# SERVIPEAT

## a tool for assessing co-benefits of peatland rewetting in a local and regional scale

Manual

Version 1.3

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# 1. Introduction

Rewetting of peatlands remains the key measure to improve their ecological status. Although every peatland is different, hydrological processes standing behind the resaturation of the formerly drained peat soil remain universal. Wishing to make peatland rewetting more feasible and wishing to provide the potential peatland managers with quantified benefits, in the DESIRE project we developed, programmed and published the SERVIPEAT tool.

SERVIPEAT is a comprehensive, modelling tool developed in the DESIRE project as an output of Workpackage 2. It is available online at https://servipeat.sggw.edu.pl and as such can be used to calculate approximate ditch spacing and heights in ditch-peatland systems desired to be rewetted, as well as the probable consequences of rewetting expressed as nitrogen retention, phosphorus outwash, water retention volume gained and approximate value of ecosystem services gained as a result of peatland rewetting. SERVIPEAT consists of complex empirical and physical algorithms that can be used either in the simplified mode (most of the parameters remain specified and related to the ditch state (Fig. 1.1) and peatland types (Fig. 1.2)) or in the full mode, where the final user can specify all of the parameters required for the calculation.


**Figure 1.1.** SERVIPEAT dialog window in the simplified mode – selection of ditch types.


**Figure 1.2.** SERVIPEAT dialog window in the simplified mode – selection of hydrological types of a peatland to be rewetted.

Calculation algorithm depends on the number of ditches in the system to be rewetted (Fig. 1.3) that should also be specified by the user by clicking on the selected figure (in the simplified mode) or by selecting the appropriate calculation algorithm in the comprehensive mode.

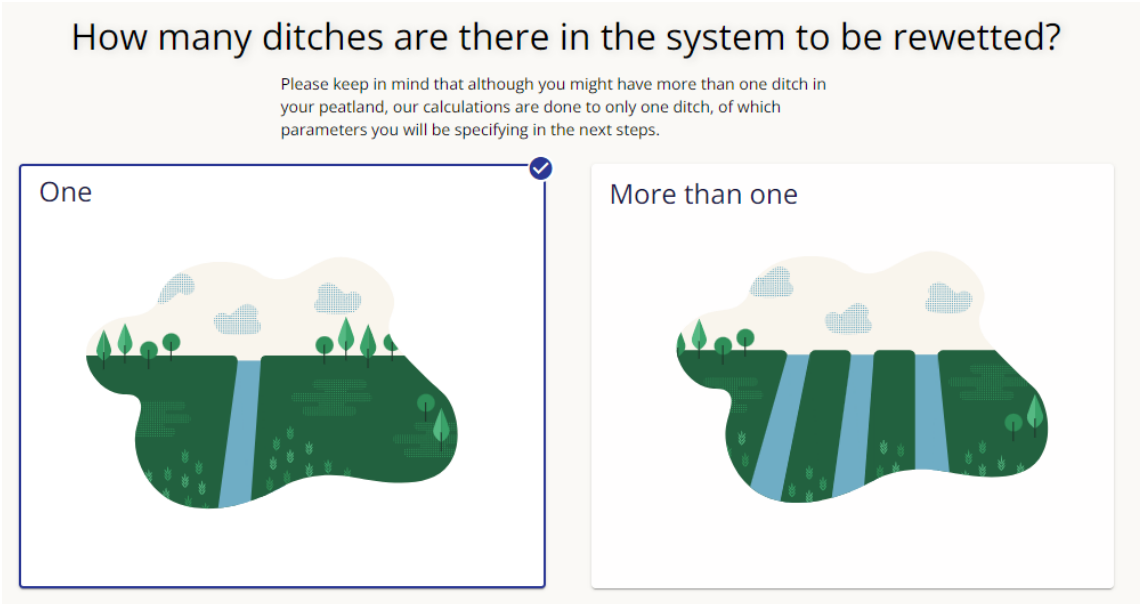


**Figure 1.3.** SERVIPEAT dialog window in the simplified mode – selection of calculation path that depend on the number of drainage ditches in the system to be rewetted.

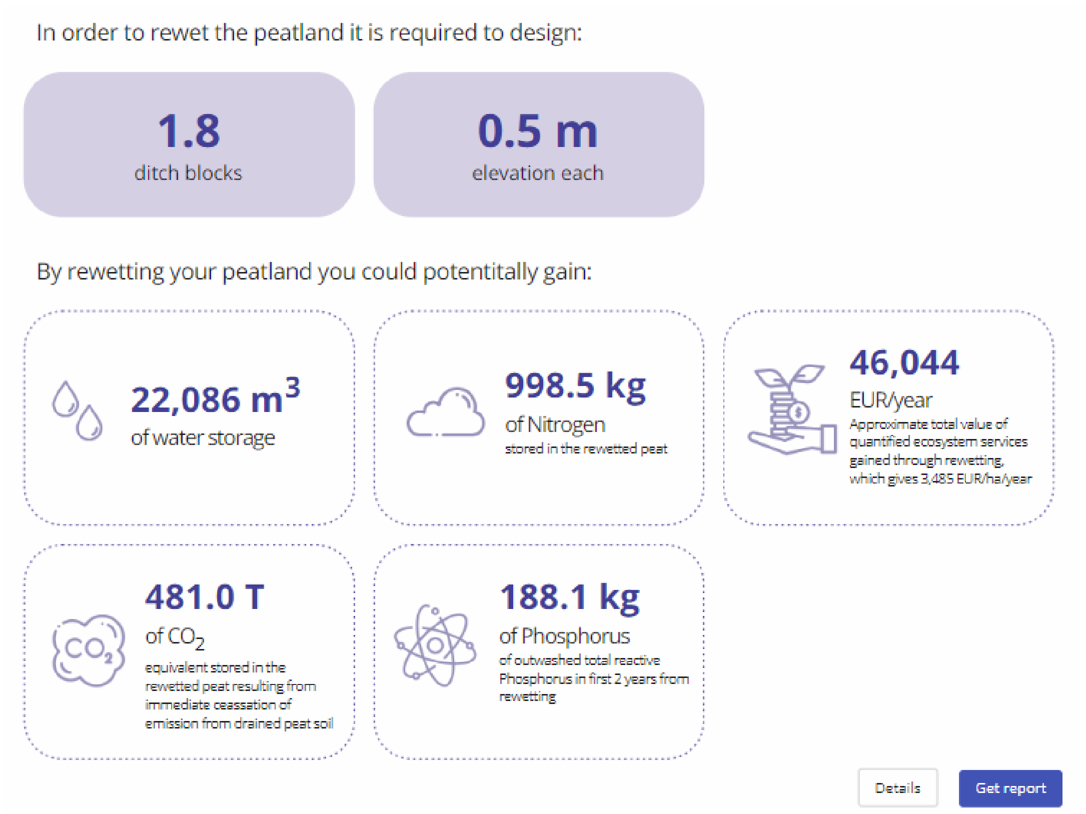


**Figure 1.4.** SERVIPEAT output dialog window.

Final users of the SERVIPEAT should be aware that the results of calculations provided by the model on its last page (Fig. 1.4) should be considered only as an indicator values and that the final development of peatland rewetting initiatives should be a subject of decent and comprehensive hydrological and biogeochemical study tailored to environmental features of the peatland to be rewetted.

# 2. Description of methods and algorithms

## 2.1 Hydrological module

### *2.1.1 General assumptions*

Water flow in peatlands under influence of drainage – irrigation systems is the effect of numerous processes (Fig. 2.1). In particular, peatland rewetting involves proper water damming height in the ditch network so that the groundwater depth within the peatland area could reach the assumed value of „s" that provides sufficient peat moisture content. In case of two parallel ditches, located at the distance equal to L from each other, „s" becomes groundwater depth in midspacing (L/2). For a single ditch scheme, s becomes the required depth at the ditch water divide.

