# Phosphorus accumulation

in a free water surface wetland
discharging into the Baltic Sea

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Master thesis in Environmental Science • 30 ECTS
Swedish University of Agricultural Sciences, SLU
Department of Aquatic Sciences and Assessment
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# Phosphorus accumulation in a free water surface wetland discharging into the Baltic Sea

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Abstract

Eutrophication has been an issue of the Baltic Sea for a long time, causing decreased water clarity due to enhanced growth of algae and oxygen depletion during the decomposition of algae. The study investigated the free water surface wetland Vilhelmsberg that treats agricultural runoff before the water reaches the Baltic Sea. The area of the wetland Vilhelmsberg (1.6 ha) corresponds to 0.1 % of its catchment area, in which the natural waterbody Maren is located upstream of the wetland. The main objectives of this study were to quantify the phosphorus (P) accumulation, evaluate the possibility of recycling the sediment and assess the risk of internal loading. Therefore, sediment cores were collected in both waterbodies and analysed for (i) total P and P fractions, (ii) metal content, (iii) plant available P and (iv) particle size distribution. Results showed that the wetland Vilhelmsberg had a low sediment accumulation rate of 0.5 cm $yr^{-1}$ and a low P accumulation of 13 kg $ha^{-1}$ $yr^{-1}$. The maximum potential P release rate of the wetland Vilhelmsberg (6.5 mg $m^{-2}$ $d^{-1}$) and Maren (5.2 mg $m^{-2}$ $d^{-1}$) indicated a small risk of internal loading. However, the proportion of clay particles at the wetland's inlet was eight times higher than at the outlet, indicating resuspension probably caused by the inflowing Sea water. Thus, it is recommended to change the outlet design to improve the functioning of the wetland. Results also showed that it will be possible to return the sediments of the wetland Vilhelmsberg to the agricultural fields since the metal content was below Swedish legislation limits, whereas the sediment from Maren had too high nickel contents. The sediment represented a good soil amendment regarding the plant available P which was much higher in the wetland (10.5 mg per 100 g DS) and Maren (12.7 mg per 100 g DS), compared to the catchment soil (3.8 mg per 100 g DS). Concluding, the wetland Vilhelmsberg functions as a P trap by decreasing the P load from agricultural land to the Baltic Sea, but it had low particle and P accumulation which could be improved by design changes.

***Keywords***: *free water surface wetland, phosphorus accumulation, internal loading, sediment accumulation*

# Swedish summary

Övergödning har länge varit ett problem för Östersjön som orsakat minskat vattnets siktdjup på grund av ökad algtillväxt och syrebrist till följd av nedbrytningen av alger. Den här studien undersökte en anlagda våtmark vid Vilhelmsberg som renar vatten från jordbruksmark innan det mynnar i Östersjön. Våtmarksytan (1,6 ha) motsvarar 0,1 % av avrinningsområdet, inom vilken den naturliga vattenytan Maren ligger några hundra meter uppströms våtmarken. Syftet med studien var att kvantifiera fosforackumulationen, utvärdera möjligheten att återanvända sedimentet och uppskatta risken för internbelastning av fosfor. Därför togs sedimentproppar i både våtmarken och Maren som analyserades med avseende på (i) totalfosfor och fosforfraktioner, (ii) metaller (iii) växttillgänglig fosfor och (iv) partikelstorleksfördelning. Resultaten visade både låg sediment ackumulationshastighet 0,5 cm per år och låg fosforackumulation 13 kg per ha våtmarksyta och år i Vilmhelmsberg våtmark. Den maximala potentiella fosforfrigörelsehastigheten i Vilmhelmsbergs våtmark (6,5 mg $m^{-2}$ $d^{-1}$) och Maren (5,2 mg $m^{-2}$ $d^{-1}$) indikerade låg risk för internbelastning av fosfor. Däremot, var proportionen lerpartiklar vid våtmarkens inlopp åtta gånger högre än vid utloppet, vilket indikerar resuspension förmodligen orsakat av inflödande vatten från Östersjön. Det är därmed rekommenderat att ändra utformningen av utloppet för att förbättra våtmarkens funktion. Resultaten visar också att det är möjligt att återföra sedimentet från Vilmhelmsberg våtmark till åkermarken, då metallhalterna låg under de svenska gränsvärdena. Däremot var nickelhalten för hög i Marens sediment. Sedimentet representerar ett bra jordförbättringsmedel med avseende på växttillgänglig fosfor som var mycket högre i våtmarken (10,5 mg per 100 g) och Maren (12,7 mg per 100 g), jämfört med medelvärdet i uppströms jordbruksmark (3,8 mg per 100 g). Sammanfattningsvis fungerade Vilhelmsberg våtmark som en fosforfälla genom att minska belastningen från jordbruksmark till Östersjön, men den hade låg partikel- och fosforretention som skulle kunna öka genom ändrad utformning.

# Popular science summary


Phosphorus is a plant nutrient that can contribute to eutrophication of waterbodies causing growth of harmful algae. Eutrophication is a long-lasting problem of the Baltic Sea. In agricultural runoff, phosphorus is mainly bound to soil particles which could potentially be captured before the water flows into the Baltic Sea. Sweden tries to capture particles for instance by constructing wetlands. Many different wetlands with various sizes, shapes and positions exist in Sweden, but only few of them are studied. This master thesis contributed to fill the knowledge gap by investigating one specific wetland with a large size, a meandering shape and a unique position directly at the Baltic Sea. The main objective of this master thesis was to assess whether the wetland Vilhelmsberg functions as a phosphorus trap. Further objectives were to estimate the risk that phosphorus gets released again from the sediment into the water and the possibility to return the sediment to agricultural fields.

The wetland Vilhelmsberg was constructed 6.5 years ago downstream of the natural waterbody Maren and with an open outlet towards the Baltic Sea. It has a large size of 1.6 ha, which corresponds to 0.1 % of the area from where water flows towards the wetland from land. For assessing the functioning, sediment samples were collected at various locations in the wetland Vilhelmsberg and Maren. The samples were analysed for the phosphorus concentration, metal content, plant available phosphorus and sizes of the soil particles.

Results showed that the accumulation of particles (16 t $ha^{-1}$ $yr^{-1}$) and phosphorus (13 kg $ha^{-1}$ $yr^{-1}$) was amongst the lowest of other Swedish wetlands. A main reason for the low accumulation probably is that there is no barrier to the Baltic Sea which can lead to additional water inflow and resuspension of settled particles. This was indicated by the results of the particle sizes of the sediment. It was found that the proportion of fine particles, which are more prone to resuspension, at the wetland's outlet was eight times smaller than at the inlet. Another main reason is that the wetland Vilhelmsberg is located downstream of the natural waterbody Maren which probably retained particles and attached P that the wetland would have received.

Furthermore, results showed that the risk that bound phosphorus will be released from the sediment into the water under certain environmental conditions was quite similar for the wetland Vilhelmsberg (6.5 mg $m^{-2}$ $d^{-1}$) and Maren (5.2 mg $m^{-2}$ $d^{-1}$).

The risks were small compared to other Swedish wetlands and corresponded to typical release rates from coastal sediment in the Baltic Sea.

There is no current need of sediment removal, due to the low particle accumulation and the small risk of phosphorus release from the sediment into the water. The plant available phosphorus in the sediment of both the wetland Vilhelmsberg and Maren was high and categorized two classes higher than the soil of agricultural fields close to them. Moreover, it will be allowed to transfer the wetland's sediment for agricultural purposes because it was below the legal limits for metal contents, whereas Maren had too high nickel contents. Hence, it is recommended to return the wetland's sediment to agricultural fields in the catchment. This will contribute to a sustainable phosphorus management.

Concluding, the wetland Vilhelmsberg functions as a phosphorus trap and helps to decrease the phosphorus input into the Baltic Sea. However, it has low particle and P accumulation which can be improved by design changes.

# Preface

From all wetlands that I have visited, the wetland of this thesis impressed me the most with its wonderful view to the Baltic Sea. It was a real pleasure for me collecting samples there and writing this thesis about it!

Many thanks to my supervisor Pia Geranmayeh for the valuable advice and quick responses. Many thanks to Hans Lindholm for sharing his local knowledge with me. Thanks to all my study colleagues for discussing and sharing my interest about nutrient recycling with you.

This thesis will always remind me on the lovely time I have spent abroad in Sweden, to all the friendships I have made and the sites I have visited.

*"All water has a perfect memory*
*and is forever trying to get back to where it was."*

*Toni Morrison*

# Table of contents


**1. Introduction ... 1**

1.1. Eutrophication ... 1
1.2. Free water surface wetlands ... 2
1.2.1. FWS wetlands for P retention ... 3
1.2.2. Factors influencing P retention ... 3
1.3. Objectives ... 5

**2. Material and methods ... 6**

2.1. Site description ... 6
2.1.1. Wetland Vilhelmsberg ... 6
2.1.2. Natural waterbody Maren ... 8
2.2. Catchment description ... 8
2.3. Estimations of loads and theoretical retention ... 10
2.3.1. Hydraulic load ... 10
2.3.2. Theoretical P load and retention ... 11
2.4. Sediment sampling and analyses ... 13
2.4.1. Sampling procedure ... 13
2.4.2. Laboratory analyses ... 14
2.4.3. Estimations of accumulation and P release rate ... 15
2.4.4. Estimation of allowed sewage sludge application ... 16
2.5. Work in ArcMap ... 16
2.5.1. Estimation of wetland's characteristics ... 17
2.5.2. Catchment delineation ... 17
2.6. Soil survey of arable land ... 18
2.7. Statistical analyses ... 18

**3. Results ... 20**

3.1. Sediment characterisation ... 20
3.1.1. Sediment accumulation rate ... 21
3.1.2. Water content, organic matter and P concentration ... 21
3.1.3. Particle size distribution ... 22
3.2. Accumulation of particles and P ... 23
3.2.1. P fractions ... 24
3.2.2. Internal loading ... 28
3.3. Qualification of the sediments for recycling ... 30
3.3.1. Metal content ... 30
3.3.2. Soil amendment ... 31
3.4. Theoretical P load and retention ... 33
3.4.1. Theoretical P load from land ... 33
3.4.2. Theoretical P load from domestic wastewater ... 33

3.4.3. Theoretical P retention ... 35

**4. Discussion ... 37**

4.1. Accumulation of particles and P ... 37
4.2. Internal loading ... 39
4.3. Maintenance need ... 41
4.3.1. Metal content ... 42
4.3.2. Soil amendment ... 44
4.4. Impact of domestic wastewater ... 45
4.5. Method uncertainties ... 46
4.5.1. Sediment depth ... 46
4.5.2. Catchment area and P load ... 47

**5. Conclusion ... 49**

**6. Recommendations and further research ... 50**

**References ... 51**

**Appendix ... 57**

# Abbreviations

| | |
|---|---|
| Ac | area of the catchment |
| Al-P | aluminium bound phosphorus |
| As | arsenic |
| Aw | area of the wetland or area of Maren |
| Aw:Ac | area of the wetland or area of Maren relative to its catchment area |
| Ba | barium |
| BW | blackwater |
| C | carbon |
| Ca-P | calcium bound phosphorus |
| Cd | cadmium |
| Co | cobalt |
| Cr | chromium |
| Cu | copper |
| DS | dry substance |
| Fe-P | iron bound phosphorus |
| FWS | free water surface |
| GW | greywater |
| Hg | mercury |
| HL | hydraulic load |
| L:W | length-to-width ratio |
| N | nitrogen |
| Ni | nickel |
| OM | organic matter |
| Org-P | organically bound phosphorus |
| P | phosphorus |
| P-AL | plant available phosphorus |
| Pb | lead |
| PE | population equivalent |
| PW-P | phosphorus in pore water |
| Q | discharge |

| | |
|---|---|
| SGU | Geological Survey of Sweden<br>(in Swedish: Sveriges geologiska undersökning) |
| SLU | Swedish University of Agricultural Sciences |
| SMHI | Swedish Meteorological and Hydrological Institute<br>(in Swedish: Sveriges Meteorologiska och Hydrologiska Institute) |
| SwAM | Swedish Agency for Marine and Water Management<br>(in Swedish: Havs- och vattenmyndigheten) |
| Swedish EPA | Swedish Environmental Protection Agency<br>(in Swedish: Naturvårdsverket) |
| TC | total carbon |
| TN | total nitrogen |
| TP | total phosphorus |
| V | vanadium |
| WW | wastewater |
| WWTP | wastewater treatment plant |
| Zn | zink |

# 1. Introduction

Phosphorus (P) is a finite resource and essential for life and the production of food (Withers et al. 2015). Worldwide, extracted P is to a great extend (95%) used for fertilizer and animal feed, and minor parts (5%) are used in industry (Hallberg & Reginiussen 2019). It is necessary to use P reserves sustainably and enhance recycling of P to ensure the need of P for future generations. Worldwide, flow of nutrients and sediments increased due to changes in land cover, higher amounts of livestock manure and applied fertilizer (Jordan et al. 2003). One of the most severe water quality issues, eutrophication of watercourses, is caused partly by P pollution (Gunes et al. 2012).

## 1.1. Eutrophication

Excessive input of plant nutrients like P and nitrogen (N) into aquatic ecosystems lead to increased primary production of algae and cyanobacteria, which can have severe consequences on the aquatic ecosystem (Boesch et al. 2006; Chislock et al. 2013; HELCOM 2018a). Algal blooms can produce toxins that are harmful to other organisms and an augmented growth of algae decreases water clarity (Chislock et al. 2013). Due to the enhanced decomposition of organic material oxygen gets consumed, resulting in oxygen depletion, which can affect species composition and cause animal death (Rathore et al. 2016; HELCOM 2018a).

Eutrophication is a long-lasting issue of the Baltic Sea (HELCOM 2018a). Over the last 50 to 100 years, the Baltic Sea has changed from an oligotrophic into a eutrophic sea while some parts became hypoxic (Conley et al. 2009; Andersen et al. 2017). The need to decrease nutrient input into aquatic ecosystems at European level gets addressed for instance with the Water Framework Directive (Land et al. 2016). Moreover, the countries surrounding the Baltic Sea are part of the Baltic Marine Environment Protection Commission – Helsinki Commission (HELCOM) that strives towards a healthy Baltic Sea environment. With the Baltic Sea Action Plan (BSAP) the countries have defined concrete actions to achieve a good environmental status by 2021 (HELCOM 2007). Even though the goal has not been fully achieved, the nutrient input for P and N has already been reduced and the member states work on an updated version of the BSAP (HELCOM 2018a, 2021a). Further P reductions of the P input are required because the concentration of P in the Baltic Sea is stagnant, whereas the concentration of N decreased in most parts

(HELCOM 2018a). This is due to a long residence time of P in the Baltic Sea (around 30 to 50 years), until it gets flushed out into the North Sea or buried into sediment (Gustafsson et al. 2017). Whereas dissolved inorganic N can get removed through transformation into its gaseous phase leading to a comparatively shorter residence time (10 years) (ibid.).

The sources of P reaching the Baltic Proper from its surrounding countries are prevailingly riverine (97 %), besides direct inputs (3 %) (HELCOM 2018b). Within the riverine P sources to the Baltic Proper, diffuse-sources (mainly from agricultural activities) have the biggest contribution (38 %), followed by point sources (33 %) and natural background loads (20 %) (ibid.). Since in Sweden most wastewater treatment plants (WWTP) already focus on P removal, diffuse sources are of great importance (Boesch et al. 2006). According to the seventh Baltic Sea Pollution Load Compilation, agricultural activities amount for more than half of the Swedish P load to the Baltic Proper (Hansson et al. 2019). Hence, it is important to reduce the nutrient losses from agriculture, especially clay soils that generally have high P losses (Ulén et al. 2007). Possible ways to reduce nutrient losses from agriculture are for instance improving fertilizer, manure and soil management, shifting from intensive to extensive agriculture, establishing buffer zones and restore or construct wetlands (HELCOM 2007; Ulén et al. 2007).

