number of *Typha* individuals per plot, in relation to explanatory variables as found by a BRT analysis. Positive effect sizes indicate more *Typha* shoots per plot than average, negative effect sizes indicate shoot numbers below the average. CV correlation was 0.77. Upward ticks along the x-axes indicate data distribution in steps of 10 %. Variables accounting for > 10 % of the variance in the response variable explained by the model are depicted in Figure 3. Furrow levels are: 1 = no furrows, 2 = small furrows, 3 = partially no furrows + partially small furrows, 4 = large furrows, 5 = partially no furrows + partially large furrows.

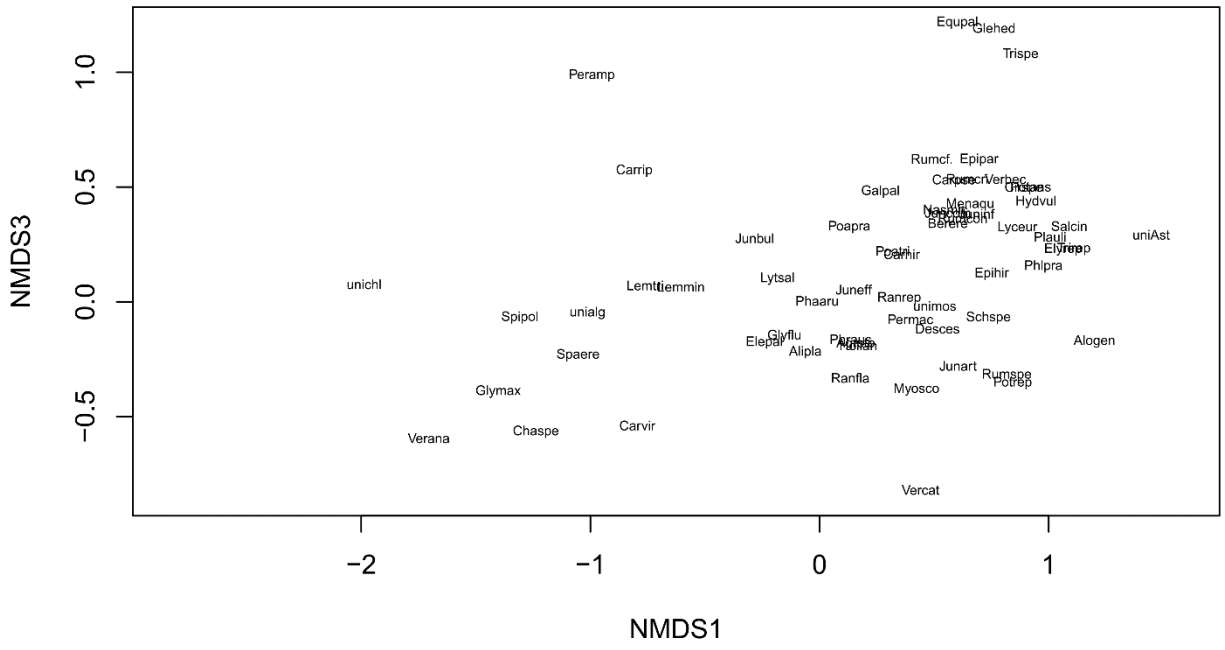

Figure A8. Non-metric Multidimensional Scaling (NMDS) depicting the species composition of the 80 sampling plots over 2021 and 2022. Species name abbreviations are given in Table A1.

Table A1. Abbreviations from Non-metric Multidimensional Scaling (NMDS) depicting the species composition with the respective species names.

Abbreviation	**Species name**
Agrsto	*Agrostis stolonifera*
Alipla	*Alisma plantago-aquatica*
Alogen	*Alopecorus geniculatus*
Berere	*Berula erecta*
Calsep	*Calystegia sepium*
Caracu	*Carex acutiformis*
Carhir	*Carex hirta*
Carpse	*Carex pseudocyperus*
Carrip	*Carex riparia*
Carvir	*Carex viridula*
Chaspe	*Chara* sp.
Cirspe	*Cirsium* sp.
Desces	*Deschampsia cespitosa* agg.
Elepal	*Eleocharis palustris* agg.
Elyrep	*Elymus repens* agg.
Epihir	*Epilobium hirsutum*
Epipar	*Epilobium parviflorum*
Equpal	*Equisetum palustre*
unialg	*unidentified algae 1*
Galapa	*Galium aparine*
Galpal	*Galium palustre* agg.
Glehed	*Glechoma hederacea*
Glyflu	*Glyceria fluitans* agg.
Glymax	*Glyceria maxima* agg.
Hollan	*Holcus lanatus*
Hydvul	*Hydrocotyle vulgaris*
Junart	*Juncus articulatus*
Junbul	*Juncus bulbosus* agg.
Juncom	*Juncus compressus*
Juneff	*Juncus effusus*
Juninf	*Juncus inflexus*
Lemmin	*Lemna minor*
Lemtri	*Lemna trisulca*
Lyceur	*Lycopus europaeus* agg.
Lytsal	*Lythrum salicaria*
Menaqu	*Mentha aquatica*
Myosco	*Myosotis scorpioides* agg.
Nasmic	*Nasturtium microphyllum*
Peramp	*Persicaria amphibia*
Permac	*Persicaria maculosa*
Phaaru	*Phalaris arundinacea*
Phlpra	*Phleum pratense*