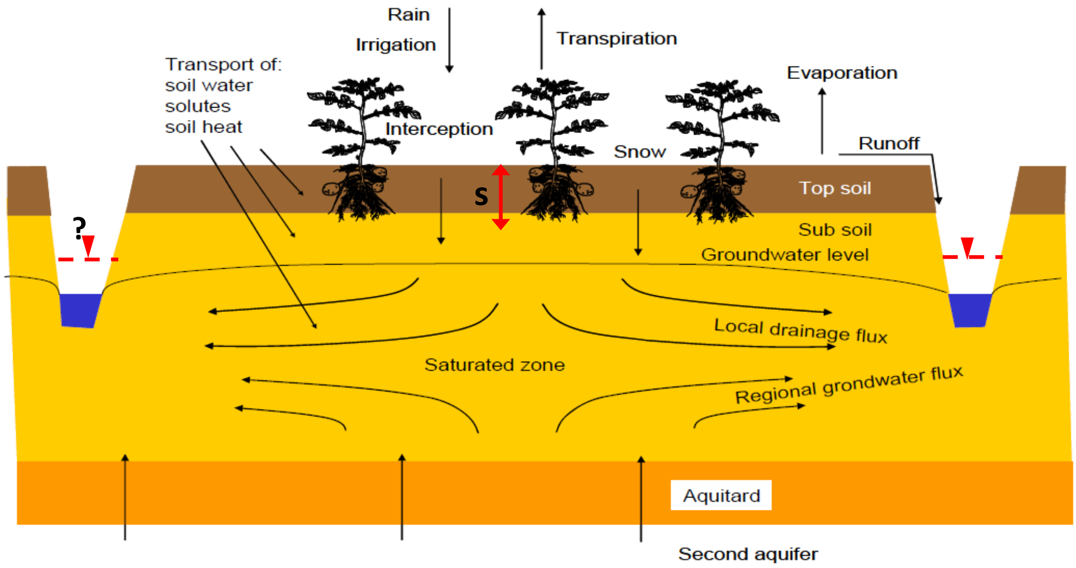


**Figure 2.1.** Model domain and transport processes (Arnold et al., 2012**).**

Water level scheme before and after weir installation in a drainage- irrigation ditch was given in Fig. 2. Then, the execution of peatland rewetting involves a solution to a following problem: which distance between the weirs provides such water damming level that causes the groundwater depth in midspacing or at a ditch water divide to reach the required value (Fig. 1). Because of the purposes of this elaboration (easy to implement, practical guidelines) a very simplified model was built, that enabled the estiamtion of weir height, the distance between the weirs, the amount of water retention and greenhouse gases emission from a peatland.

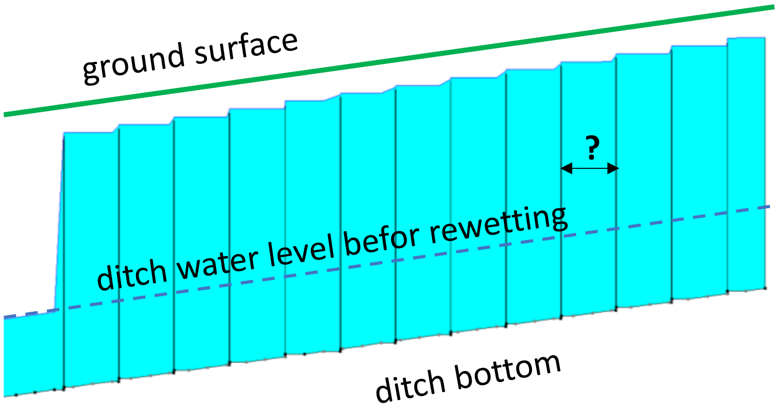


**Figure 2.2.** Water level in a ditch (before and after rewetting)

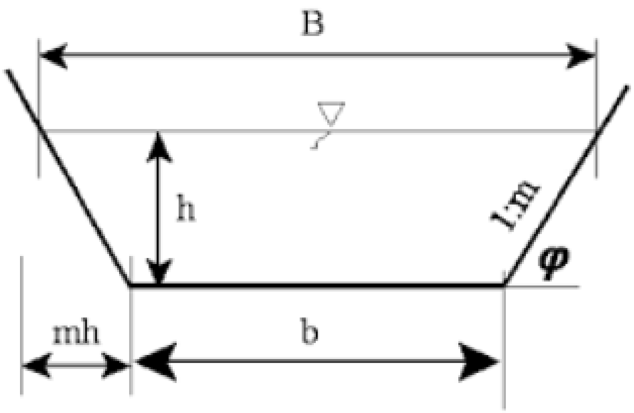


**Figure 2.3.** Ditch geometry scheme**.**

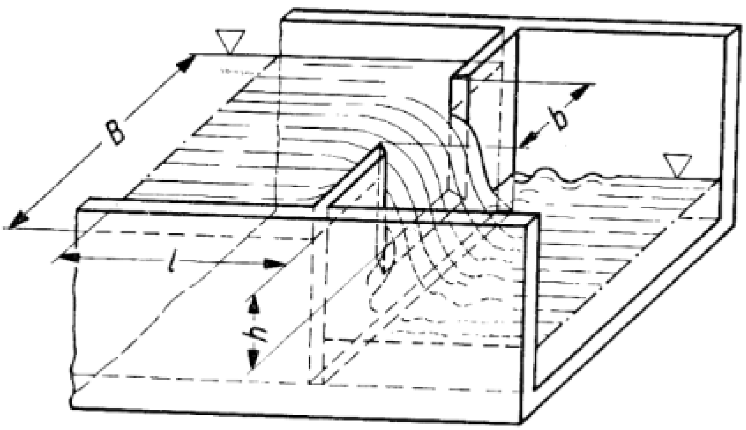


**Figure 2.4.** Weir water flow scheme.

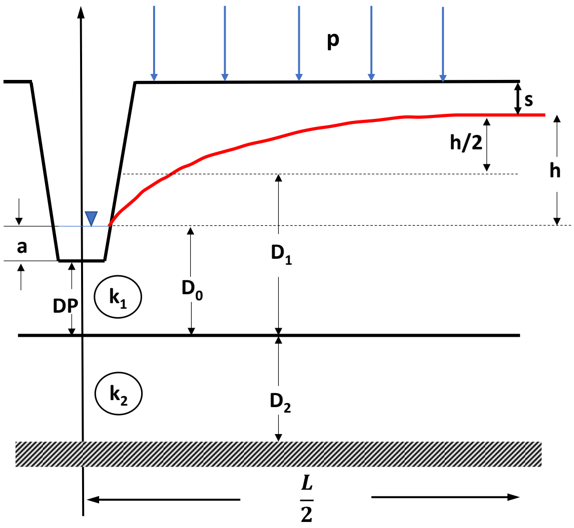


**Figure 2.5.** Assumed scheme of groundwater inflow to a ditch

The elaborated model involves three components, where the fist one describes water flow in a ditch, the second refers to the ditch backwater curve and the last is dedicated to groundwater inflow to a ditch in a flat area fed mainly by rainfall water. All three components execute calculations in steady- state conditions.

### *2.1.2 Gradually varied steady flow in a drainage ditch*

The utilized model assumed a trapezoidal ditch geometry (Fig. 2.3), which, in such situation, becomes defined by three parameters:

d – ditch depth [m],

b – ditch bottom width [m],

ns – bank slope (1:ns).

The ditch discharge is calculated by Mannings' formula (Chadwick et al., 2021):

$$Q = \frac{1}{n} A R_h^{\frac{2}{3}} J^{\frac{1}{2}} \qquad \text{(Eq. 2.1)}$$

where:

A – active cross-sectional area [$m^2$],

$R_h$ – hydraulic radius [m],

J – hydraulic slope [-],

n – roughness coefficient [$m^{-1/3}/s$],

Q – discharge [$m^3/s$].