## 1.2. Free water surface wetlands

Since the 1990s Sweden restored and created wetlands with the initial focus on enhancing biodiversity and reducing N, because many years N was assumed to be the most limiting nutrient for primary production (Land et al. 2016). However, both N and P are essential for combating eutrophication and often P is the most limiting nutrient in fresh and brackish water (Boesch et al. 2006; Conley et al. 2009). Over the past years many free water surface (FWS) wetlands with the aim of P retention have been created in Sweden to reach the national environmental goals “zero eutrophication” and “thriving wetlands” (Land et al. 2016)

FWS wetlands are also known as surface flow treatment wetlands and can be distinguished from subsurface flow treatment wetlands (Fonder & Headley 2010; Dotro et al. 2017). The latter is often used for domestic wastewater and the water usually flows either horizontally or vertically through a porous media where the major treatment occurs (Fonder & Headley 2010). Contrary, FWS wetlands are similar to natural wetlands and mostly used for storm water from agricultural, industrial or urban areas due to their capacity to deal with different water amounts and levels (ibid.). Besides nutrient retention FWS wetlands also have benefits for wildlife and recreation (Fonder & Headley 2010; Koskiaho et al. 2003)

### 1.2.1. FWS wetlands for P retention

Soil particles with associated P from land can be transported away through tile drains or surface runoff caused by water erosion (Sveistrup et al. 2008; Djodjic et al. 2018). Clay particles compared to silt or sand have a large specific surface area, which means that more nutrients can be attached to them (Sveistrup et al. 2008). In agricultural runoff from clay or silty soils, P is mainly transported in particulate form and sedimentation of particles with associated P is considered to be an essential retention process for P in FWS wetlands (Ulén 2004; Weisner et al. 2016). Especially in boreal regions biological activity is small and FWS wetlands primarily act as a sedimentation basin (Koskiaho et al. 2003). Hence, FWS wetlands with the purpose of P treatment from agricultural runoff operate in every climate, also in cold regions (Fonder & Headley 2010). Contrary to agricultural runoff, P in treated domestic wastewater mainly is in dissolved form. If large parts of the P draining into wetlands are in dissolved form, less retention can be expected (Johannesson et al. 2015). Hence, it is important to estimate the share of domestic wastewater on the entire P load that a wetland receives.

Unlike N removal, P removal is not permanent because there is no significant transformation into the gaseous phase and therefore P rather accumulates and gets stored in the wetland sediment (Kadlec & Reddy 2001). FWS wetlands serve as P storages over long-term by accretion of sediments and short-term by uptake through biota and sorption to wetlands soil and sediment (Kadlec 2005; Di Luca et al. 2017). Most of the P that is assimilated through vegetation gets released again during its decomposition (Kadlec 2005). Depending on the potential availability of the bound P and the environmental conditions, stored P can get mobilized again and contribute to internal loading that enables a transport of P further downstream and prevents water quality improvements (Lannergård et al. 2020).

### 1.2.2. Factors influencing P retention

The main factors influencing particle and P retention are the concentration and size of particles as well as the shape, size and hydraulic load (HL) of the wetland (Stephan et al. 2005). In theory, smaller particles need more time to settle than bigger particles, however, clay particles are often aggregated and settling velocities can be similar to silt (Braskerud 2003). In order to optimize sedimentation processes it is important to ensure low water flow velocities and long water residence times (Koskiaho et al. 2003). Sedimentation is not everlasting and particles are prone to resuspension due to turbulence caused by wind, waves, bioturbation or high-flow conditions (e.g. during snow melting or extreme precipitation events) (Geranmayeh et al. 2018). Resuspension of settled particles also depends on the particle size, variations in water flow velocities and the presence of vegetation (Braskerud 2001; Geranmayeh et al. 2018). For instance, smaller particles are more susceptible to resuspension than coarse particles (Johannesson et al. 2011). Analysing the particle size distribution of the sediment helps to see whether particles have sufficient time to settle and whether there is a risk of resuspension.

The shape of the wetland influences the mean water residence time and how the water is spread (Wörman & Kronnäs 2005). Wetlands with a long and narrow shape have a better distribution of water over the entire wetland area and longer water residence times, however they are also prone to higher flow velocities (Johannesson et al. 2015). Regarding the area of the wetland in relation to the catchment area (Aw:Ac) there is no optimal ratio (ibid.). Often wetlands with a larger surface area (Aw) compared to the catchment area (Ac) show higher relative P retention (% of P load) due to more time for particle settling and sorption, whereas they show lower area-specific P retention (g $m^{-2}$ of Aw) (Braskerud et al. 2005). The area-specific P retention is positively correlated with the HL, however it might be that the retention decreases after a HL breakpoint has been reached (Johannesson et al. 2015). Due to a high HL, the created turbulence can lead to the mobilisation of particles which enables desorption processes of P and to the release of loosely bound P fractions from sediment pore water (PW-P) (Lannergård et al. 2020). Sediment is essential for buffering P concentration from the overlying water column where P can potentially get removed from the water or released into it (Di Luca et al. 2017). In sediments, P can be bound to calcium (Ca-P), iron (hydr)oxides (Fe-P), aluminium (hydr)oxides (Al-P) or be present in microorganisms and organic matter (Org-P) (Lannergård et al. 2020). The P fractions play different roles in internal loading and potentially available P fractions are PW-P, Fe-P and labile Org-P (ibid.). The availability of P fractions is additionally influenced by environmental conditions like pH or oxygen levels (ibid.). For instance Ca-P might be released into the water column, if the pH goes below 4 (Di Luca et al. 2017), whereas Al-P is more stable regarding the pH (Lannergård et al. 2020). Redox sensitive P fractions (Fe-P) are potentially mobile and might be released into the water under anoxic conditions, that can occur just below the sediment surface, under ice cover of the wetland or due to an accumulation of plant litter (Søndergaard et al. 2003; Johannesson et al. 2011). Knowledge on internal P loading in constructed wetlands is still insufficient in literature. Analysing the different P fractions helps to understand exchange processes of P and whether the sediment is likely to be an internal source of P.

To assess the functioning of a wetland as a P trap, it is important to calculate how much particles and associated P are accumulated (Maine et al. 2007). Maintenance is also relevant for an effective operation of FWS wetlands (Chen 2011), and after a certain time the settled particles in wetlands need to be removed (Kynkäänniemi et al. 2013). The dredged sediment could be returned to agricultural fields if the sediment characteristics comply with legislation values for maximum allowed metal contents. Additionally, by analysing the plant available P (P-AL), the role of sediment as soil amendment can be estimated. Hence, nutrients can potentially be recycled which contributes to a circular economy and sustainable P management to ensure long-term access on P for future generations.

A current research issue is that many different wetlands with various shapes, sizes and positions exist in Sweden, but only few of them are studied. To strengthen research in this field, many wetlands with different designs need to be studied. This master thesis contributed to fill the research gap by investigating the functioning of one specific wetland Vilhelmsberg with a meandering shape, a large size and a unique position close to the Baltic Sea.

## 1.3. Objectives

The objectives of this study were to:

(i) quantify the P accumulation in the wetland Vilhelmsberg.

(ii) estimate the risk of internal loading regarding different P fractions in the sediment samples.

(iii) assess the current need for sediment removal and the possibility to recycle the sediments.

(iv) evaluate the influence on the P load from the upstream natural waterbody Maren and a private WWTP.

# 2. Material and methods

## 2.1. Site description

### 2.1.1. Wetland Vilhelmsberg

The wetland Vilhelmsberg is located in South-East Sweden in the Nynäshamn Municipality in Stockholm County (58°54'55.7" N, 17°49'2.6" E) and is part of the estate Djursnäs Säteri. It was constructed in 2014 and partly financed through the Rural Development Programme by the European Union and Sweden. It discharges directly into the Baltic Sea, more precisely the Baltic Proper or according to the division made in 2018 by HELCOM, the Western Gotland Basin (HELCOM 2021b). This sub-basin of the Baltic Sea showed a high eutrophication status according to the integrated eutrophication assessment 2018 (HELCOM 2018a, 2021b). The wetland Vilhelmsberg is categorized as FWS wetland and has a meandering shape with peninsulas separating the water flow path (Figure 1). The FWS area excluding peninsulas is 1.6 ha (Table 1). The length is 780 m and the average width is 20 m resulting in a length-to-width ratio (L:W) of 39. The inlet is an open ditch with a cross section of 3 m and the outlet is an open transition into the Baltic Sea with a cross section of 53 m. The measured water level of the wetland Vilhelmsberg was between 1.1 and 1.3 m at the sampling date, however, at that time it was approximately 20 cm above its relative water level in Landsort Norra, which is the closest sea observation station to the study site (SMHI 2021a). Since the construction of the wetland Vilhelmsberg, the lowest and highest water levels that have been measured at Landsort Norra close to the study site were -52 and +89 cm compared to relative water level. Thus, the wetland Vilhelmsberg was exposed to huge water fluctuations, but had a typical water level of 0.9 to 1.1 m referring to the relative water level of the Baltic Sea. Due to the water depth, emergent vegetation was only found on the edges of the wetland. Additionally, only very sparse submerged vegetation could be detected in the wetland.

The marine bottom material at the wetland Vilhelmsberg is postglacial, mud clay with the surface substrate soft clay (SGU 2021a). During construction the rather loose dredged soil material had been placed close to the wetland and it partly moved back into the water when a house was constructed close to the outlet. This soil material reached into the water with a curved shape, which can be seen on Figure 1 directly after the outlet on the right side.

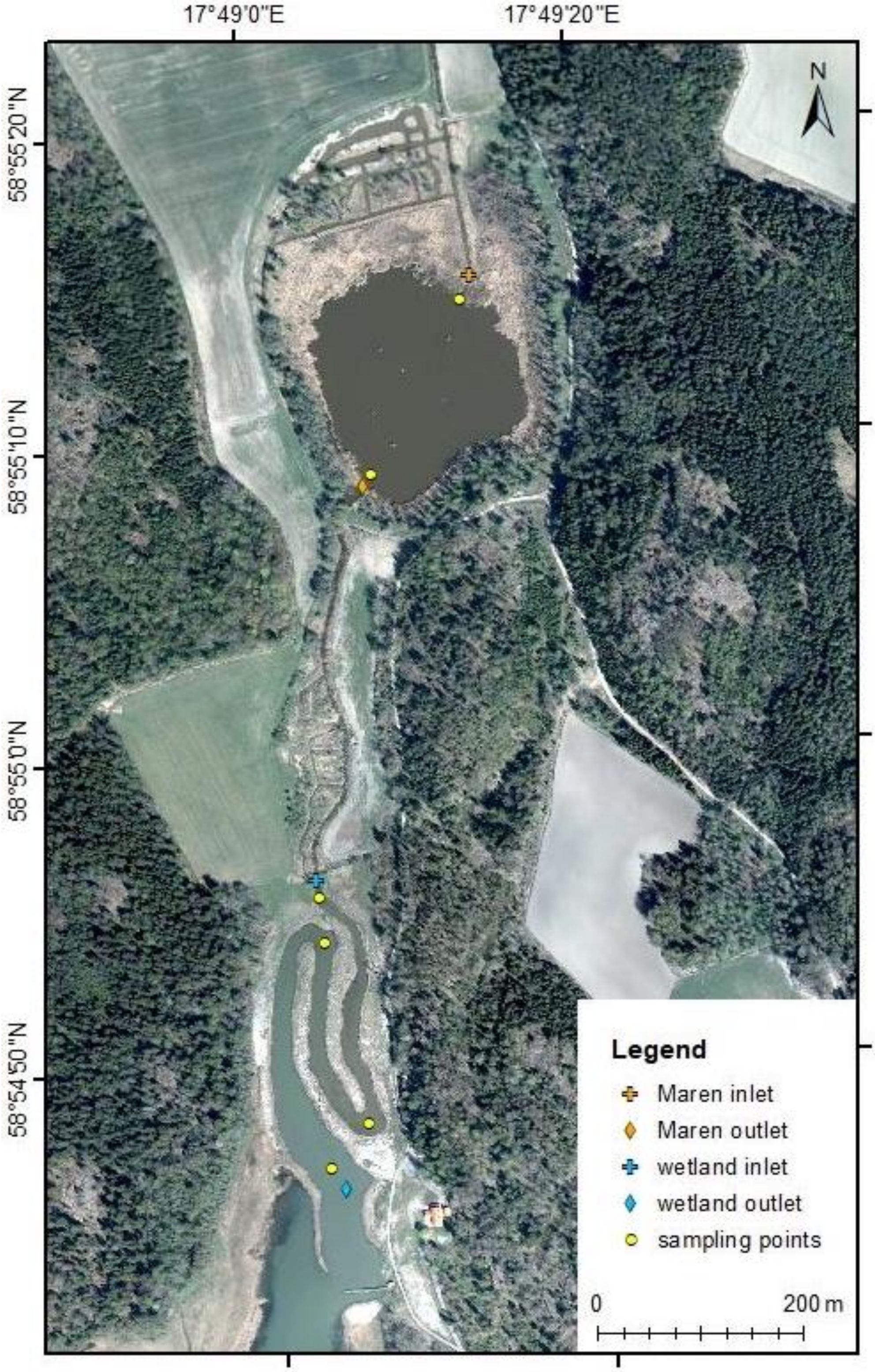


*Figure 1. Map of the meandering wetland Vilhelmsberg with an open transition into the Baltic Sea without barrier, and upstream the natural waterbody Maren. Sediment sampling points are marked with yellow circles. Inlets (pluses) and outlets (diamonds) are marked with orange signs in Maren and blue signs in the wetland Vilhelmsberg. (Picture source: Eniro, adopted in GIS with the coordinate system SWEREF 99 TM)*

## 2.1.2. Natural waterbody Maren

The natural waterbody Maren is located 419 m upstream of the wetland Vilhelmsberg following the water flow path (Figure 1). Maren was created naturally due to uplifting land from a former shallow bay of the Baltic Sea. Compared to the wetland Vilhelmsberg, Maren has a larger FWS area of 3.3 ha and a much smaller L:W of 3 resulting from a length of 234 m and an average width of 78 m (Table 1). Maren has a depth of 2 m at the centre and is shallower towards the edges, where it mostly has emergent vegetation. The inlet and outlet of Maren are open ditches with a width of 6 m each. Manmade ditches with islands and different water flow paths can be seen in Figure 1 in the adjacent area before Maren, that are used for rearing approximately 350 ducklings every year from spring until autumn. Similar ditches with islands were constructed between Maren and the wetland Vilhelmsberg, but are not in use for rearing ducks.

# 2.2. Catchment description

The catchment of the wetland Vilhelmsberg is 1325 ha large and includes the 1290 ha large catchment of Maren (Table 1). The area of each waterbody in relation to the respective catchment area (Aw:Ac) is 0.12 % for the wetland Vilhelmsberg, whereas it is 0.26 % for Maren. The average surface runoff in the catchment is 226 mm $yr^{-1}$ corresponding to a HL of 192 m $yr^{-1}$ for the wetland Vilhelmsberg and 88 m $yr^{-1}$ for Maren.