Phraus	*Phragmites australis*
Plauli	*Plantago uliginosa*
Poapra	*Poa pratensis* agg.
Poatri	*Poa trivialis*
Potans	*Potentilla anserina*
Potrep	*Potentilla reptans*
Ranfla	*Ranunculus flammula*
Ranrep	*Ranunculus repens*
Ransce	*Ranunculus sceleratus*
Rumcon	*Rumex conglomeratus*
Rumcri	*Rumex crispus*
Rumcf.	*Rumex* cf. *obtusifolius*
Rumspe	*Rumex* sp.
Schspe	*Schoenoplectus* sp.
Salcin	*Salix cinerea*
Salspe	*Salix* sp.
Spaere	*Sparganium erectum* agg.
Spipol	*Spirodela polyrhiza*
Tar.sp	*Taraxacum* sp.
Tricf.	*Trifolium* cf. *pratense*
Trirep	*Trifolium repens* agg.
Trispe	*Trifolium* sp.
uniAst	*unidentified Asteraceae*
Verana	*Veronica anagallis-aquatica*
Verbec	*Veronica beccabunga*
Vercat	*Veronica catenata*
unimos	*unidentified moss*
unichl	*unidentified chlorobiont*

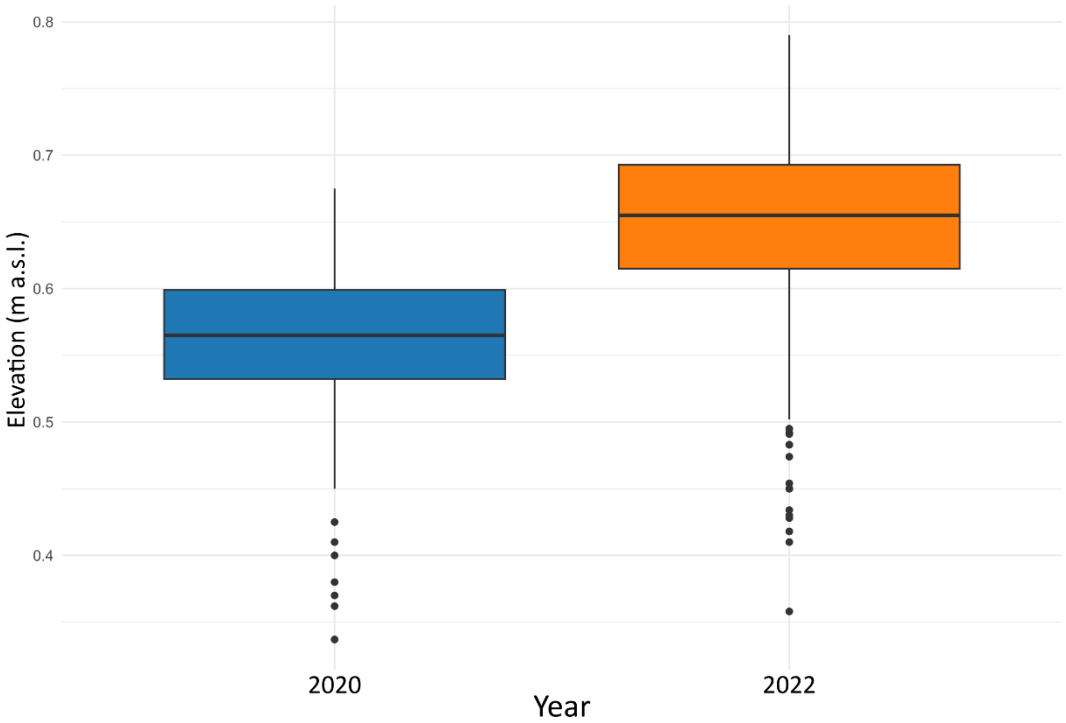

Figure A9. Boxplots of soil elevations of the plots measured in 2020 and 2022.