Basing on literature references (Chow, 1959) three states of ditch maintenance were assigned roughness coefficients – n values in the applied modeling procedure (non maintained ditch n=0.3, scarcely managed n = 0.05, well maintained ditch = 0.01 ). The weir discharge *Q* was defined for a rectangular broad crested weir (Fig.2.4) by a following formula (Chadwick et al., 2021):

$$Q = \frac{2}{3}\mu b\sqrt{2g}h^{\frac{3}{2}} \qquad \text{(Eq. 2.2)}$$

where:

μ - discharge coefficient [-],

b – crest length [m],

h – water table elevation over the crest of the weir [m],

g – gravity accelaration [m/s$^2$].

The width of the weir *b (crest length)* is approached through the width of the water table in a ditch for a given water depth. The discharge coefficient takes the value of 0.44 after available literature sources (Kubrak et al., 2004) for the conditions of a rough inlet and also a rough crest. Dammed water level position between two weirs (Fig. 2.6) was calculated in the proposed modelling procedure by Bernoulli equation (Kubrak et al., 2004):

$$z_1 + H_1 + \frac{\alpha v_1^2}{2g} = z_2 + H_2 + \frac{\alpha v_2^2}{2g} + h_{str} \qquad \text{(Eq. 2.3)}$$

where:

$H_1$, $H_2$ – water depth in the upstream and downstream cross-section [m],

$v_1$, $v_2$ – water velocities in the upstream and downstream cross-section [m/s],

$z_1$, $z_2$ – height above the reference level [m],

$h_{str}$ – energy losses [m],

α – Coriolis coefficient (α=1.1)

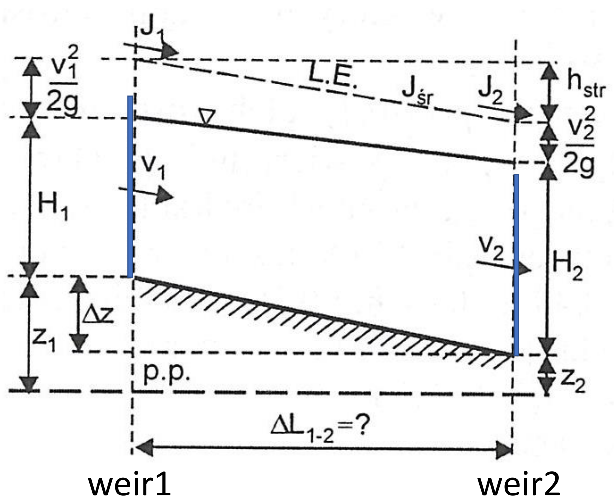


**Figure 2.6.** Scheme for the calculation of the distance between weirs

Knowing that:

$$z_1 - z_2 = i\Delta L$$

$$h_{str} = J_{śr}\Delta L = \frac{1}{2}(J_1 + J_2)\Delta L \qquad \text{(Eq. 2.4)}$$

where:

i – longitudinal slope of the ditch bottom [-].

equation (3) can be transformed in order to calculate the distance between weirs (Fig. 2.6):

$$\Delta L = \frac{H_2 - H_1 + \frac{\alpha\left(v_2^2 - v_1^2\right)}{2g}}{i - \frac{J_1 + J_2}{2}} \qquad \text{(Eq. 2.5)}$$

The slopes occurring in Equation 2.5: J1 and J2 are calculated by Manning formula:

$$J_1 = \frac{V_1^2 n^2}{R_1^{4/3}} \qquad \text{(Eq. 2.6)}$$

The solution of equation 5 requires determination of depth increase *dH* from the value of $H_1$ in the upstream cross-section to $H_2$ in the downstream cross-section (Fig. 2.6).

### *2.1.3 Steady state groundwater flow in the ditch impact zone*

Two schemes were taken into account with respect to groundwater inflow to a ditch: 1) inflow to two parallel, non- fully penetrating ditchs (Fig. 2.2 and 2.7) inflow to a single, non-fully penetrating ditch (Fig. 2.8).

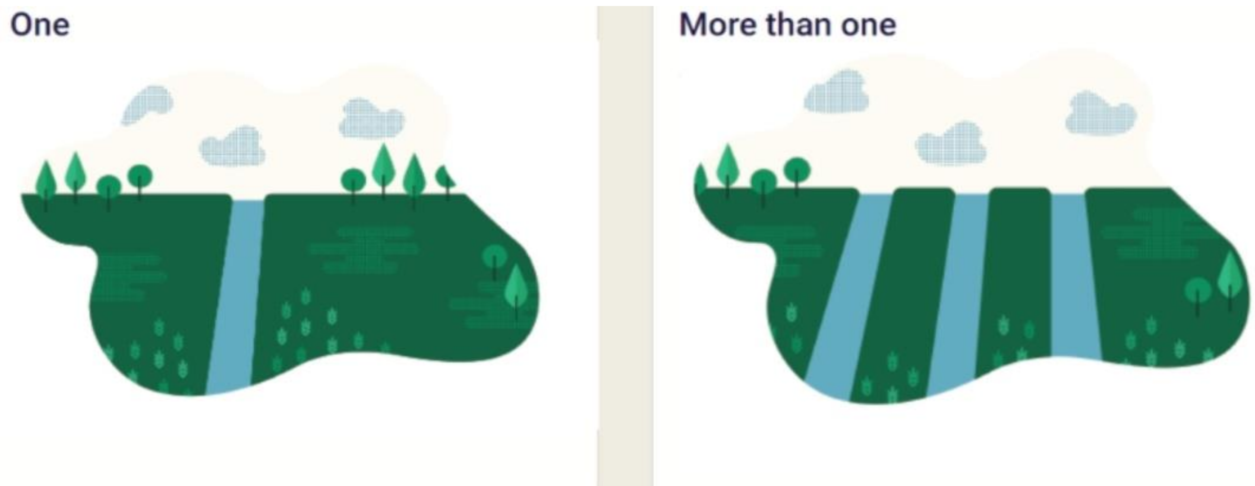


**Figure 2.7.** Scheme of a single ditch and multiple parallel, non -fully penetrating ditches

### *2.1.4 Groundwater inflow to two paralel, non- fully penetrating ditches*

In case of the scheme no. 1) the description of groundwater inflow to a ditch was adopted after Ernst (Ernst, 1956) with relevant assumptions given in Fig. 2.5. The groundwater inflow, according to those assumptions, takes place in a flat, isothropic porous medium consisting of two layers, fed by a constant infiltration of rainfall water. Top layer is formed by peat, underlain by mieral deposits of a hydraulic conductivity equal to $k_2$. The permeability of the lower layer is considerably higher than of the top layer ($k_2>20 \cdot k_1$). Moreover, the summarized thickness of the radial flow layer $D_0$ and the mineral soil layer $D_2$ is not higher than one fourth of ditch spacing (L/4). It is also assumed that the ditch bottom penetrates only to the peat layer (top one). In such a case the loss of hydraulic head is given by a following equation (Ernst, 1956):

$$h = p\frac{a+h}{k_1} + p\frac{L^2}{8(k_1 D_1 + k_2 D_2)} + p\frac{L}{\pi k_1} ln\frac{4D_0}{u} \qquad \text{(Eq. 2.7)}$$

where:

h -total head loss, groundwater table elevation above the dich water level in midspacing [m],

p – groundwater inflow from a unit, drained area, equal to infiltration rate [m/d],

L – ditch spacing [m],

a – ditch water depth [m],

u – ditch wetted perimeter [m],

$k_1$ – hydraulic conductivity of the top layer [m/d],

$k_2$ – hydraulic conductivity of the lower layer [m/d],

$D_1$ – top layer thickness of a horizontal flow direction [m],

$D_2$ – lower layer thickness of a horizontal flow direction [m],

$D_0$ – radial flow layer thickness [m].