*Table 1. Overview of estimated wetland and catchment characteristics including the area of the wetland as well as Maren (Aw), the catchment area (Ac), the area of the wetland as well as the area of Maren in relation to its catchment area (Aw:Ac), the length-to-width ratio (L:W) and hydraulic load (HL)*

| | Aw (ha) | Ac (ha) | Aw:Ac (%) | L:W | HL (m $yr^{-1}$) |
|---|---|---|---|---|---|
| wetland Vilhelmsberg | 1.6 | 1325 | 0.12 | 39 | 192 |
| Maren | 3.3 | 1290 | 0.26 | 3 | 88 |

The land cover in the catchment of the wetland Vilhelmsberg included 933 ha forests (70 %), 212 ha arable land (16 %), 94 ha open land (7 %), 33 ha artificial surfaces including buildings and roads (3 %), 26 ha water (2 %), 24 ha mires (2 %), and 2 ha clear-cuts (<1 %) (Figure 2). The dominant soil types of the arable land in the catchment were silty clay (40 %) and clay (33 %) according to the digital soil map (Piikki & Söderström 2019). According to the landowner’s soil survey that was conducted by Eurofins (2012) the average contents of the different soil textures of arable land in the catchment were 52 % clay, 36 % silt and 12 % sand. The measured pH ranged from 5.1 to 7.1 with an average of 6.3. The measured P-AL ranged from 2 to 12 mg per 100 g DS with an average of 3.8 mg per 100 g DS and a median of 3.3 mg per 100 g DS. Moreover, horse paddocks were located in the southern part of the catchment with about 8 horses and in the northernmost part (no information on horse number or P-AL available). The P-AL status on fields close to the horse paddock in the southern part was above average, indicating higher P input due to animal excreta.

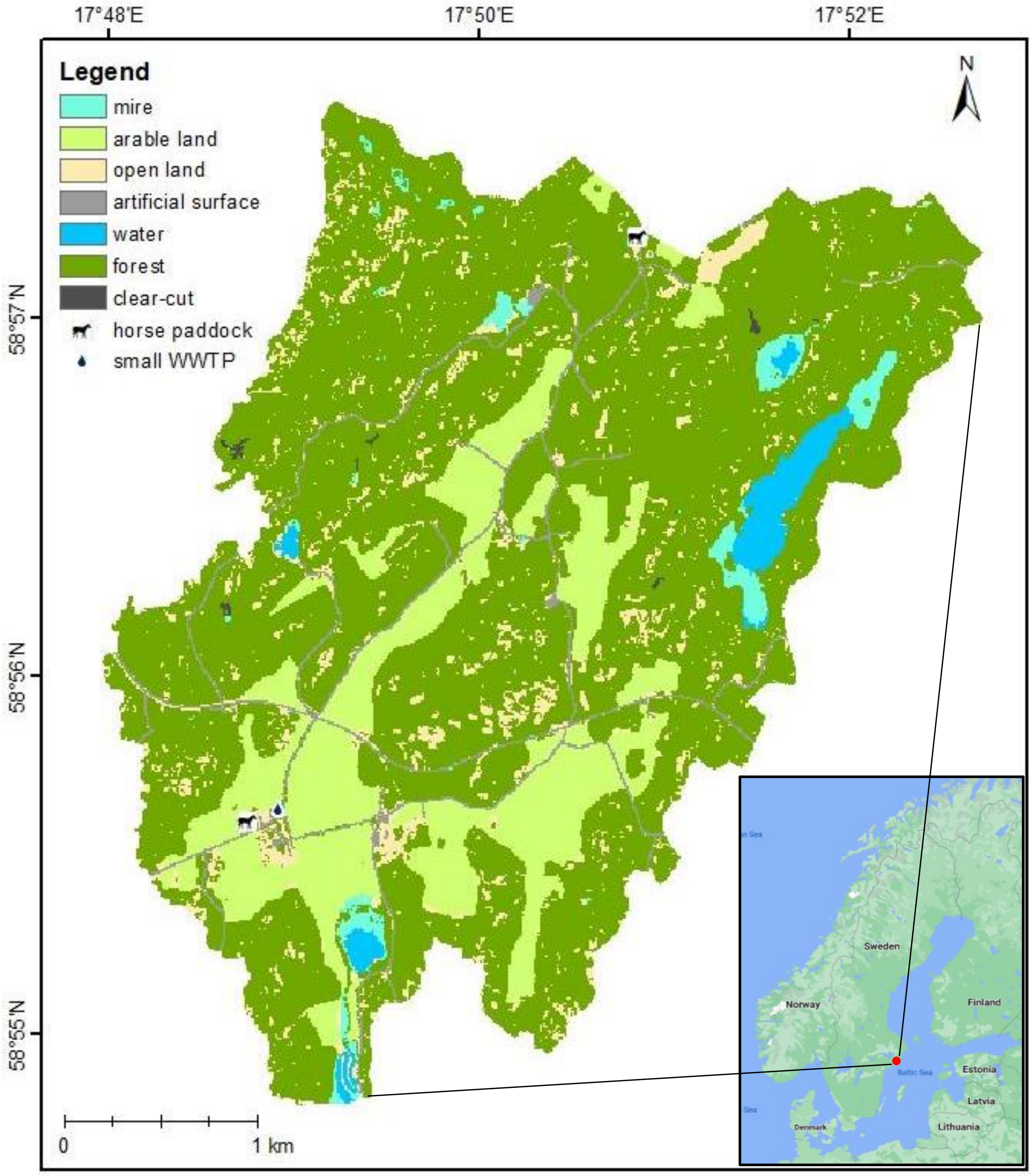


*Figure 2. The catchment of the wetland Vilhelmsberg showing land cover divided into mires, arable land, open land, artificial surfaces (including roads and buildings), forests and clear cuts. The wetland Vilhelmsberg is located at the southernmost part of the catchment. The location of the catchment within Sweden is indicated on the map in the bottom right corner with a red point. (Source of national ground cover data: Swedish EPA; map source: Google)*

A private underground WWTP is located in the catchment and discharges the treated water into the field ditch (Figure 3), 965 m upstream of the inlet of Maren. The WWTP received wastewater from several households including a slaughterhouse for wild animals. The treated wastewater corresponded to 9 population equivalents (PE) for blackwater and 0.25 PE for greywater. The slaughterhouse had a grease separator before the water entered the treatment system that consisted of a three-chamber tank for sludge separation followed by an infiltration bed with a sand filter. Additionally, it was estimated that individual houses located in the catchment amount to a total of 53 PE for blackwater. It was assumed that the individual houses had a similar treatment system like the WWTP but in a smaller format. Contrary to the private WWTP, the individual houses are not subject to regular inspections by legal authorities.

*Figure 3. Outlet of the drainage pipe of the WWTP into a field ditch at low flow conditions leading to standing water in the field ditch. The flow direction of the runoff water is indicated with blue arrows. (picture: Manuela Watschka)*

## 2.3. Estimations of loads and theoretical retention

### 2.3.1. Hydraulic load

For calculating the HL from the upstream catchment, local runoff data needed to be considered and converted to the area of the catchment, before relating it to the area of the wetland. According to the division made by SMHI (2021c), the wetland Vilhelmsberg is located in the sub-catchment area Fållnäsviken (identification number 5668). As a basis for calculating the HL, daily local water flow data from this sub-catchment area was used from the date when construction had been finished (16 October 2014) to the date before sampling (12 April 2021). First, the specific annual runoff for this time period was calculated by dividing the average discharge in the sub-catchment ($Q_{sub\text{-}catchment}$) with the area of the sub-catchment ($A_{sub\text{-}catchment}$) (equation 1). Afterwards the specific runoff got multiplied with the area of the catchment (Ac) to obtain the total annual water discharge in the catchment ($Q_{Ac}$) (equation 2). Finally, by dividing it through the area of the wetland (Aw) the yearly HL could be obtained (equation 3).

$$\text{specific runoff}\left[\frac{\text{m}}{\text{yr}}\right] = \frac{Q_{\text{sub-catchment}}\left[\frac{\text{m}^3}{\text{s}}\right] * 60 * 60 * 24 * 365}{A_{\text{sub-catchment}}\,[\text{m}^2]} \qquad (1)$$

$$Q_{Ac}\left[\frac{\text{m}^3}{\text{yr}}\right] = \text{specific runoff}\left[\frac{\text{m}}{\text{d}}\right] * \text{Ac}\,[\text{m}^2] \qquad (2)$$

$$\text{HL}\left[\frac{\text{m}}{\text{yr}}\right] = \frac{Q_{Ac}\left[\frac{\text{m}^3}{\text{d}}\right]}{\text{Aw}\,[\text{m}^2]} \qquad (3)$$

## 2.3.2. Theoretical P load and retention

*Theoretical P load from arable land*

The P load from different land cover categories including arable land, forested land, open land, mires and clear-cuts was estimated in consideration of the leaching concentrations for the respective land areas. For the leaching concentration of arable land, Sweden got divided into 22 different leaching regions based on production areas that have similar climate conditions and run-off patterns (Johnsson et al. 2019). The catchment of the wetland Vilhelmsberg is located in the leaching region six "Mälar- and Hjälmarbygden". For the other mentioned land categories, Sweden also got divided into different leaching regions (Hansson et al. 2019), where the catchment is located in the South-Eastern leaching region. The P load for each mentioned land cover category was calculated by multiplying the respective leaching concentration with the respective land area in the catchment and the specific runoff calculated for the catchment (equation 4).

$$\text{Land P load}\left[\frac{\text{kg}}{\text{yr}}\right] = \text{P leaching conc.}\ \left[\frac{\text{mg}}{\text{l}}\right] * \text{land cover area}\ [\text{km}^2] * \text{specific runoff}\ \left[\frac{\text{mm}}{\text{yr}}\right] \tag{4}$$

*Theoretical P load from domestic wastewater*

The P load from domestic wastewater included calculations for a WWTP and for individual houses located in the catchment. Information about treatment system and PE of the WWTP was provided by the owner, whereas the wastewater treatment situation and PE of individual houses was unknown and was estimated in this study. One house that is located close to the wetland's outlet (Figure 1) did not get considered in this study because the produced blackwater gets collected in a closed tank and picked up by the municipality, and the treated greywater gets discharged into the Baltic Sea after the outlet of the wetland.

For calculating the outflowing P load from the WWTP, received analysis results of P concentrations in outflowing treated wastewater from the years 2019 and 2020 were used and multiplied with the amount of wastewater ($Q_{WW}$) (equation 5). The amount of wastewater of the WWTP was calculated by considering the PE for blackwater ($PE_{BW}$) and greywater ($PE_{GW}$) as well as the daily produced amount of blackwater ($Q_{BW}$) and greywater ($Q_{GW}$) per PE (equation 6). The typical values of produced wastewater amounts per PE and day were taken from the Swedish Agency for Marine and Water Management (SwAM 2016).

$$\text{WWTP P load}_{\text{out}}\left[\frac{\text{kg}}{\text{yr}}\right] = \text{Q}_{\text{WW}}\left[\frac{\text{l}}{\text{d}}\right]\ *\ \text{P conc.}_{\text{out}}\ \left[\frac{\text{mg}}{\text{l}}\right]\ * 365 * 10^{-6} \tag{5}$$

$$\text{WWTP Q}_{\text{WW}}\left[\frac{\text{l}}{\text{d}}\right] = \text{PE}_{\text{BW}} * \text{Q}_{\text{BW}}\left[\frac{\text{l}}{\text{PE,d}}\right] + \text{PE}_{\text{GW}} * \text{Q}_{\text{GW}}\left[\frac{\text{l}}{\text{PE,d}}\right] \tag{6}$$

The received analysis results were also used for estimating the P treatment reduction of the WWTP in 2019 and 2020. Therefore, the inflowing P load was calculated under consideration of the PE, amounts and concentration of both black and greywater (equation 7). Values for the P concentration in the untreated black and greywater per PE and day were taken from Olshammar et al. (2015). The inflowing P concentration of the mixed wastewater was calculated by dividing the daily inflowing P load by the daily amount of wastewater (equation 8). Finally, the P treatment reduction (P red.) for both years was calculated using equation 9.

$$\text{WWTP P load}_{\text{in}}\left[\frac{\text{mg}}{\text{d}}\right] = \text{PE}_{\text{BW}} * \text{Q}_{\text{BW}}\left[\frac{\text{l}}{\text{PE,d}}\right] * \text{P conc}_{\text{BW}}\left[\frac{\text{mg}}{\text{l,PE,d}}\right] + \text{PE}_{\text{GW}} * \text{Q}_{\text{GW}}\left[\frac{\text{l}}{\text{PE,d}}\right] * \text{P conc}_{\text{GW}}\left[\frac{\text{mg}}{\text{l,PE,d}}\right] \quad (7)$$

$$\text{WWTP P conc.}_{\text{in}}\left[\frac{\text{mg}}{\text{l}}\right] = \frac{\text{P load}_{\text{in}}\left[\frac{\text{mg}}{\text{d}}\right]}{\text{Q}_{\text{WW}}\left[\frac{\text{l}}{\text{d}}\right]} \quad (8)$$

$$\text{P red.}\,[\%] = \left(1 - \frac{\text{P conc.}_{\text{out}}\left[\frac{\text{mg}}{\text{l}}\right]}{\text{P conc.}_{\text{in}}\left[\frac{\text{mg}}{\text{l}}\right]}\right) * 100 \quad (9)$$

The P load from domestic wastewater of individual houses distributed in the catchment was calculated under consideration of PE, the amount of wastewater, the P concentration in untreated wastewater and the required treatment reduction by law (equation 10). The PE for individual houses in the catchment were estimated by researching in online address books and maps and combined with provided local information. It was assumed that only blackwater gets produced and values for the produced amounts and P concentration in untreated wastewater were taken from the same source as for the calculation of the WWTP. The treatment reduction was assumed to comply with guidelines for a normal level of protecting environment and health from the Swedish EPA (NFS 2006:7) on treating domestic wastewater from individual houses and community facilities for up to 25 PE.

$$\text{Individual houses P load}_{\text{out}}\left[\frac{\text{kg}}{\text{yr}}\right] = \text{Q}_{\text{WW}}\left[\frac{\text{l}}{\text{PE,d}}\right] * \text{P conc}_{\text{in}}\left[\frac{\text{mg}}{\text{l,PE,d}}\right] * \text{PE} * (1 - \text{P red.}\%) * 365 * 10^{-6} \quad (10)$$

*Theoretical P load and retention of Maren and the wetland Vilhelmsberg*

It was assumed that the P load to the wetland Vilhelmsberg consisted on the one hand of P that was not retained in Maren and on the other hand of P load from land downstream Maren that drained directly to the wetland Vilhelmsberg (equation 11). In order to obtain the land cover areas of this residual land part, first the land cover areas for the entire catchment of the wetland Vilhelmsberg were identified and then the land cover areas of Maren got subtracted from it. Thus, the P load for that area was calculated with equation 4.

It was assumed that the P load of Maren consisted of the P load from land in Maren's catchment and P coming from treated domestic wastewater. The area-specific P load for Maren was calculated by taking the sum of P loads from domestic wastewater and land within the catchment of Maren and dividing it by its water area (equation 12). For the WWTP the average value of the calculated outflowing P loads of the years 2019 and 2020 was taken.

For both waterbodies, the theoretical area-specific P retention was calculated with a regression model for P retention that was established by Weisner et al. (2016) by means of sampling and analysing 15 different wetlands in Southern Sweden (equation 13).

$$\text{wetland Vilhelmsberg P load}\left[\frac{\text{kg}}{\text{ha,yr}}\right] = \frac{\text{Maren P load}\left[\frac{\text{kg}}{\text{yr}}\right] - \text{Maren P retention}\left[\frac{\text{kg}}{\text{yr}}\right] + \text{Land P load}\left[\frac{\text{kg}}{\text{yr}}\right]}{\text{wetland Aw [ha]}} \tag{11}$$

$$\text{Maren P load}\left[\frac{\text{kg}}{\text{ha,yr}}\right] = \frac{\text{WWTP P load}_{\text{out}}\left[\frac{\text{kg}}{\text{yr}}\right] + \text{Individual houses P load}_{\text{out}}\left[\frac{\text{kg}}{\text{yr}}\right] + \text{Land P load}\left[\frac{\text{kg}}{\text{yr}}\right]}{\text{Maren Aw [ha]}} \tag{12}$$

$$\text{theoretical P retention}\left[\frac{\text{kg}}{\text{ha,yr}}\right] = -0.0003 * \left(\text{P load}\left[\frac{\text{kg}}{\text{ha,yr}}\right]\right)^2 + 0.4584 * \text{P load}\left[\frac{\text{kg}}{\text{ha,yr}}\right] \tag{13}$$

## 2.4. Sediment sampling and analyses

### 2.4.1. Sampling procedure

The sampling was carried out on two subsequent days, 13 and 14 April 2021. Sediment core samples were collected at two sampling points in Maren and four sampling points in the wetland Vilhelmsberg (Figure 1).

The sediment cores were taken from a small rowboat using a Willner gravity sediment-coring device with a rod and a tube screwed on it. After pushing the device into the bottom, a trigger was used to close the top of the device. The created vacuum allowed to lift up the sediment core. Attention had been taken to close the bottom of the sediment core tube with a rubber stopper before it reached the water surface. On the boat the sediment-coring device got removed and the top of the tube got closed with another rubber stopper. Several sediment cores needed to be collected at one sampling point in order to get enough sediment for the various analyses.

At each sampling point the water depth was measured with a rod that was put into the water until it reached the top of the sediment surface. An attempt to measure the sediment depth was made by pushing the rod down from the sediment surface until substantial force must be applied where it reaches the bottom (Simpson & Wu 2014). However, it was not possible to feel the difference due to the rather soft bottom and the wind-induced water movement where little force needed to be applied already to keep the rod straight. Therefore, the sediment depth needed to be estimated by means of the collected sediment cores.

Back on land, the collected sediment cores got observed and the sediment depth got measured with a ruler from outside the tube. Then, the water above the sediment was carefully removed and a slicer with a centimetre scale was used for separating the sediment from the sediment core tube into a separate cup slice by slice. This was done until a difference in the substrate could be observed to ensure that the entire sediment layer and not the underlying layers got collected. Gravel or large organic matter (e.g. twigs or roots) that was either easily detectable or disturbing the slicing, was removed from the sample.

### 2.4.2. Laboratory analyses

The collected samples were analysed by different laboratories for: (i) total phosphorus (TP) and P fractions, (ii) metal content, (iii) P-AL, total nitrogen (TN) and total carbon (TC), and (iv) particle size distribution. The metal analysis was performed by ALS Scandinavia in Stockholm, whereas all other analyses were performed by laboratories at SLU in Uppsala.

(i) The water content, loss on ignition, TP and P fractions were analysed at the Department of Aquatic Sciences and Assessment at SLU. The water content was determined by using a freeze drier (-40°C, 96 hr). Afterwards, the loss on ignition (organic matter content) of the freeze-dried samples was determined by using a muffle oven (550 °C, 2 hr). The wet bulk density was determined with the analysed organic matter and water content of each sample using a method from Pajunen (2000). Moreover, a sequential chemical extraction was performed where stepwise extractants were added to the sediment sample which removed different fractions of bound P. Firstly, PW-P was determined with double de-ionized (MQ) water. Secondly, Fe-P was determined with bicarbonate buffered dithionite (BD) solution. Next, Al-P and Org-P were determined with a sodium hydroxide (NaOH) solution. Finally, Ca-P was determined with a hydrochloric acid (HCl) solution. The sum of all P fractions (PW-P, Fe-P, Al-P, Org-P and Ca-P) was considered as TP.

(ii) The chosen metal analysis package "MS-1 Metals (11) in soil, sludge and sediment, HNO3 digestion" comprised 11 metals (As, Ba, Cd, Co, Cr, Cu, Hg, Ni, Pb, V, Zn). The metal analysis was performed with a sector field mass spectrometry with inductively coupled plasma (ICP-SFMS) according to SS-EN ISO 17294-2:2016 and US EPA Method 200.8:1994. The sample preparations for the metal analysis included drying at 50 °C, sieving <2 mm, grinding and dissolution in nitric acid ($HNO_3$).

(iii) The analyses of P-AL, TN and TC were performed by the Soil and Plant Laboratory at the Department of Soil and Environment at SLU. The TN and TC were analysed with the elemental analyser LECO TruMac. The P-AL was analysed by extraction with ammonium lactate and by using an inductively coupled plasma (ICP-AES) spectrometer.

(iv) The analysis of the particle size distribution was performed by the Soil Physics Laboratory at the Department of Soil and Environment at SLU. The sample preparation included air-drying, grinding and sieving with a 2 mm mesh. Afterwards, particle sizes below 2 mm were determined by using a laser diffraction particle size distribution analyser (Partica LA-950 V2 from

HORIBA). The results were clustered into following fractions: fine clay (<0.2 µm), medium clay (0.2 - 0.63 µm), coarse clay (0.63 - 2 µm), fine silt (2 - 6 µm), medium silt (6 - 20 µm), coarse silt (20 - 60 µm), fine sand (60 - 200 µm), medium sand (200 - 600 µm), coarse sand (600 - 2000 µm).

### 2.4.3. Estimations of accumulation and P release rate

*Sediment accumulation rate*

The average sediment depth at each sampling point was estimated by using the sediment depths of the collected sediment cores. In order to obtain an annual sediment accumulation rate, first the average sediment depth of the wetland Vilhelmsberg was calculated by taking the average of the beforehand calculated sediment depths at each sampling point. Then, it was divided by the past 6.5 years since construction until the sampling dates in the middle of April 2021 (equation 14). For Maren the yearly sedimentation rate could not be calculated because it is unknown when the natural waterbody developed and how deep the sediment layer at the inlet is.

$$sediment\ accumulation\ rate\ \left[\frac{cm}{yr}\right] = \frac{average\ sediment\ depth\ [cm]}{years\ since\ construction\ [yr]} \tag{14}$$

*Accumulation of particles and P*

The wet bulk density and dry fraction of each sample were provided by the analysis of the Laboratory from SLU. First, the dry bulk density (g $cm^{-3}$) of each sample was calculated by multiplying the respective wet bulk density (g $cm^{-3}$) with the respective dry fraction. For each P fraction the concentration per mass (mg $g^{-1}$ DS) was multiplied with the dry bulk density (g $cm^{-3}$) in order to obtain the concentration per volume (mg $cm^{-3}$). The TP concentration was obtained by summing up the respective values of all P fractions (PW-P, Fe-P, Org-P, Al-P, Ca-P).

At each sampling point of the wetland Vilhelmsberg the accumulation of particles and P was estimated. For the particle accumulation the dry bulk density was multiplied with the sediment depth and a factor of 100 for converting the unit (equation 15). For the P accumulation at each sampling point, the TP concentration per volume was multiplied with the sediment depth and a factor of 100 for converting the unit (equation 16). The sediment depth of the sediment cores that were collected for the analyses of TP and particle density was 3 cm at all sampling points within the wetland Vilhelmsberg. Afterwards, the average accumulation of particles and P of the four sampling points of the wetland Vilhelmsberg was taken and divided by the years of construction in order to obtain the annual accumulation of particles (t $ha^{-1}$ $yr^{-1}$) and P (kg $ha^{-1}$ $yr^{-1}$). For Maren the accumulation of particles and P could not be estimated because of its unknown age and sediment depth at the inlet.

$$\text{particle accumulation}\ \left[\frac{\text{t}}{\text{ha}}\right] = \text{dry bulk density}\ \left[\frac{\text{g}}{\text{cm}^3}\right] * \text{sediment depth [cm]} * 100 \tag{15}$$

$$\text{P accumulation}\ \left[\frac{\text{kg}}{\text{ha}}\right] = \text{TP conc.}\left[\frac{\text{mg}}{\text{cm}^3}\right] * \text{sediment depth [cm]} * 100 \tag{16}$$

*P release rate*

The sum of PW-P and Fe-P represented mobile P that is likely to be released under anoxic conditions (Pilgrim et al. 2007). The maximum potential P release rate for each sampling point was calculated with a linear regression that Pilgrim et al. (2007) established by incubating samples from lake sediment at warm temperatures (equation 17). The annual maximum potential P release rate (kg $yr^{-1}$) was calculated with equation 18 both for the wetland Vilhelmsberg and Maren by multiplying the respective water area with the average maximum potential P release rate of the four samples in wetland Vilhelmsberg respectively two samples in Maren.

Potentially available P is considered the sum of PW-P, Fe-P and labile Org-P, whereof the latter can get estimated by subtracting Org-P that gets measured in deeper sediment layers (Lannergård et al. 2020). For this master thesis no background Org-P was measured and thus the labile Org-P was estimated with a linear regression from wetland sediment (Michélsen 2021) (equation 19).

$$\text{P release rate} \left[\frac{\text{mg}}{\text{m}^2\text{,d}}\right] = 15.1 * \text{ mobile P} \left[\frac{\text{g}}{\text{m}^2\text{,cm}}\right] - 1.7 \tag{17}$$

$$\text{annual P release rate} \left[\frac{\text{kg}}{\text{yr}}\right] = \text{average P release rate} \left[\frac{\text{mg}}{\text{m}^2\text{,d}}\right] * \text{Aw} \left[\text{m}^2\right] * 365 * 10^{-6} \tag{18}$$

$$\text{Labile Org-P} \left[\frac{\text{mg}}{\text{g DS}}\right] = 0.7038 * \text{ Org-P} \left[\frac{\text{mg}}{\text{g DS}}\right] + 0.0043 \tag{19}$$

### 2.4.4. Estimation of allowed sewage sludge application

The maximum amount of sewage sludge that is allowed to be added to agricultural soils was calculated with equation 20 under consideration of the maximum allowed amount of metals in sewage sludge that is used for agricultural purposes according to regulation SNFS 1994:2 (with two amendment regulations SNFS 1998:4, SNFS 2001:5) (Swedish EPA 1994). This was done for each sampling point and the average of the wetland Vilhelmsberg by using the metal content and dry fraction obtained within the metal analysis and the wet bulk density obtained within the TP analysis.

$$\text{amount}_{sewage\ sludge} \left[\frac{\text{m}^3}{\text{ha,yr}}\right] = \frac{\text{metal content} \left[\frac{\text{mg}}{\text{kg DS}}\right] * \text{dry fraction} * \text{wet bulk density} \left[\frac{\text{g}}{\text{cm}^3}\right]}{\text{max. allowed amount of metals in sewage sludge} \left[\frac{\text{g}}{ha,yr}\right]} \tag{20}$$

## 2.5. Work in ArcMap

The GIS (Geographic Information System) ArcMap 1.7 was used for estimating wetland and catchment characteristics with the coordinate system SWEREF 99 TM as a spatial reference. Most of the layers used in this study including maps, orthophotos and the digital elevation model (DEM) were taken from the SLU download service GET. Additionally, national ground cover data of 2018 was downloaded from the Swedish Environmental Protection Agency (EPA) and soil maps of the arable land were downloaded from the Geological Survey of Sweden (SGU) that produced "clay content maps" in collaboration with SLU.

### 2.5.1. Estimation of wetland's characteristics

The characteristics of the wetland Vilhelmsberg and Maren were estimated in GIS ArcMap by means of an orthophoto from 2017 with a resolution of 0.25 m. For delineating the area, the FWS was considered which included emergent vegetation at the inlet of Maren. Whereas swampy areas surrounding Maren and peninsulas reaching into the wetland Vilhelmsberg were not included.

The position of the inlet of the wetland Vilhelmsberg was right after a small bridge, whereas the position of the wetland's outlet was uncertain. In order to exclude the part where soil material eroded down while constructing a nearby house, the outlet point was chosen upstream of it. Then, the outlet line was drawn by considering the flow direction of the wetland Vilhelmsberg. The length from the inlet to the outlet of the wetland Vilhelmsberg was calculated using the tool "Collapse Dual Lines To Centreline", however, centrelines can only be established between two separated lines (ESRI 2021). Therefore, the polygon of the area was transformed into lines using the tool "Polygon To Line" and then the inlet and outlet parts were cut out in order to get two separated lines. The average width was calculated from width measurements at every 40 m along the centreline of the wetland Vilhelmsberg.

The points for the inlet and outlet of Maren were chosen based on the orthophoto where the open ditch ended respectively started. The length of Maren was measured with a straight line from the inlet to the outlet point. The average width was calculated from width measurements at every 40 m along this line. For both the wetland Vilhelmsberg and Maren, the L:W was calculated by dividing the total length by the average width. Moreover, the sampling points were placed along the same lines that were used for the length measurements in order to measure the distance from the inlet by using the tool "Split Line at Points".

### 2.5.2. Catchment delineation

The catchment was delineated in GIS ArcMap by using DEM with a resolution of 2 m. The catchment until the inlet point of the wetland Vilhelmsberg was delineated by processing received flow accumulation and flow direction data from SLU. However, these flow data cut off the Baltic Sea including major parts of the wetland Vilhelmsberg. Therefore, the catchment for the outlet point of the wetland Vilhelmsberg was delineated separately and then southern parts were added manually to the polygon of the catchment of the inlet point. In order to establish flow and accumulation data, first the tool "Fill" was applied to remove small imperfections of the DEM by filling sinks in the surface raster. Then, with the filled DEM, the tool "Flow Direction" was applied, which created a raster of flow directions from each single cell to the steepest downslope neighbour cell. As a result, the cells got one out of eight different colours, indicating in which neighbour cell the water will flow. With the filled flow direction raster as an input, the flow accumulation was derived that gave the number of cells where water flows to a particular cell. If more cells drained into one cell, it would have a higher accumulation value compared to its neighbour cells (de Smith et al. 2018). Due to different colours for the cells, it was possible to see a stream network of cells with high accumulation.

For the catchment delineation of both the inlet and outlet point it was necessary that the points were placed at a cell of the appearing stream network of cells with high flow accumulation. This was done with the tool "Snap Pour Point", which snapped the pour point to the cell with the highest accumulation of water flow. Then, in both cases the size of the catchment was estimated with the tool "Watershed" using both the pour point and flow direction data as an input. The resulting watershed needed to be transformed from raster cells into a polygon using the tool "Raster to Polygon" to be able to use it further and calculate the area.

Having delineated the catchment, as a next step, the land cover and the soil type of arable land within the catchment was obtained by using certain raster layers on national ground cover (with a resolution of 10 m) and arable soil maps (with a resolution of 50 m). In all cases, first the specific raster layer was clipped to the shape of the catchment by using the tool "Clip (Data Management)". For calculating the area of the different categories of land cover and soil type it was necessary to transform the clipped raster file into multipart polygons by using the tool "Raster to Polygon". The mean content of each soil texture class was found at the layer properties under the folder "Symbology" by selecting classify and show mean.

## 2.6. Soil survey of arable land

The analysed P-AL and particle size distribution in the sediment were compared with results of a soil survey of the arable land in the catchment that was conducted in July 2012 by Eurofins Food and Agro Testing Sweden AB. Soil samples were taken at 203 different locations, mostly within the catchment of the wetland Vilhelmsberg (Appendix Figure 15). Only fields located within the catchment of the wetland Vilhelmsberg were used for comparison. The soil survey comprised amongst others the following analyses: pH, P-AL, P-HCl and the percentage of clay and sand. The percentage of silt was estimated in this thesis by subtracting the percentages of clay and sand from 100.