Equation 2.7. enables to determine ditch water depth, for which the estimated value of *h* is reached for assumed inflitration *p* and assumed *s* – groundwater depth in midspacing of the ditches.

### *2.1.5 Groundwater inflow to a single, non- fully penetrating ditch*

The case of the inflow to a single ditch is governed by the scheme given in Fig. 2.9.

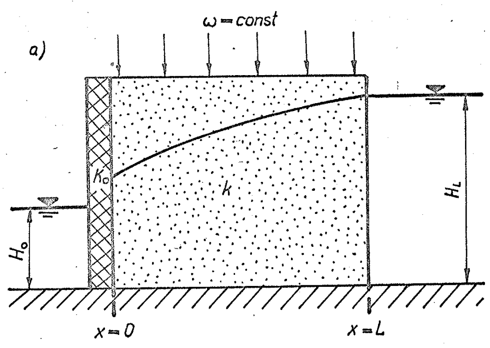


**Figure 2.9.** Groundwater inflow to a single ditch in unconfined conditions.

According to it, the implemenation of the additional resistance $L_{DO}$ helps to represent the non-fully penetrating ditch situation. The groundwater head $H_0$ at the outflow zone (ditch) and $H_L$ at the recharge (inflow) side are described by proper boundary conditions. The distance defined as L stands for the ditch impact range (the distance from the ditch to its' water divide). For such assumptions, the depression curve h(x) and the discharge q(x) are given by following formulas (Polubarinova-Kocina, 1952):

$$h(x) = \sqrt{H_O^2 + (H_L^2 - H_O^2)\frac{x+L_{DO}}{L+L_{DO}} + \frac{\omega}{k}\left(\frac{L^2x+L^2L_{DO}}{L+L_{DO}} - x^2\right)} \quad \text{(Eq. 2.8)}$$

$$q(x) = (H_L^2 - H_O^2)\frac{k}{2(L+L_{DO})} + \omega\left(\frac{L^2}{2(L+L_{DO})} - x\right) \quad \text{(Eq. 2.9)}$$

where:

ω- recharge through the surface [m/d]

$x \in [0, L]$

The ditch impact range L (Eq. 2.10) is calculated after the transformation of (Eq. 2.9) for the condition q(x=L) = 0 at the ditch water divide. The values of $H_0$ (ditch water level before rewetting) and $H_L$ at the water divide are known.

$$L = \sqrt{L_{DO}^2 + \frac{k}{\omega}(H_L^2 - H_0^2)} - L_{DO} \quad \text{(Eq. 2.10)}$$

Groundwater head $H_L$ is approached through a known depth of (sd) from the land surface to the groundwater table at the water divide. The additional resistance $L_{DO}$ introduced by (Eq. 2.8) is calculated as follows (Kostjakov, 1960):

$$L_{DO} = 2 \cdot 0.73 \cdot M \cdot lg\frac{2 \cdot M}{\pi \cdot WP} \quad \text{(Eq. 2.11)}$$

where:

WP – wetted perimitter of the ditch cross-section [m] (Fig. 2.5)

M – the distance from the ditch water table to the top of the impermeable layer [m] (Fig. 2.5).

The required water damming height in a ditch is calculated after the transformation of the formula (10) assuming , that the head value $H_L$ is estimated adequate to the adopted value of the groundwater depth (s) at the water divide after rewetting.

$$H_0 = \sqrt{H_L^2 - \frac{\omega}{k}(L^2 - 2LL_{DO})} \qquad \text{(Eq. 2.12)}$$

The equivalent hydraulic condictivity k, given in Eq. 8-11 and Fig. 2.9) is calculated as the weighted average:

$$k = \frac{m_1 \cdot k_1 + m_2 \cdot k_2}{m_1 + m_2} \qquad \text{(Eq. 2.13)}$$

where:

$m_1$ – peat thickness

$m_2$ – mineral layer thickness

### *2.1.6 Calculation scheme for the case of two parallel ditches*

The above - described components of a steady, gradually varied open channel flow with hydraulic structures (weirs) and groundwater inflow to the parallel, non- fully penetrating ditches impact zone were incorporated into the modeling procedure. The relevant calculation scheme is given in Fig. 2.10.

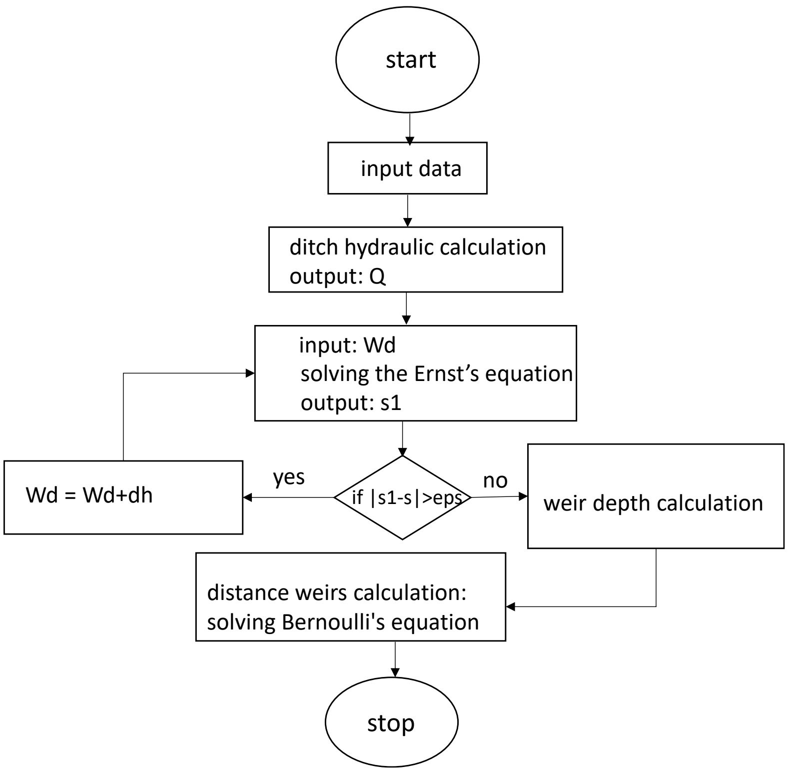


**Figure 2.10.** Calculation scheme (scenario no. 2- parallel ditches)

The calculations follow a relevant algorithm:

Step 1:

Definition of the ditch geometry, groundwater inflow and their hydraulic characteristics

Following data is required:

d – ditch depth [m]

ns – slope of the ditch bank (1:ns)

b – ditch bottom width [m]

i – slope of the ditch bottom [-]

LD – ditch length [m]

Hydraulic characteristics of the ditch bed:

dH – depth increase from $H_1$ to $H_2$ ($dH = H_2 - H_1$) [m]

hzw – ditch depth before rewetting [m]

n – Maninng's roughness coefficient [$m^{-1/3}/s$]

Groundwater inflow characteristics:

L – ditch spacing [m] (the scheme of two, parallel ditches)

$D_2$ – mineral layer thickness [m]

$k_1$ – peat hydaulic condoctivity [m/d]

$k_2$ – mineral layer hydraulic conductivity [m/d]

s – required groundwater depth in midspacing after rewetting [m] (the scheme of two parallel ditches) or required groundwater depth at the water divide (the scheme of a single ditch)

sd – groundwater depth at water divide before rewetting (the scheme of a single ditch)

DH – ditch bottom elevation over the peat layer bottom [m].

Required rainfall inflitration.

Following data is needed:

p – the rate of infiltration [m/d]

Step 2

Basing on Manning's equation (Eq. 2.1) the ditch discharge is calculated.

Step 3

With the use of the Ernst formula (Eq. 2.7) the groundwater head loss and the depth to water table (*s1*) in midspacing for the corresponding ditch depth *hzw* before rewetting are calculated.