## 2.7. Statistical analyses

The statistical analyses were carried out in Excel by means of the tool "Data Analysis" that is available under the add-in program "Analysis ToolPak". Thus, Pearson correlation coefficient (r) could be established that showed to which extend two or more variables varied together independent of the measurement unit (Microsoft 2021).

Furthermore, linear regression lines were established and the adjusted $R^2$ was generated which showed the goodness of fit. In this master thesis, the given $R^2$ represented the adjusted $R^2$ that considered the number of sampling points. A significance level of 0.05 was chosen, meaning that a p-value lower than 0.05 would have a statistically significant relationship.

# 3. Results

## 3.1. Sediment characterisation

At Maren, the sediment cores from both sampling points consisted of light brown mineral sediment with organic vegetation parts. At the inlet the top 4 cm were finer particles, while the bottom could not be detected (Figure 4). Contrary, the bottom of the outlet core of Maren could be detected and was grey clay.

In all sampling points in the wetland Vilhelmsberg the surface sediment consisted of light brown mineral matter (Figure 4). At the sampling point at the inlet and the first curve (Figure 1), it was followed by black organic material with orange spots of precipitated iron. The sampling points that were located closer to the upstream land area (inlet and second curve) showed a grey clay bottom, whereas the two sampling points closer to the ocean (outlet and first curve) had a comparatively softer brownish-grey clay bottom.

During sampling, the water clarity increased from the inlet towards the outlet of the wetland Vilhelmsberg, which is also slightly visible on Figure 1 by a brownish colour in the beginning that gets lighter towards the end of the wetland. At the second curve the water already had a better clarity than at the sampling points before, so that the top of the sediment coring device could be seen in the water while inserting it into the sediment. At the outlet of the wetland Vilhelmsberg even the bottom was visible, whereas the bottom could not be seen at all other sampling points.


*Figure 4. Sediment cores collected at the different sampling points (Pictures: Pia Geranmayeh)*

### 3.1.1. Sediment accumulation rate

In the wetland Vilhelmsberg the average sediment accumulation layer was 3.5 cm, whereof the highest was at the outlet (4.0 cm) and the lowest at the second curve (3 cm). The average sediment accumulation layers at the inlet and the first curve were similar (3.5 cm). Since the construction of the wetland Vilhelmsberg the average sediment accumulation rate was 0.5 cm $yr^{-1}$.

In Maren the sediment accumulation was lower at the outlet (9.5 cm) than at the inlet. The sediment depth at Maren's inlet could not be defined because no bottom layer was observed in the collected sediment core tubes. Hence, the entire substrate in the collected core tubes was assumed to be the sediment accumulation layer, and it was probably larger than that (> 45 cm). For the analyses only the top 12 cm of the accumulated sediment were used.

### 3.1.2. Water content, organic matter and P concentration

The average water content of the sediment was higher in Maren (80 %) than in the wetland Vilhelmsberg (71 %) (Table 2). The wet bulk density ranged from 1.11 to 1.28 g $cm^{-3}$ in the wetland Vilhelmsberg and from 1.08 to 1.15 g $cm^{-3}$ in Maren. Amongst all sampling points, the wetland's inlet had the lowest water content (58 %) and the highest wet and dry bulk density (1.28 respectively 0.54 g $cm^{-3}$).

Even though, the organic matter content varied more within the wetland Vilhelmsberg than in Maren, the average organic matter content was 18 % for both waterbodies (Table 2). Within Maren, the difference in organic matter content was 9 % between the inlet (14 %) and outlet (22 %). Within the wetland Vilhelmsberg the highest difference in organic matter was 26 % between the second curve (9 %) and the outlet (35 %).

Results from the P analysis showed that the wetland Vilhelmsberg had a lower TP concentration per mass (mg $g^{-1}$ DS), but higher per volume (mg $cm^{-3}$) compared to Maren (Table 2). The disparity was probably due to a higher average density and lower water content of the sediment in the wetland. It will be better to compare TP concentrations per volume rather than per mass to avoid misconceptions if the water content in sediment samples has large disparities as it was the case in this thesis.

Within the wetland, the sediment at the second curve had the highest TP concentration. The sediment at the inlet had the second highest TP concentration per volume although it had the lowest per mass, due to a high density. Similarly, within Maren the TP concentration per volume at the inlet exceeded the outlet due to a higher sediment density, even though it was vice versa for the TP concentration per mass.

*Table 2. Sediment characteristics including water and organic matter content, wet and dry bulk density and TP concentration on average and at the different sampling points for Maren and the wetland Vilhelmsberg*

| | Maren | | | wetland Vilhelmsberg | | | | |
|---|---|---|---|---|---|---|---|---|
| | average | inlet | outlet | average | inlet | curve 1 | curve 2 | outlet |
| water content (%) | 80 | 75 | 84 | 71 | 58 | 80 | 72 | 74 |
| organic matter content (%) | 18 | 14 | 22 | 18 | 15 | 12 | 9 | 35 |
| bulk density (g $cm^{-3}$) | | | | | | | | |
| wet | 1.12 | 1.15 | 1.08 | 1.18 | 1.28 | 1.12 | 1.19 | 1.11 |
| dry | 0.23 | 0.29 | 0.17 | 0.35 | 0.54 | 0.22 | 0.33 | 0.29 |
| TP conc. (g $kg^{-1}$ DS) | 1.08 | 0.97 | 1.19 | 0.85 | 0.56 | 0.90 | 1.32 | 0.60 |
| TP conc. (mg $cm^{-3}$) | 0.24 | 0.28 | 0.20 | 0.28 | 0.31 | 0.20 | 0.44 | 0.17 |

### 3.1.3. Particle size distribution

In both waterbodies, silt was the dominating fraction accounting for over half of the particles in the sediment, with 68 % in the wetland and 56 % in Maren (Figure 4). The second highest particle fraction was clay (20 %) in the wetland Vilhelmsberg and sand (28 %) in Maren.

On average, the wetland Vilhelmsberg showed a higher content of silt (68 %) and clay (20 %) than Maren (56 % silt and 17 % clay). Within the wetland Vilhelmsberg, the clay content at the inlet was three times higher than at the curves and eight times higher than at the outlet, indicating resuspension caused by the Baltic Sea. At the inlet, even fine clay (4 %) and medium clay (29 %) could be found.

Similarly, in Maren, the clay content at the inlet was two times higher than at the outlet and included fine clay (3 %) and medium clay (13 %). Moreover, no significant correlation between clay content and TP concentration per volume or per mass was found.

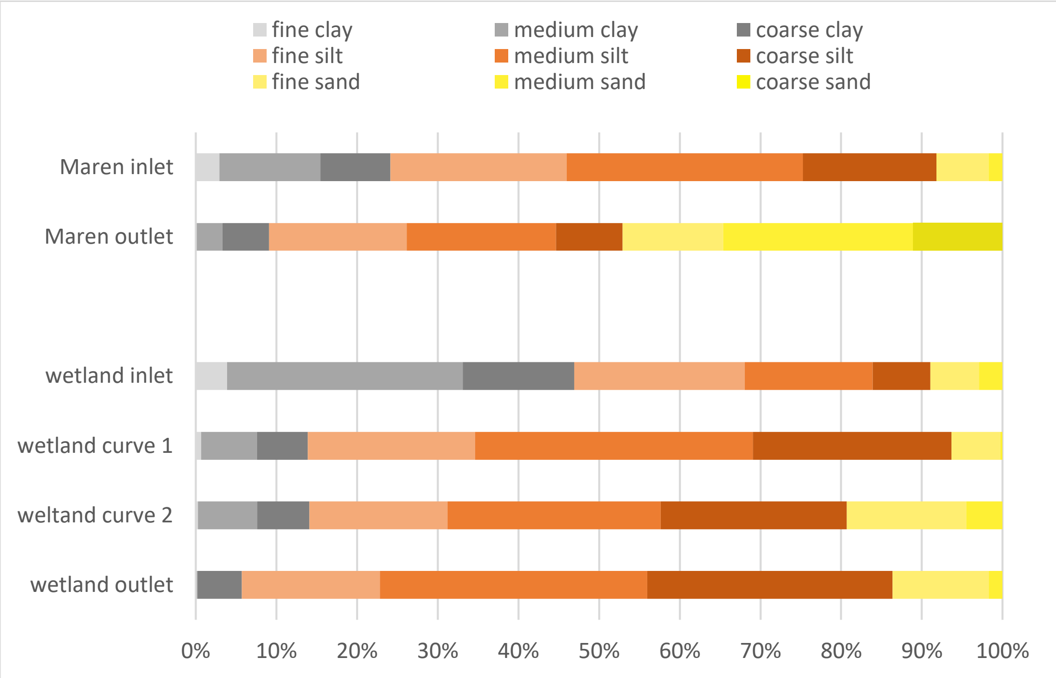


*Figure 5. Particle size distribution at the different sampling points including following fractions: fine clay (<0.2 µm), medium clay (0.2 - 0.63 µm), coarse clay (0.63 - 2 µm), fine silt (2 - 6 µm), medium silt (6 - 20 µm), coarse silt (20 - 60 µm), fine sand (60 - 200 µm), medium sand (200 - 600 µm), coarse sand (600 - 2000 µm).*

## 3.2. Accumulation of particles and P

Since the construction of the wetland Vilhelmsberg, the particle accumulation was 16 t $ha^{-1}$ $yr^{-1}$. The highest particle accumulation was at the inlet which had the highest density, followed by the second curve and the outlet (Figure 6). The lowest particle accumulation was at the first curve which also had the lowest density.
The P accumulation of the wetland Vilhelmsberg was 13 kg $ha^{-1}$ $yr^{-1}$ since its construction. The P accumulation was the highest at the second curve, followed by the inlet and the first curve (Figure 6). The lowest P accumulation was at its outlet.

No significant correlation was found between the distance from the inlet and the accumulation of particles or P. Likewise, no significant correlation between the accumulation of particles and P could be detected.

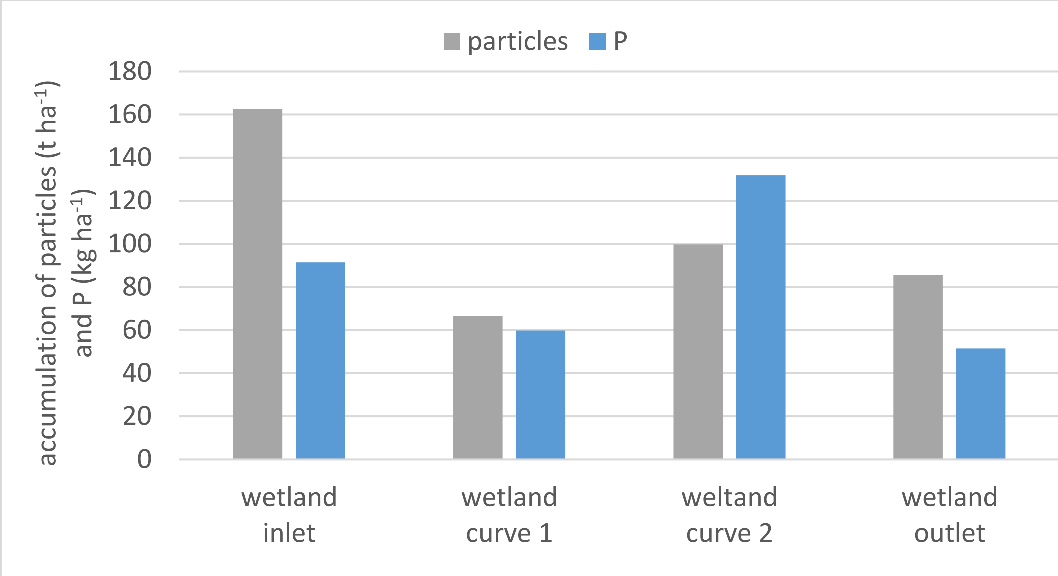


*Figure 6. Accumulation of particles (grey) and P (blue) in the sediment top 3 cm at the different sampling points in the wetland Vilhelmsberg*

### 3.2.1. P fractions

The share of each P fraction on the TP concentration varied between sampling points (Figure 7). In the wetland Vilhelmsberg the major P fractions were on average 35 % Ca-P and 30 % Org-P, followed by minor P fractions Al-P (17 %) and Fe-P (15 %). Likewise, in Maren the highest average P fraction was Ca-P (33 %), however Al-P (28 %) was higher and Org-P (23 %) was lower compared to the wetland Vilhelmsberg. On average, the relative amounts of Fe-P (14 %) and PW-P (2 %) in Maren were quite similar to the wetland.

Within each waterbody the share of Ca-P was highest at the outlets and the share of Al-P was highest at the inlets. The share of Fe-P was the highest in the first curve of the wetland (21 %). Contrary to all other sampling points, in the second curve of the wetland, the relative amount of Org-P was the highest (50 %), outreaching all other P fractions. Therefore, in this sampling point the relative amounts of Fe-P (10 %) and Al-P (9 %) were the lowest of all sampling points. At all sampling points PW-P had the lowest share (<4 %).

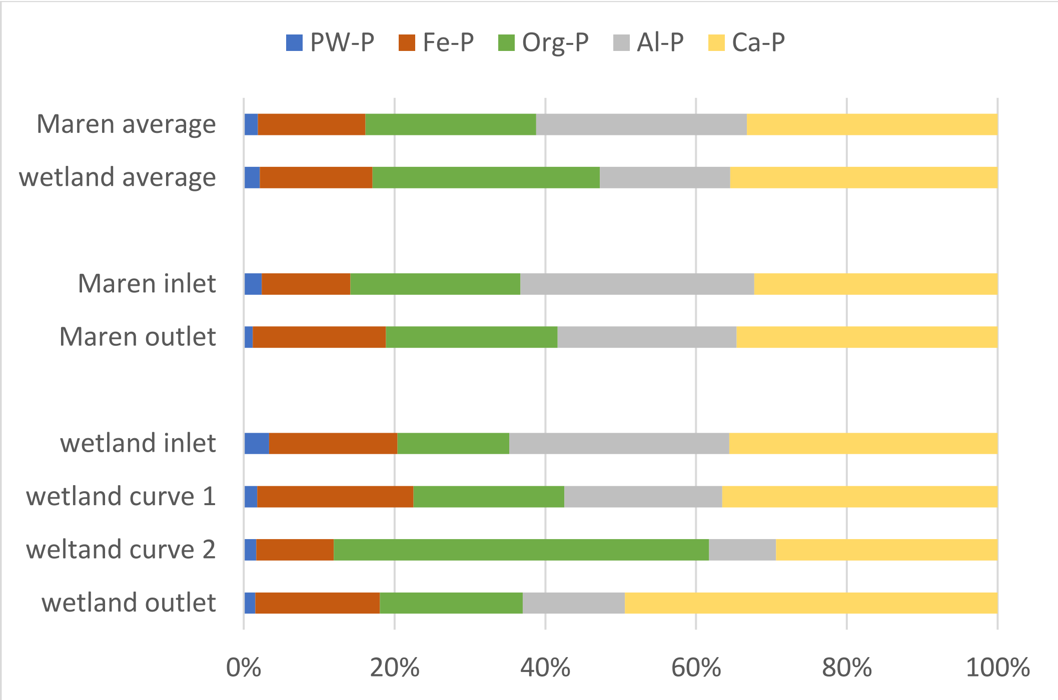


*Figure 7. The share of P fractions on the TP concentration per volume in the accumulated 3 cm sediment in the wetland Vilhelmsberg and 12 respectively 9 cm in Maren's inlet respectively outlet. P fractions include P bound to calcium (Ca-P), iron (Fe-P) or aluminium (Al-P) and P present in porewater (PW-P) or organic material (Org-P).*

In both waterbodies, on average Ca-P had the highest absolute and relative values of all P fractions, followed by Al-P in Maren and Org-P in the wetland Vilhelmsberg. Within the wetland Vilhelmsberg, the absolute Ca-P and Org-P concentrations were highest at the second curve, even though it had the lowest share of Ca-P (29 %) of all sampling points.