Step 4

The condition is verified, if the calculated depth s1 is higher than the one targeted by rewetting (s). If positive, then the ditch water level is increased by dh (assumed as dh=0.01m) and returned to a previous step - number 3. If the condition is not fulfilled (negative) then the ditch water elevation calculated in the prevoius step is assumed as weir damming height (H2).

Step 5

For the known Q (Step 2), basing on Eq. 2.2, water table elevation over the weir crest (parameter h in formula 2) is calculated, and then the weir height (H2-h).

Step 6

Utilizing Eq. 2.5 and 2.6 and the introduced depth increase, the distance between weirs is determined $L_{12}$. The ditch length DL and the konwn vale of $L_{12}$ lead to estimations of weir number ($DL/L_{12}$).

### *2.1.7 Calculation scheme for the case of a single ditch*

The relevant scheme for this case was given in Fig. 2.8 and except for the steps 3 and 4 becomes analogous to the case of two parallel ditches.

Step 3

Basing on the input data the range of depression curve (L) is determined for the assumed depth (sd) to groundwater table at the water divide. Eq. 2.10 is solved and next the LDO through Eq. 2.11.

Step 4

For the required groundwater depth after rewetting (s) the proper water damming level in the ditch is calculated. Equation 2.12 is then solved.

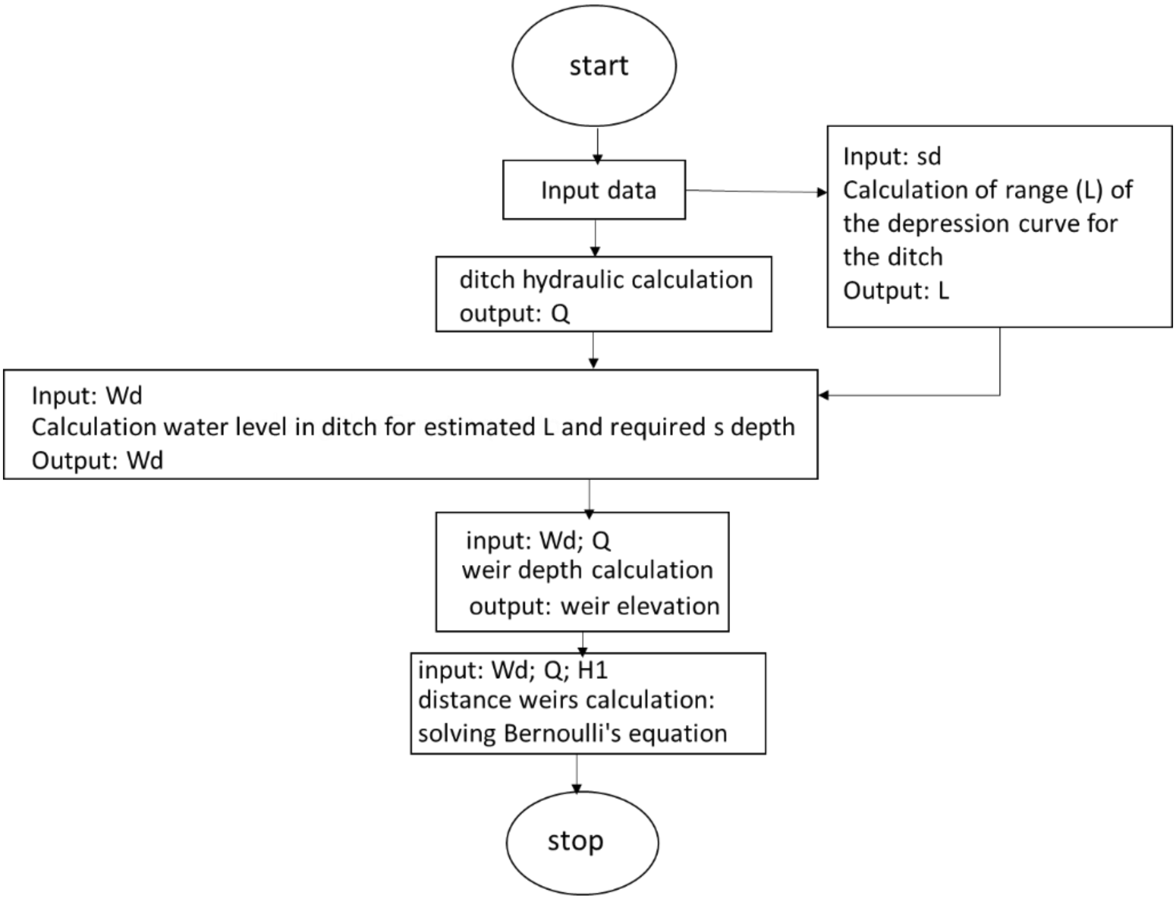


**Figure 2.11.** Calculation scheme (single ditch)

### *2.1.8 Estimation of the volume of water retention as a rewetting result*

The amount of water retention after rewetting utilizes depression curve equation h(x) - number (Eq. 2.8). It is calculated numerically with spatial discretization step dx along the ditch ($dx=L_{12}/N$) and dy ($dy=L/2/M$) from the ditch to midspacing point (the scheme of two parallel ditches). For a single ditch scheme dy = L/M (where L stands for the ditch impact range). N and M are numbers of calculation steps and $L_{12}$ is the distance between weirs determined in step no. 6.

### *2.1.9 Estimation of the value of stored water*

Value of water stored in the rewetted peatland is calculated automatically by the SERVIPEAT module as multiplication of the volume of water stored in the rewetted peatland due to rewetting and the unit value of water storage calculated by Stachowicz et al. (2022) and expressed as 0.53 EUR $\cdot$ $m^3$ $\cdot$ $year^{-1}$.

### *2.1.10 Estimation of costs of ditch blocks*

This estimation must be done outside of the SERVIPEAT module, as costs of ditch block construction strongly depend on the geographic location of planned measures, countries and types of ditch blocks planned. When calculating the approximate costs of ditch blocks construction it is advisable to use the data collected and presented by Stachowicz et al. (2022; Tab. 2.1).

**Table 2.1.** Estimated peatland rewetting costs based on available data from public procurement procedures of peatland rewetting. Source: Stachowicz et al., 2022.

| Country | Location (type of peatland) | Year of action | Type of action | Total cost of one action [EUR] |
|---|---|---|---|---|
| Poland | Słowiński NP (bog) | 2021 | Blocking of a small ditch (+/- 2.0 m) with bags filled with peat and strengthened by wood | 90 |
| Poland | Słowiński NP (bog) | 2021 | Wood-peat block of a small ditch (+/- 2.0 m) | 400 |
| Poland | Słowiński NP (bog) | 2021 | Wooden sheet pile | 1500 |
| Poland | Słowiński NP (bog) | 2019 | Wood-peat block + double sheet pile of a small ditch (+/- 2.0 m) | 1200 |
| Poland | Słowiński NP (bog) | 2019 | Wood-peat block + double sheet pile of a small ditch (+/- 2.0 m) | 1150 |
| Poland | Słowiński NP (bog) | 2019 | Damming spillway of a ditch | 900 |
| Poland | Słowińskie Błota (bog/fen) | 2017 | Damming large ditches (+/- 5.0 m wide) with various types of blocks (averaged value) | 1500 |
| Poland | Bagno Kusowo (bog) | 2017 | Solid wood-peat ditch blocks | 1850 |
| Lithuania | Aukštumala Peatland (bog) | 2016 | Damming drainage ditches<br>1) peat dams (1.0-1.5 m),<br>2) plastic dams (1.0-2.0 m wide, 2 m deep)<br>3) composite dams with water outflow pipe (mixed peat-plastic, geotextile, water tube, elbow for water level regulation, timber logs; 10 m long, 5 wide) | 1) 50<br>2) 80<br>3) 3000 |
| Lithuania | Sachara Peatland (bog) | 2020 | Damming drainage ditches<br>1) peat dams (1,0-2,0 m)<br>2) plastic dams (4-10 m wide, 3 m deep) | 1) 150<br>2) 1580 |
| Lithuania | Žuvintas Biosphere Reserve (fen) | 2021 | Damming hand dug ditches (2 m wide). 1 Dam with culvert (metal pipe) and water level regulation by pulling metal plates 5 m length, 3 m wide | 3630 |
| Belarus | Dziki Nikar (fen) | unknown | Damming drainage ditches with peat dams | 300 |
| Belarus | Dzikoje (fen) | unknown | Damming drainage ditches with peat dams | 430 |
| Belarus | Solomenka (fen) | unknown | Damming drainage ditches with peat dams and wooden dams | 1120 |
| | | | **AVERAGE** | **1114** |