Both Ca-P and Org-P were significantly correlated with the TP concentration per mass and per volume (Figure 8 and Figure 9), but no significant correlation could be found with the other P fractions or organic matter. The correlation between Org-P and TP concentration was partly influenced by the very high value at one sampling point. By disregarding this outlying sampling point at the second curve, the relationship between Org-P and TP concentration per volume was not significant any longer ($r = 0.68$; $R^2 = 0.29$; $p = 0.202$; $n = 5$), whereas per mass it got even stronger ($r = 0.99$; $R^2 = 0.98$; $p = <0.001$.; $n = 5$).

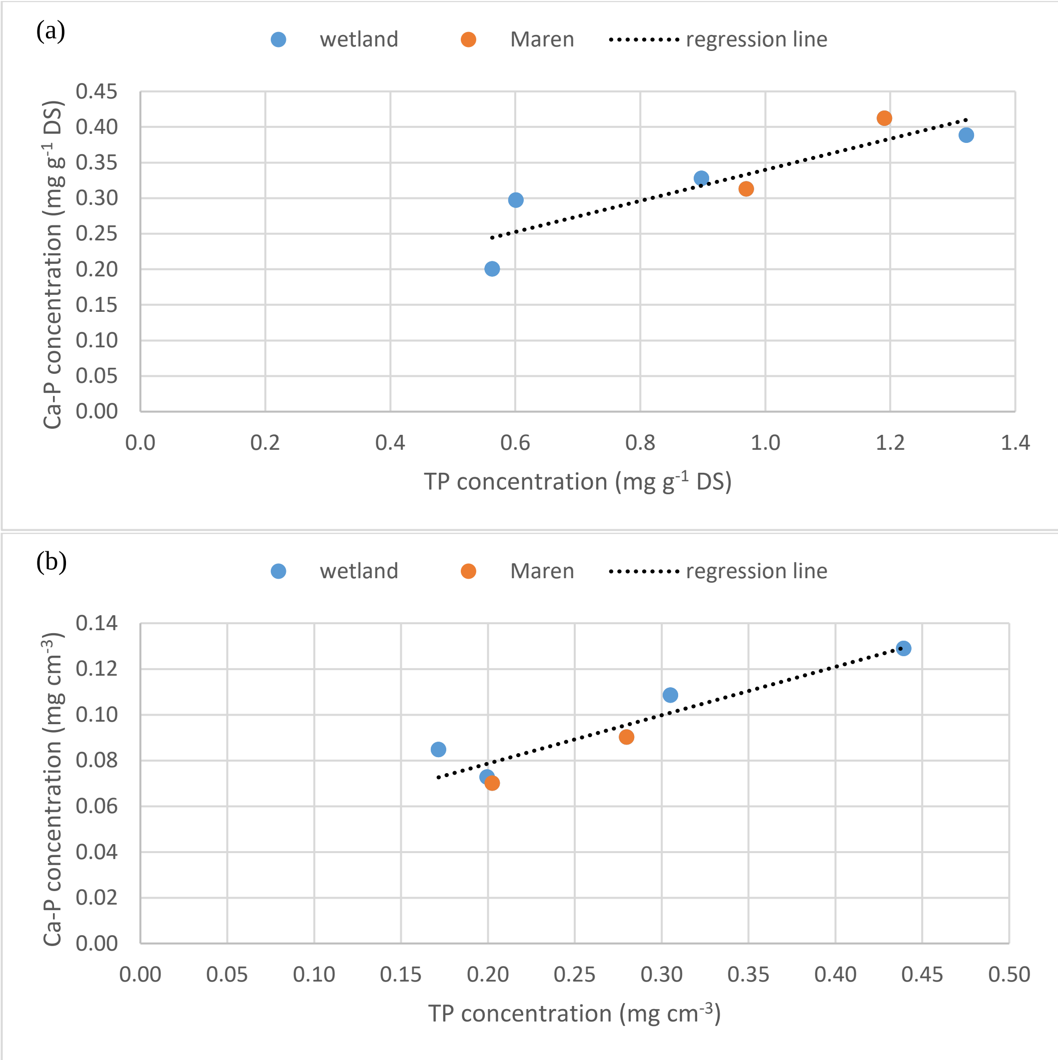


*Figure 8. Linear regression between TP concentration and Ca-P concentration (a) per mass ($r$ = 0.90, $R^2$=0.73, $p$ = 0.018, $n$ = 6) and (b) per volume ($r$ = 0.93, $R^2$=0.83, $p$ = 0.007, $n$ = 6)*

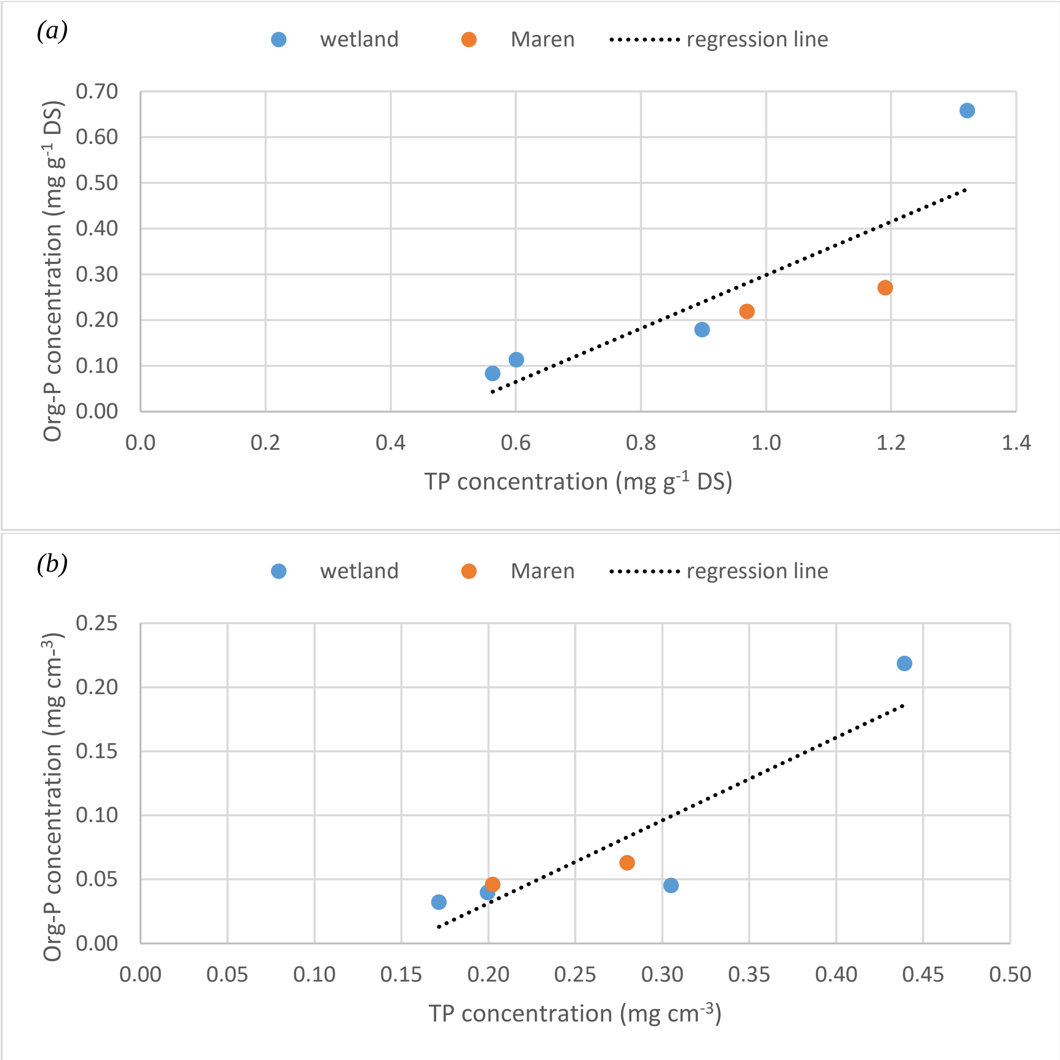


*Figure 9. Linear regression between TP concentration and Org-P concentration (a) per mass ($r = 0.85$; $R^2 = 0.65$; $p = 0.031$; $n = 6$) and (b) per volume ($r = 0.90$; $R^2 = 0.76$; $p = 0.015$; $n = 6$)*

Even though the proportion of organic matter was highest at the outlet of the wetland Vilhelmsberg, this sampling point had low Org-P concentration per mass and the lowest Org-P concentration per volume. Contrary to that, the second curve of the wetland Vilhelmsberg had the lowest organic matter content, but the highest Org-P concentration of all six sampling points. However, this pattern was not constant throughout the wetland, because the inlet had the second highest values of both the organic matter content and Org-P concentration per volume. Overall, there was no significant correlation found between organic matter content and Org-P concentration.

Generally, in both waterbodies at the inlet compared to the outlet, the organic matter content was lower, though the Org-P concentration per volume was higher. However, this was probably influenced by the higher densities at the inlets compared to the respective outlets since the Org-P concentrations per mass were lower at the inlets compared to the outlets.

### 3.2.2. Internal loading

The average mobile P (consisting of PW-P and Fe-P) concentration was 0.5 g $m^{-2}$ $cm^{-1}$ in the wetland Vilhelmsberg and 0.4 g $m^{-2}$ $cm^{-1}$ in Maren (Table 3). Calculated with Pilgrims' equation (17), the maximum potential P release rate based on mobile P was quite similar for the wetland Vilhelmsberg (6.5 mg $m^{-2}$ $d^{-1}$) and Maren (5.2 mg $m^{-2}$ $d^{-1}$). Within Maren, the P release rate was almost the same, whereas it varied more within the wetland Vilhelmsberg. It was higher at the inlet, followed by the second curve and then the first curve, whereas the outlet had the lowest P release rate. Taking the respective water area into consideration the annual P release rate was higher for Maren (63 kg $yr^{-1}$) than for the wetland Vilhelmsberg (37 kg $yr^{-1}$). The share of mobile P on TP was quite similar in both waterbodies. The mobile P accounted for 18 % of the TP concentration in the wetland Vilhelmsberg and 17 % of TP concentration in Maren. There was a tendency that mobile P concentration increased with TP, however there was no significant correlation.

*Table 3. Mobile P, labile Org-P and maximum potential P release rate on average and at the different sampling points for Maren and the wetland Vilhelmsberg*

| | Maren | | | wetland Vilhelmsberg | | | | |
|---|---|---|---|---|---|---|---|---|
| | average | inlet | outlet | average | inlet | curve 1 | curve 2 | outlet |
| mobile P (g $kg^{-1}$ DS) | 0.18 | 0.14 | 0.23 | 0.15 | 0.12 | 0.20 | 0.16 | 0.11 |
| mobile P (mg $cm^{-3}$) | 0.04 | 0.04 | 0.04 | 0.05 | 0.06 | 0.05 | 0.05 | 0.03 |
| mobile P (g $m^{-2}$ $cm^{-1}$) | 0.39 | 0.40 | 0.38 | 0.48 | 0.62 | 0.45 | 0.52 | 0.31 |
| P release rate (mg $m^{-2}$ $d^{-1}$) | 5.2 | 5.3 | 5.1 | 6.5 | 8.7 | 6.1 | 7.2 | 4.0 |
| labile Org-P (g $kg^{-1}$ DS) | 0.18 | 0.16 | 0.19 | 0.19 | 0.06 | 0.13 | 0.47 | 0.08 |
| labile Org-P (mg $cm^{-3}$) | 0.04 | 0.05 | 0.03 | 0.06 | 0.03 | 0.03 | 0.16 | 0.02 |

From the potentially available P (consisting of PW-P, Fe-P and labile Org-P), PW-P made up the smallest part in each sample (Figure 10). On average Maren and the wetland Vilhelmsberg had quite similar potentially available P concentrations. Due to different densities the potentially available P concentration in the wetland Vilhelmsberg was slightly lower per mass and slightly higher per volume, compared to Maren. Within the wetland Vilhelmsberg, the labile Org-P fraction was almost always lower than the Fe-P fraction except at the second curve that had a higher amount of labile Org-P. Within Maren, the Fe-P was slightly higher than the labile Org-P at the outlet, whereas it was vice versa at the inlet, most likely due to the emergent vegetation close to the inlet of Maren. Overall, it was found that the available P concentration was positively correlated with TP concentration for both per mass and per volume (Figure 11).

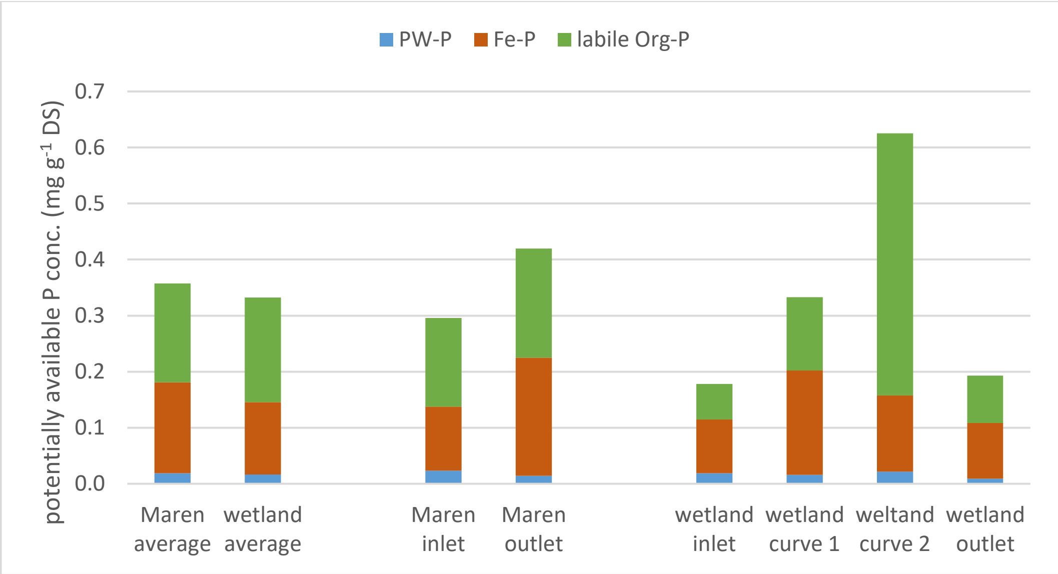


*Figure 10. Concentration per mass of potentially available P fractions in the sediment on average and at the different sampling points for the wetland Vilhelmsberg and Maren*