## 2.2 Greenhouse gasses

### *2.2.1 Introduction*

Wetlands in the accumulation phase act as a greenhouse gas (GHG) store through photosynthetic assimilation of carbon dioxide from the atmosphere and sequestration of organic matter (Brix et al. 2001). However, natural, well-hydrated wetlands are also a source of greenhouse gases, mainly methane. Annually, only a dozen or so percent of the net carbon bound by wetlands can be released into the atmosphere as methane. Due to both gases' different infrared absorption characteristics and atmospheric persistence, methane's global warming potential (GWP) is more than twenty times greater by weight than $CO_2$ over a 100-year time scale. The instantaneous carbon balance indicates that while some wetlands function as net sinks of $CO_2$, they still increase the greenhouse effect due to the release of $CH_4$. However, in the longer-term (> 100 years), the impact of $CH_4$ is lower than that of $CO_2$, and wetlands become an effective trap for greenhouse gases. The balance between net $CO_2$ assimilation and $CH_4$ emissions shows whether a wetland can be considered a sink or a net source of greenhouse gases and thus defines the role of wetlands as a regulator of global climate change. A peatland in the accumulation phase increases its carbon pool each year. Still, this process varies in intensity depending on various environmental conditions. In general, peatlands accumulate less carbon in higher latitudes and more in the equatorial zone, and these values range from 10-30 g C m-2 year-1 to even over 200 g C m-2 year-1. The situation changes drastically after the fen is drained. Reducing soil water content increases the activity of microorganisms decomposing organic matter under aerobic conditions, which leads to its mineralization and disappearance and the release of carbon dioxide and nitrous oxide (nitrous oxide) into the atmosphere. Every year, 50.9 Mha of peatlands drained worldwide for forestry, crops, or grassland emit ~ 2 Gt of carbon dioxide due to microbial oxidation of peat, or fires, causing ~ 5% of all anthropogenic greenhouse gas emissions. Indonesia's peatlands release the most carbon dioxide - even more than 500 Mt $CO_2$ per year. European Union peatlands emit ~ 200 Mt $CO_2$/year; Germany is the largest issuer in the EU, followed by Finland, Poland, Ireland, Romania, Sweden, Latvia, Lithuania, and the Netherlands. Polish drained peat soils used for agriculture or forestry are the source of about 34 Mt eq. $CO_2$ and peat mines add 1.9 Mt of eq. to this pool. $CO_2$ (Kotowski 2021). Emission factors (EFs) by IPCC are commonly used to assess the amount of greenhouse gas emissions from wetlands. Regrettably, the EFs are sometimes too generalized and do not always reflect the subtleties and differences in $CO_2$ emissions related, for example, to the drainage depth of a peat complex (Tiemeyer et al., 2019). GHG emission is not simply linked to the depth of drainage, the depth of peat, and soil organic matter content. A review of the literature on emissions from organic soils provides various, often contradictory, data. E.g., Jurczuk (2011) indicated an almost linear relationship between the depth to water in the range of 30-70 cm and the amount of soil organic mass mineralization, and Renger et al. (2002) demonstrated that a drop in the water level from 0.3 to 0.8 m has doubled the emissions. However, searching for general patterns, Tiemeyer et al. (2019) concluded that lowering the groundwater level below 0.40-0.60 m does not always involve a drastic increase in $CO_2$ emissions. Often, the intensity of the C loss only slightly increases or even stabilizes by the depth of 60-70 cm (Mundel 1976). A decrease in water level is necessary but not sufficient for the linear increase in GHG flux. The amount of carbon dioxide emissions results directly from the presence of decomposing organic matter. However, the organic matter content in soil is not a crucial factor. Soils with SOC content ~ 5% can emit as much GHGs as histosols (Tiemeyer et al. 2016).

The emission should be linked to habitat fertility (although there are also many contradictory data here). The intensity of soil respiration conditioned by fertility, pH, nitrate content, and peat decomposition rate has been shown to affect soil respiration (Norberg et al., 2018). Despite all the reservations and uncertainties, we accepted a mean annual water level as a sufficient explanatory variable for GHG fluxes (e.g., Couwenberg et al., 2011). While the generalized statistical relationship between groundwater level and GHGs emission is highly uncertain at the small scale, it can be considered robust at the regional scale.

### *2.2.2 Carbon dioxide and methane*

We used a non-linear Gompertz response functions for emission of C-$CO_2$ (t C $ha^{-1}year^{-1}$) from drained organic soils (< −0.1 m) in relation to depth to groundwater table (WT) as proposed by (Tiemeyer et al, 2020). In the formula:

$$CO_2\text{-}C(WT) = CO_2\text{-}C_{min} + CO_2\text{-}C_{diff}\, e^{-ae^{bWT}} \qquad \text{(Eq. 2.13)}$$

$CO_2$-$C_{min}$ is the lower asymptote, $CO_2$-$C_{diff}$ the difference between upper and lower asymptote, while a and b are fitting parameters. The coefficients were adopted from the comprehensive modeling of the German national GHG inventory:

$$CO_2\text{-}C_{min} = -0.93\text{t C ha}^{-1}\text{ yr}^{-1},$$

$$CO_2\text{-}C_{diff} = 11.00\text{t C ha}^{-1}\text{ yr}^{-1},$$

$$a = 7.52 \text{ and } b = 12.97\text{ m}^{-1} \qquad \text{(Eq. 2.14)}$$

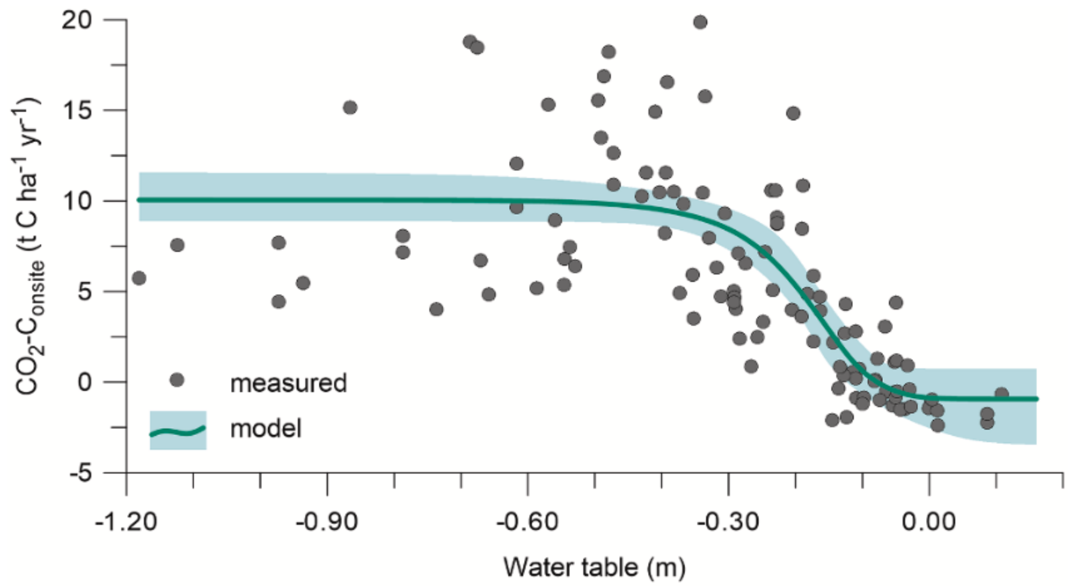


**Figure 2.12.** Response of on-site $CO_2$-C emissions from organic soils to mean annual water table and coefficients of the fitted Gompertz function with the parameter values as listed in Eq. 2.14. Source: TBA.

Deep drained organic soils do not emit methane or they are even a small sinks of $CH_4$. $CH_4$ emission begins at a groundwater depth of around −0.2 m, and progress either linearly or exponentially along with the rising GW level.

$$CH_4(WT) = CH_{4min} + ce^{-dWT} \quad \text{(Eq. 2.15)}$$

For the groundwater depth in the range between -0.2 and -0.1 m, the function parameters for various land use categories are as of the Tiemeyer (Tab. 2.3).

**Table 2.3.** Coefficients of the $CH_{4\ land}$ response functions. Source: TBA.

| Land-use category | $CH_{4\ min}$ (kg $CH_4$ $ha^{-1}$ $yr^{-1}$) | c (–) | d ($m^{-1}$) |
|---|---|---|---|
| Forest land | −2.9 | 2260 | −31.3 |
| Cropland, grassland, settlement | 3.5 | 17,055 | −42.3 |
| Drained unutilized land, rewetted organic soils | 1.3* | 292 | −5.6 |

* Fixed at the mean value of all measurements with WT < −0.3 m.

**Table 2.4.** Emission factors for emissions from ditches used in German GHG inventory. Source: TBA.

| Land-use category | $CH_{4\ ditch}$ (kg $CH_4$ $ha^{-1}$ $yr^{-1}$) | Comment |
|---|---|---|
| Forest land | 217 | IPCC, 2014 |
| Cropland | 1165 | IPCC, 2014 |
| Grassland, settlement | 948 | Weighted average of shallow and deep drained grassland (IPCC, 2014) with 34% shallow drained grassland (Bechtold et al., 2014) |
| Drained unutilized land | 217 | as Forest land (no fertilization) |
| Peat extraction | 542 | IPCC, 2014 |
| Rewetted organic soils | – | Refilled and blocked ditches assumed to be part of the landscape mosaic (IPCC, 2014) |

$CH_4$ emissions from ditches (before renaturation) must be taken into account in the calculation.

## 2.3 Assessing the biochemical effects of wetland restoration

### *2.3.1 Soil indicators*

Effective soil quality indicators are attributes that allow for a relatively accurate assessment of changes in the soil environment under the influence of human activity (Fennessy and Wadrop, 2016). A good and robust indicator must possess the following features:

- It is relatively easy and inexpensive to measure.

- It is sensitive to environmental changes caused by disturbances and anthropogenic stress.

- It reflects well the intensity of anxiety or disturbances.

- It reflects well and is related to ecological processes.

- It is not susceptible to seasonal changes.

The "three-tier framework" approach organizes indicators hierarchically according to the effort they require. Indicators range from cheap, routine measurements to intensive chemical and biochemical tests.

Level 1 - quick and straightforward indicators to assess the condition

This is basic information about the physical and chemical properties of soils, easy to measure and interpret. These include pH, bulk density, electrical conductivity, total organic carbon, nitrogen, phosphorus, C: N: P ratio, extractable nutrients, etc.

Level 2 - moderate-intensity indicators

They require more advanced field and laboratory methods, have greater sensitivity, and provide better insight into disturbed ecosystem processes and functions. Many indicators have been developed for P enrichment evaluation. These include, for example, extractable and porewater nutrients, measurements of the biomass of microorganisms, and the content of C, N, P in the biomass of microbes. An interesting approach is assessing soil saturation by P, fractionation of phosphorus compounds, and determining the amount of easily digestible P, which can be released after restoration (rewetting) and, consequently, changing redox conditions.

Level 3 - intensive indicators.

They are based on detailed biological and chemical information. They refer to the composition of microorganisms, the rate of microbiological processes, enzymatic activity, soil and C accretion rates using the Cs-137 isotope, etc.

Soil indicators are an essential component of restoration and mitigation activities. Easily measurable parameters are beneficial, which provide insight into the ecological status of a restored wetland and assess the transformation of nutrients and conditions for the development of plant communities. Such indicators may be soil organic C and N, bulk density (Hossler et al., 2011), pH. The bulk density can be an integrated measure of the organic C content and porosity and is related to biochemical processes such as denitrification, plant biomass production, and microbiological activity. Soil C and N show strong relationships with a whole range of diverse and sometimes difficult-to-measure ecosystem processes (Fennessy and Wadrop, 2016).

Looking for a simple indicator that could be useful for assessing changes in wetland habitats following rehydration, we focused on pH and the C to N ratio.

These two simple-to-perform measures turned out to quite accurately reflect the trophic changes that will occur in histosols after their rewetting.

### *2.3.2 Threat from internal eutrophication caused by the release of readily available P from soils*

Rewetting peatlands was found to support the sequestration of organic carbon and restore buffer zones as primary measures to mitigate non-point agricultural pollution loads. Wetland buffer zones were proven to have high and long-term capacity to reduce nitrate fluxes. However, altered rewetted peatlands can release dissolved substances, resulting in severe biogeochemical constraints on achieving restoration goals (Cabezas et al., 2013). Unfortunately, it is widely recognized that the rewetting of wetlands on former agricultural land: grasslands and croplands, could potentially result in the release of soil phosphorus and become a cause of eutrophication of soil, groundwater, and adjacent watercourses (Smolders et al. 2006; Banaszuk et al. 2011). After rewetting, phosphorus concentrations in pore-water were up to three orders of magnitude greater than under pristine conditions (Zak et al., 2008).

Thus, we can expect an increased pollution impulse that can last for years instead of the expected water quality improvement. However, estimating the duration of P release is complex, and the reoxidation and readsorption of P at the redox boundary should be critical environmental factors in mitigating the water pollution problem (Banaszuk et al. 2016).

The long-term agricultural use of histosols may result in total phosphorus (TP) accumulation, which in the top 50 cm layer of soils can range from 50 to over 300 g P $m^{-2}$. A significant part of phosphorus occurs in compounds with metal oxides as redox-sensitive phosphorus characterized by variable dynamics. Soil anoxia that will arise in the peat after rewetting at low oxygen supply can cause dissolution of reductive Fe (III) compounds, leading to a high discharge rate of Fe (II) and P. As a result, a significant release of P is expected, which can amount up to 6 g P $m^{-2}$ (the so-called NH4Cl_P + BD_P fractions), followed by severe pollution of ground-

and surface waters. In addition, soil eutrophication can support the development of fast-growing generalists, mainly Phragmites australis and Typha sp., instead of desired plant composition targeted by restoration planners (Kreyling et al. 2021). The restoration may create a novel ecosystem with no past analog far from an "ideally reconstructed" ecosystem, entirely referring to its historical predecessors. However, they may provide some ecosystem services comparable to natural mires. In addition, the benchmark for landscape restoration depends on the timeframe used as a reference point (Manton et al., 2020).
Elevated release of P and its export to adjacent watercourses is expected for nutrient-rich rewetted peatlands whose upper soil layers are built of highly decomposed peat where molar Fe:P ratios are less than 10 (Zak et al. 2010).
Lab analyses revealed a clear relationship between habitat pH and the C: N ratio and the amount of redox-sensitive phosphorus. Fluviogenous mires with pH close to neutral and lower C: N ratio have a substantially higher pool of mobile phosphorus than acid, raised bogs. **Thus, two parameters C: N and pH, could be successfully used as a proxy to assess the phosphorus eutrophication potential of rewetted peatlands.**
It is much easier and cheaper to measure the pH value of histosols than the C: N ratio of organic matter. For the purposes of this manual, developed for practitioners, potential phosphorus flux is calculated from the pH of the soil. The relationship was developed for peatland, situated in lake basins and river valleys (Fig. 2.13).