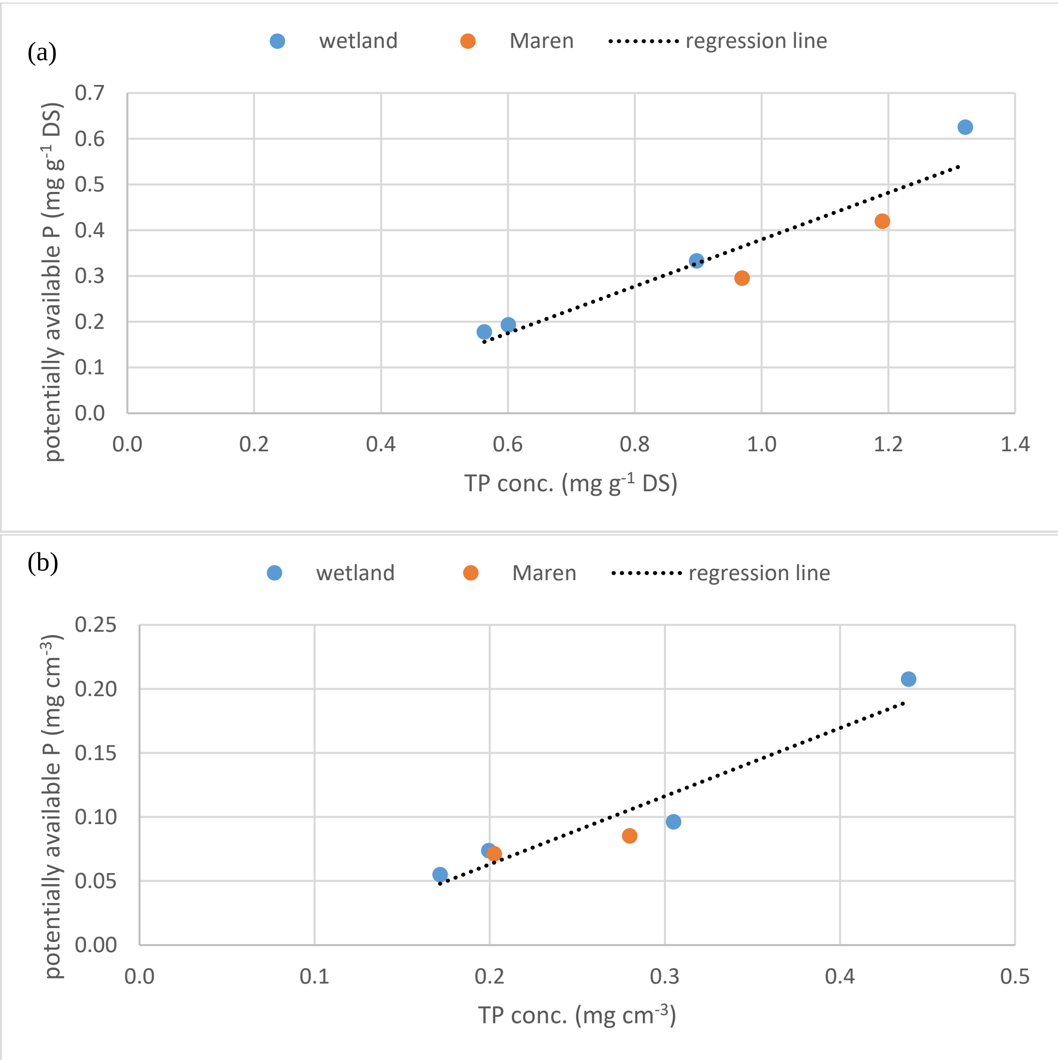


*Figure 11. Linear regression between potentially available P concentration and TP concentration (a) per mass ($r = 0.94$; $R^2 = 0.86$; $p = 0.005$; $n = 6$) and (b) per volume ($r = 0.95$; $R^2 = 0.88$; $p = 0.003$; $n = 6$)*

## 3.3. Qualification of the sediments for recycling

### 3.3.1. Metal content

In all samples collected at the wetland Vilhelmsberg the contents of all metals (Cd, Cr, Cu, Hg, Ni, Pb, Zn) that are listed in the ordinance (1998:944) were below the maximum allowed limit value. This means that the sludge is allowed to be transferred for agricultural purpose regarding the metal content (Table 4). All metal contents in Maren were below the maximum allowed limits except nickel, which means that it is not allowed to transfer the sludge from Maren for agricultural purposes.

*Table 4. Metal contents (g $kg^{-1}$ DS) in the sediment at the different sampling points and maximum allowed limit value (g $kg^{-1}$ DS) for each metal according to ordinance (1998:944)*

| | | Maren | | wetland Vilhelmsberg | | | |
|---|---|---|---|---|---|---|---|
| | legal limit | inlet | outlet | inlet | curve 1 | curve 2 | outlet |
| metal content (g $kg^{-1}$ DS) | | | | | | | |
| Cd, cadmium | 2.0 | 1.5 | 1.6 | 0.4 | 0.7 | 0.5 | 1.1 |
| Cr, chromium | 100.0 | 49.3 | 50.4 | 49.1 | 59.3 | 48.0 | 47.4 |
| Cu, copper | 600.0 | 35.9 | 42.6 | 33.6 | 45.1 | 33.8 | 40.1 |
| Hg, mercury | 2.5 | <0.2 | <0.2 | <0.2 | <0.2 | <0.2 | <0.2 |
| Ni, nickel | 50.0 | 80.2 | 75.4 | 37.0 | 49.0 | 40.2 | 37.9 |
| Pb, lead | 100.0 | 25.2 | 24.6 | 20.4 | 21.9 | 18.1 | 22.9 |
| Zn, zinc | 800.0 | 209.0 | 204.0 | 105.0 | 148.0 | 105.0 | 119.0 |

The residual analysed metals (As, Ba, Co, V) are not listed in the ordinance (1998:944), however, when comparing them with not-legally binding reference values for sensitive soils, only vanadium was below the reference value in all sampling points at Maren and the wetland Vilhelmsberg (Table 5). The barium content was only above the reference value at the inlet of the wetland Vilhelmsberg, while the arsenic content was only above the reference value at the second curve. The cobalt content was mostly above the reference value in Maren and in the wetland Vilhelmsberg, except the outlet where the analysed metal content exactly meet the reference value.

*Table 5. Metal contents (g $kg^{-1}$ DS) of residual analysed metals that are not listed in the ordinance (1998:944) and recommended values for sensitive soils (Swedish EPA 2009)*

| | | Maren | | wetland Vilhelmsberg | | | |
|---|---|---|---|---|---|---|---|
| | guideline | inlet | outlet | inlet | curve 1 | curve 2 | outlet |
| metal content (g $kg^{-1}$ DS) | | | | | | | |
| As, arsenic | 10.0 | 5.3 | 7.3 | 5.4 | 5.9 | 14.8 | 5.9 |
| Ba, barium | 200.0 | 120.0 | 146.0 | 832.0 | 110.0 | 113.0 | 84.8 |
| Co, cobalt | 15.0 | 37.1 | 38.8 | 16.1 | 21.1 | 17.2 | 15.0 |
| V, vanadium | 100.0 | 69.4 | 62.9 | 62.4 | 68.8 | 58.1 | 53.0 |

Considering the maximum mass of metals that are allowed in sewage sludge that is spread per ha agricultural land, nickel is the most limiting metal (Table 6). The maximum allowed amount of sludge applied on agricultural fields if it is taken only from the inlet of the wetland Vilhelmsberg will be 2.1 $m^3$ $ha^{-1}$ $yr^{-1}$ over a period of 7 years. This means if the sludge gets applied only once in 7 years, maximal 14.7 $m^3$ $ha^{-1}$ will be allowed. If the sludge is taken from the entire wetland and mixed, the average nickel concentration leads to a maximum allowed amount of sludge of 2.6 $m^3$ $ha^{-1}$ $yr^{-1}$ for spreading on agricultural soils over a period of 7 years respectively 18 $m^3$ $ha^{-1}$ once in 7 years.

*Table 6. Maximum allowed amount of metals in sewage sludge that is added to agricultural soils according to regulation SNFS 1994:2 (with amendment regulations 1998:4, 2001:5) and the resulting allowed amount of sludge spreading from the wetland Vilhelmsberg based on different metal content*

| | legal limit | sludge from wetland Vilhelmsberg ($m^3$ $ha^{-1}$ $yr^{-1}$) | | | | |
|---|---|---|---|---|---|---|
| | ($g$ $ha^{-1}$ $yr^{-1}$) | average | inlet | curve 1 | curve 2 | outlet |
| regulated metals | | | | | | |
| Cd, cadmium | 0.75 | 4.9 | 5.4 | 5.3 | 5.8 | 3.7 |
| Cr, chromium | 40.00 | 3.3 | 2.5 | 3.4 | 3.2 | 4.6 |
| Cu, copper | 300.00 | 33.6 | 27.8 | 33.7 | 34.5 | 40.7 |
| Hg, mercury | 1.50 | 31.3 | 23.4 | 38.0 | 29.1 | 40.8 |
| Ni, nickel | 25.00 | 2.6 | 2.1 | 2.6 | 2.4 | 3.6 |
| Pb, lead | 25.00 | 5.1 | 3.8 | 5.8 | 5.4 | 5.9 |
| Zn, zinc | 600.00 | 21.5 | 17.8 | 20.6 | 22.2 | 27.4 |

### 3.3.2. Soil amendment

On average, the wetland Vilhelmsberg had 0.4 % TN, 4.1 % TC and 10.5 mg P-AL per 100 g DS. Whereas Maren had slightly higher values on average for all three parameters: 0.5 % TN, 7.7 % TC and 12.7 mg P-AL per 100 g DS. Still, the average P-AL of the wetland Vilhelmsberg was twice as high as the average P-AL measured in the soil of the catchment in 2012. This means that in terms of P-AL the sediment will be a good soil amendment in the catchment.

Within the wetland Vilhelmsberg P-AL was highest at the first curve, followed by the outlet (Figure 12). Contrary to that, TN and TC were both highest at the outlet, followed by the first curve. The inlet of the wetland Vilhelmsberg showed the lowest values of all three parameters, shortly behind the second curve. Within Maren all three parameters were higher at the outlet than at the inlet.

A decreasing trend was found between P-AL and clay content, however it was slightly above the significance level ($r = -0.80$; $R^2 = 0.55$; $p = 0.056$; $n = 6$). No significant correlation was found between P-AL and TP concentration.

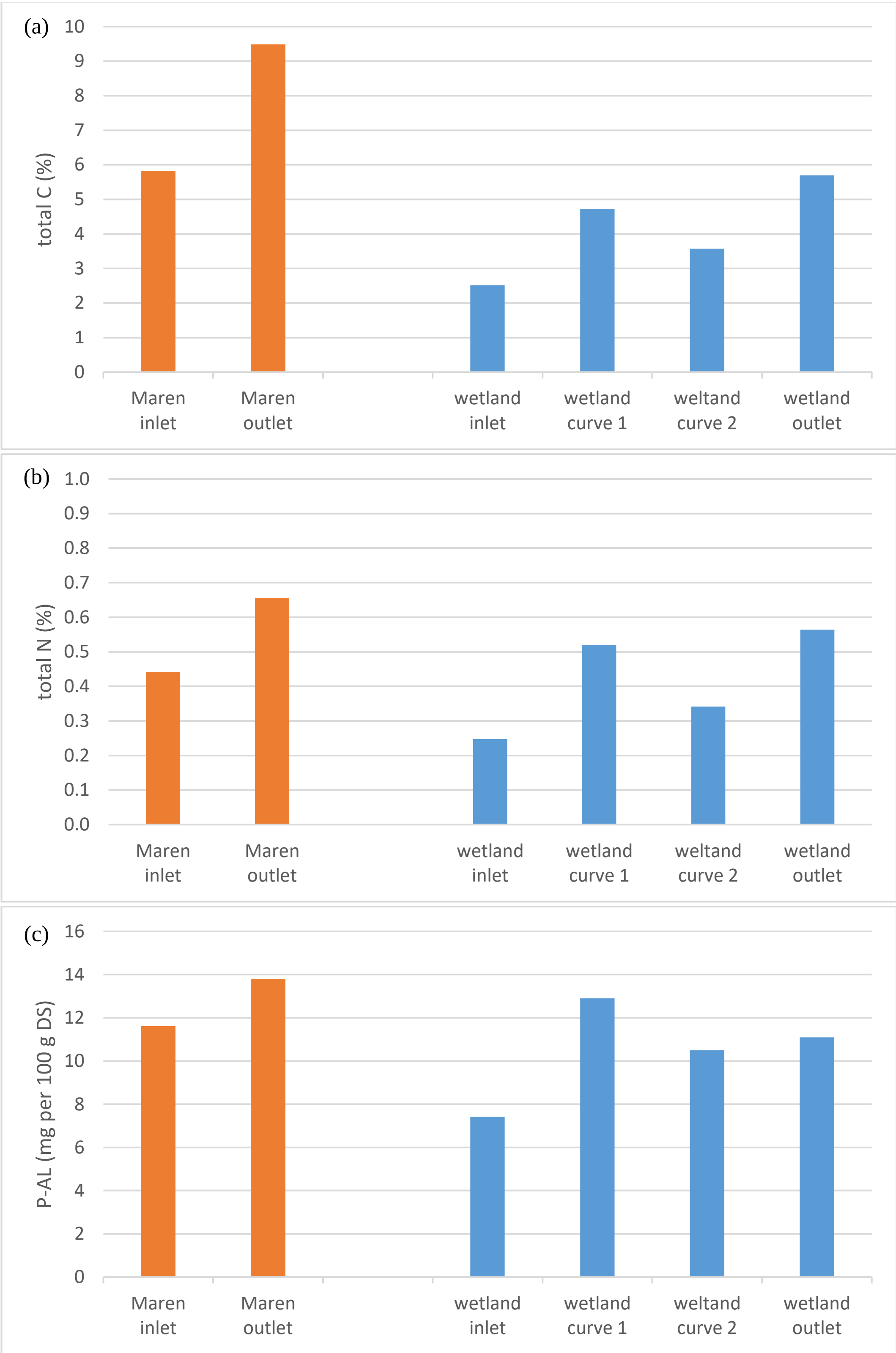


*Figure 12. Content of (a) TC, (b) TN and (c) P-AL at the different sampling points in Maren and wetland Vilhelmsberg.*

## 3.4. Theoretical P load and retention

### 3.4.1. Theoretical P load from land

The P load from land was estimated for the entire catchment of the wetland Vilhelmsberg, but also divided into Maren's catchment and the residual area (Table 7). Generally, the P leaching concentrations from arable land are 30 times higher than from forests. Due to the higher P concentrations, it was estimated that arable land accounted for over 80 % of P load from land. Although forested land took up the most space, it only accounted for less than 15 % of the P load from land. Minor parts of the P load originated from open land with less than 4 %, mires with less than 2 % and clear-cuts with less than 1 %.

*Table 7. Land P load depending on the land cover and respective P leaching concentration for the catchment of the wetland Vilhelmsberg and Maren and the residual area between the wetland Vilhelmsberg and Maren*

| | area (ha) | P conc. (mg $l^{-1}$) | P load (kg $yr^{-1}$) | P load (%) |
|---|---|---|---|---|
| wetland Vilhelmsberg's catchment | | | | |
| arable land | 211.6 | 0.390 | 186.6 | 84.83 |
| clear cut | 1.7 | 0.021 | 0.1 | 0.04 |
| forest | 933.4 | 0.013 | 27.4 | 12.48 |
| mire | 24.5 | 0.013 | 0.7 | 0.33 |
| open land | 94.3 | 0.024 | 5.1 | 2.33 |
| Maren's catchment | | | | |
| arable land | 207.1 | 0.390 | 182.7 | 84.90 |
| clear cut | 1.7 | 0.021 | 0.1 | 0.04 |
| forest | 910.5 | 0.013 | 26.8 | 12.44 |
| mire | 22.7 | 0.013 | 0.7 | 0.31 |
| open land | 91.4 | 0.024 | 5.0 | 2.31 |
| residual area of the wetland's catchment | | | | |
| arable land | 4.5 | 0.390 | 3.9 | 81.68 |
| clear cut | 0.0 | 0.021 | 0.0 | 0.00 |
| forest | 22.9 | 0.013 | 0.7 | 13.94 |
| mire | 1.8 | 0.013 | 0.1 | 1.12 |
| open land | 2.9 | 0.024 | 0.2 | 3.26 |

### 3.4.2. Theoretical P load from domestic wastewater

The P coming from domestic wastewater consisted of discharges from individual houses and a WWTP located within the catchment. It was estimated that individual houses in total amount to 53 PE. Taking typical daily amounts of wastewater production (170 l $PE^{-1}$ $d^{-1}$) and P concentrations in wastewater (10 mg $l^{-1}$ $PE^{-1}$ $d^{-1}$) into consideration it was estimated that individual houses in total produced about 9 $m^3$ $d^{-1}$ wastewater with a P concentration of 10 mg $l^{-1}$ and a P load of 90 g $d^{-1}$

(Table 8). By assuming a treatment reduction of 70 % it was estimated that the treated wastewater from the individual houses had a P concentration of 3 mg $l^{-1}$. Considering the amount of wastewater (9,010 l $d^{-1}$), the P load from all individual houses together was estimated to be 27,030 mg $d^{-1}$ respectively 9.9 kg $yr^{-1}$.