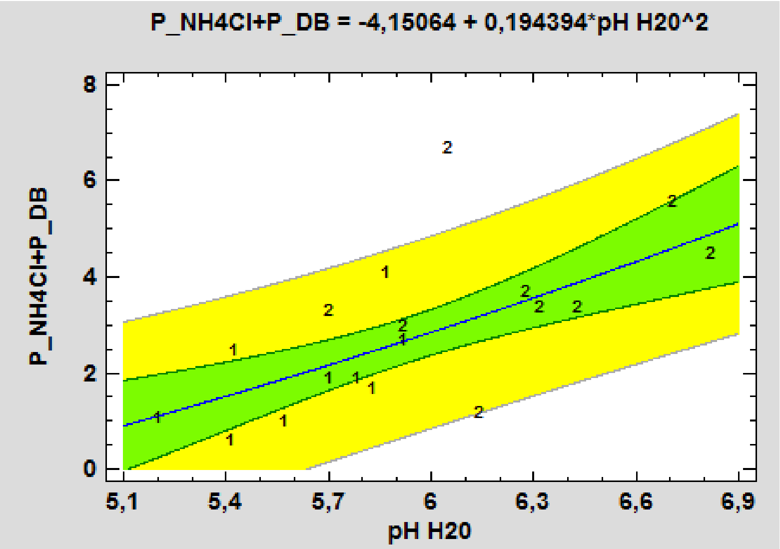


**Figure 2.13.** Relation between the emission of total reactive phosphorus and pH of the topsoil. Source: TBA.

The equation of the fitted model used in the ServiPeat approach is:

$$P\_NH4Cl+P\_DB = -4{,}15064 + 0{,}194394*pH_{H20}\hat{}2 \qquad \text{(Eq. 2.16)}$$

### 2.3.3 *Release of N-$NO_3$ from rewetted wetlands*

The transformation of the organic matter in the top layer of over-dried histosols leads to a significant transformation of soil nitrogen. One of the effects is the mineralization and the release of nitrogen compounds that (especially nitrate nitrogen) can be present in soils in significant amounts. Wetland restoration can lead to the substantial release of nitrates. The amount of nitrogen depends on many factors: the type of peat and its decomposition rate, its pH, the intensity of previous agricultural use (fertilization), etc. (Ilnicki and Szajdak, 2016). During the year, the amount of N mineralized may range from 140 to over 350 kg / ha in a 20 cm layer of soil. Nitrogen release correlates with the drainage depth; Frąckowiak (1980) found mineralization decrease with the depth of soil drainage. It seems that the pH of upper layer of drained histosol (with an annual average groundwater depth ~50-60 cm) is a good measure of the amount of nitrate nitrogen released after rewetting.

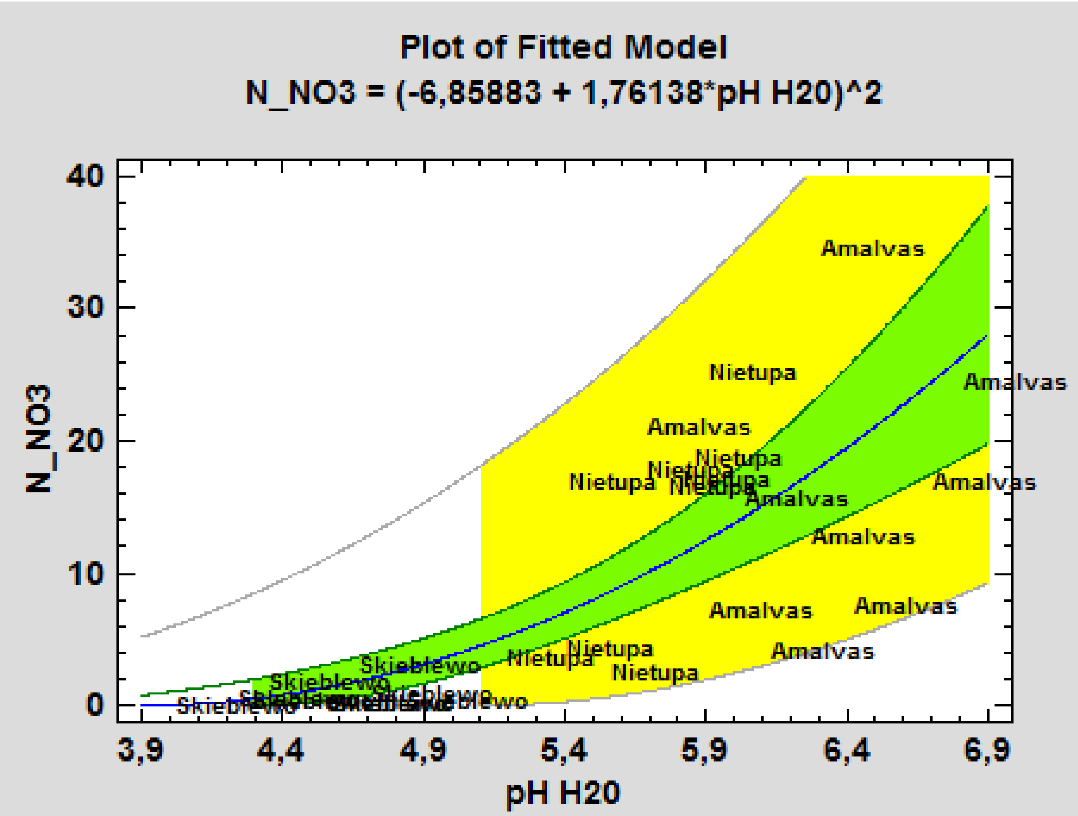


**Figure 2.14.** Relation between the emission of $N\_NO_3$ and pH of the topsoil. Source: TBA

Relationship between the amount of mobile N-NO3 **(g / m2)** and $\mathbf{pH_{H2O}}$ used in the ServiPeat approach is based on the Eq. 2.17:

$$N\_NO_3\ (g/m^2) = (-6{,}85883 + 1{,}76138 * pH\ H_20)^2 \qquad \text{(Eq. 2.17)}$$

The released nitrogen is taken up by plants and microorganisms, a significant part of it turns into a gas as a result of denitrification. Only a certain amount of nitrogen can be leached from soil and contaminate surface and groundwater. To assess the amount of mineral nitrogen losses from soils, we adopted the approach and data presented by Joosten et al. (2015), which assumes strongly simplified nearly linear relationship between depth to groundwater table and mean annual N los from 1 hectare of peatland. This simplification ensures that the calculated size is strongly "conservative" and overestimated. N release varies between 20 kg N $ha^{-1}$ $y^{-1}$ in grassland with average annual water table -50 cm below surface, to 5 kg N kg N $ha^{-1}$ $y^{-1}$ for tall reed with groundwater -5 cm below surface. In case of using a wetland as a pasture (annual g.w. depth -10 cm) an additional 5 kg of N release must be considered. Moreover, in sites with groundwater discharge (“upwelling”) a release of N can be higher by ~ 20 kg $ha^{-1}$ $y^{-1}$ of N because of higher throughflow rates.

### *2.3.4 Relationship between depth to water (in meters) and mean annual nitrogen leaching*

export = sqrt(-37,8231 - 1324,46*depth)

depth = depth to groundwater in meters with the sign "-" (with negative values) e.g. -0,5 m

export = kg N $ha^{-1}$ $y^{-1}$

## 3. Acknowledgements

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