The measured P concentrations in the outflowing treated wastewater of the WWTP were higher in 2019 and lower in 2020. The average P concentration of the two years was 3.7 mg $l^{-1}$. This resulted in an average P load of 15,399 mg $d^{-1}$ respectively 2.1 kg $yr^{-1}$ by considering estimated wastewater amounts of wastewater (1,560 l $d^{-1}$). Compared to individual houses, the WWTP had a smaller P load in the treated wastewater, however it also included the wastewater of less PE. By considering given PE for black and greywater and typical produced wastewater amounts (l $PE^{-1}$ $d^{-1}$), the total amount of wastewater was estimated about 1.5 $m^3$ $d^{-1}$. Additionally, by considering typical P concentrations in untreated black and greywater, it was estimated that the P concentration in the total untreated wastewater is 9.8 mg $l^{-1}$. Hence, it was estimated that the average treatment reduction of the WWTP of 2019 and 2020 was 63 %, which is lower than the recommended treatment reduction.

*Table 8. Estimated P loads from domestic wastewater separated into the WWTP and the sum of individual houses within the catchment. (BW = blackwater; GW = greywater; $Q_{WW}$ = amount of wastewater; red. = P treatment reduction)*

| | PE | $Q_{WW}$ (l $PE^{-1}d^{-1}$) | P conc. (mg $l^{-1}PE^{-1}d^{-1}$) | P load (mg $d^{-1}$) | $Q_{WW}$ (l $d^{-1}$) | P conc. (mg $l^{-1}$) | red. (%) | P load (kg $yr^{-1}$) |
|---|---|---|---|---|---|---|---|---|
| WWTP | | | | | | | | |
| inflow BW | 9.00 | 170 | 10.0 | 15300 | 1530 | | | |
| inflow GW | 0.25 | 120 | 1.3 | 39 | 30 | | | |
| inflow total | | | | 15339 | 1560 | 9.8 | | |
| outflow 2019[a] | | | | | | 6.3 | 36 | 3.6 |
| outflow 2020[a] | | | | | | 1.1 | 89 | 0.6 |
| individual houses | | | | | | | | |
| inflow | 53.00 | 170 | 10.0 | 90100 | 9010 | 10.0 | | |
| outflow | | | | 27030 | 9010 | 3.0 | 70[b] | 9.9 |

*a. The outflow P concentrations (mg $l^{-1}$) were measured by Eurofins Environment Testing Sweden AB in 2019 and 2020. In this study those values were combined with the estimated amounts of wastewater (l $d^{-1}$) in order to obtain the outflowing P load (kg $yr^{-1}$).*

*b. 70 % is the recommended treatment reduction for a normal level of protection in the general advice from the and from Swedish EPA (NFS 2006:7) on treating domestic wastewater from individual houses.*

In order to assess the impact of domestic wastewater on the P load better, the composition of the different P load origins within the upstream catchment of the wetland Vilhelmsberg are presented in Figure 13.

It was evaluated that the WWTP only accounted for 1 % of the entire P load from the upstream catchment and individual houses accounted for 4 %. The main part of the P load originated from arable land that accounted for 80 %, followed by forests with 12 % and open land with 2 %. Other land cover categories like clear-cuts and mires were below 1 %.

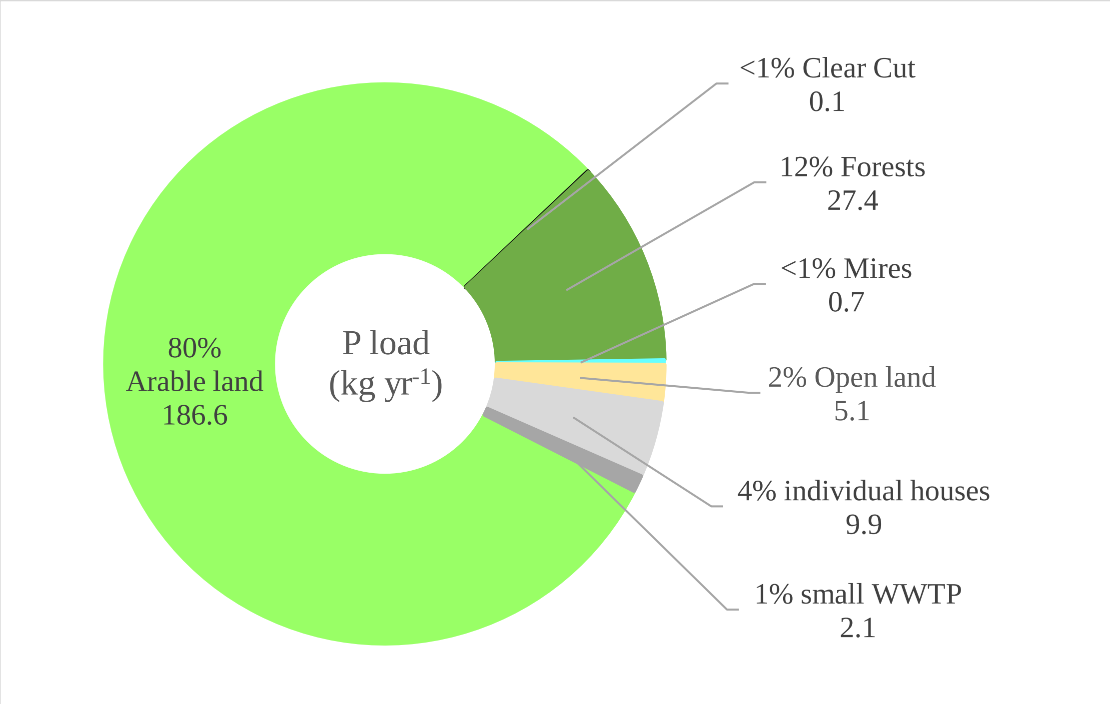


*Figure 13. Composition of the P load origins within the catchment of the wetland Vilhelmsberg in absolute amounts (kg yr$^{-1}$) below the labels.*

It needs to be considered that these estimations only included P load from the up-stream catchment. However, an uncertain part of the P load might also come from the Baltic Sea because there is no barrier at the wetland's outlet.

### 3.4.3. Theoretical P retention

The theoretical estimated annual P load to the wetland Vilhelmsberg would have been 232 kg yr$^{-1}$ by neglecting Maren (Figure 14). However, 98% of the P load (227 kg yr$^{-1}$) flew through Maren where it is subject to retention. Based on the the-oretical P load (equation 13), it was estimated that Maren retained 99 kg P yr$^{-1}$ and transferred 128 kg P yr$^{-1}$ further to the wetland Vilhelmsberg. Hence, Maren re-tained 43 % of the P load that the wetland Vilhelmsberg would have received. Tak-ing the retention of Maren into consideration, the wetland Vilhelmsberg received a P load of 133 kg yr$^{-1}$.

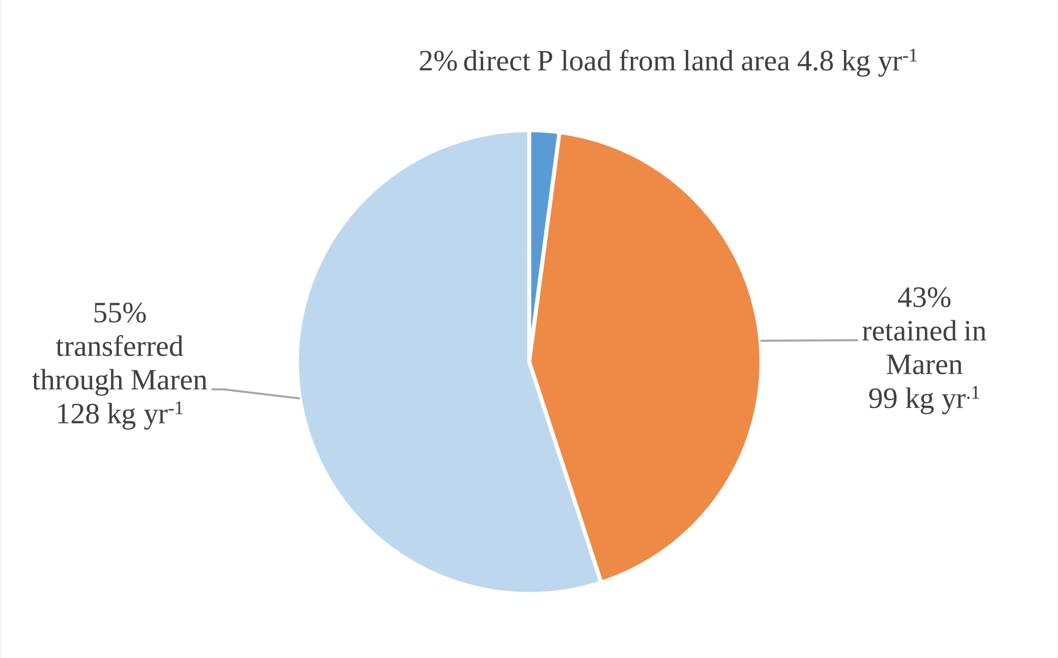


*Figure 14. P load of the entire catchment including retention and transfer trough Maren*

Even though the annual P load is higher for Maren (227 kg $yr^{-1}$) than for the wetland Vilhelmsberg (133 kg $yr^{-1}$), the area-specific P load for Maren is lower due to a larger water area (Table 9). Based on the area-specific P load it was estimated that the theoretical P retention of Maren (30 kg $ha^{-1}$ $yr^{-1}$) was smaller than that of the wetland Vilhelmsberg (37 kg $ha^{-1}$ $yr^{-1}$) (Table 9).

*Table 9. Overview of the theoretical P load and retention of Maren and the wetland Vilhelmsberg*

| | Maren | wetland Vilhelmsberg |
|---|---|---|
| theoretical P load (kg $yr^{-1}$) | 227 | 133 |
| theoretical area-specific P load (kg $ha^{-1}$ $yr^{-1}$) | 68 | 85 |
| theoretical area-specific P retention (kg $ha^{-1}$ $yr^{-1}$) | 30 | 37 |

# 4. Discussion

Wetlands will be essential to mitigate pollution of watercourses, if they function properly (Braskerud et al. 2000). The main purpose of this study was to assess the functioning of the wetland Vilhelmsberg. Commonly, many different factors influence the effective functioning of wetlands like HL, P load, size, shape and position of the wetland. For wetlands with the aim of P retention a key indicator for its functioning is the accumulation of particles and associated P.

## 4.1. Accumulation of particles and P

Overall, it was found out that the wetland Vilhelmsberg functions as a P trap, however the particle and P accumulation was low (16 t $ha^{-1}$ $yr^{-1}$ and 13 kg $ha^{-1}$ $yr^{-1}$) compared to other Swedish wetlands. For instance ten wetlands in the same province Södermanland estimated a mean annual accumulation between 7 and 130 t $ha^{-1}$ $yr^{-1}$ for particles and between 8 and 96 kg $ha^{-1}$ $yr^{-1}$ for P (Corbee 2021). Compared to that the wetland Vilhelmsberg would be in the lower range, but still showing a higher P accumulation than three out of the ten investigated wetlands. Another investigation of seven wetlands in Southern Sweden by Johannesson et al. (2015) estimated that the particle accumulation ranged from 13 to 108 t $ha^{-1}$ $yr^{-1}$, while P accumulation varied between 11 and 175 kg $ha^{-1}$ $yr^{-1}$. Only one of the seven studied wetlands (Wiggeby) had an even lower particle and P accumulation than the wetland Vilhelmsberg. Similarities of both wetlands were a high HL and that they were in the same leaching region with high clay content (33 to 52 %) in the catchment's arable topsoil. The estimated HL of the wetland Vilhelmsberg (192 m $yr^{-1}$) was higher than the modelled HL in five of seven wetlands investigated by Johannesson et al. (2015) where HL ranged from 22 to 646 m $yr^{-1}$. The additional water inflow of the Baltic Sea probably led to an even higher HL of the wetland Vilhelmsberg and could lead to resuspension of settled particles. This was considered a main reason for the low accumulation, since too high HL can negatively affect the P accumulation (Kynkäänniemi 2014). For instance, high flow events during a year can lead to limited water retention times, which results in lower particle settling (Kynkäänniemi et al. 2013). A high HL can on the other hand bring more aggregated particles from the catchment that have faster settling velocities (Braskerud 2003). In order to prevent the breaking up of aggregates, it is important that wetlands are located closer to the agricultural fields (Strand & Weisner 2011). In the case of the wetland Vilhelmsberg, the position might be sub-optimal for high nutrient retention because Maren receives most particles before they enter the wetland. Preferably, the wetland should have been placed upstream of Maren to mitigate the influence of the Baltic Sea and to capture more particles. A common reasons for low P accumulation in wetlands is that the landowners choose the position of the wetland that might be sub-optimal for high nutrient retention (Strand & Weisner 2011).

Another reason for the comparatively low particle and P accumulation might also be that the soil texture in the catchment of the wetland Vilhelmsberg is mainly clay (52 %). Johannesson et al. (2015) found a negative correlation between clay content in the catchment top soil and accumulation of particles and P. Often the clay content in the top soil of arable land is reflected in the sediment of wetlands (Braskerud et al. 2000). The sediment at the inlet of the wetland Vilhelmsberg had a quite similar clay content (47 %) to the catchment top soil of the arable land (52 %), indicating that it captured soil particles from its catchment. Most clay particles settled at the inlet, while towards the outlet proportionally less clay particles were in the sediment, indicating that it is long enough for settling of small particles. Contrary, if the proportion of clay particles in wetlands was higher at the outlet than at the inlet, it would indicate too little time to settle for smaller particles (ibid.). However, clay particles may also had been resuspended due to ocean water inflow, since smaller particles are more prone to resuspension (Johannesson et al. 2015). Resuspension of particles will also more likely occur with changing water velocities and if there is no vegetation (ibid.), which applied to the wetland Vilhelmsberg. Furthermore, fish could easily enter the wetland Vilhelmsberg since there is no barrier to the Baltic Sea. Bioturbation is likely to be caused by fish that feed on the sediment leading to resuspension of settled particles. In the wetland Vilhelmsberg, for instance pikes were spotted that act as predator species for unwanted species like carps that disturb the sediment in P wetlands (Ellis et al. 2003).

The accumulation of P also depends on the P load, which probably got lowered due to retention of P in Maren. The P load was not measured in this study, but it was estimated theoretically. Comparing the theoretical P load of the wetland Vilhelmsberg (85 kg ha$^{-1}$ yr$^{-1}$) from the upstream catchment with other theoretical P loads of wetlands in Sweden it lied within the middle range (Corbee 2021) or lower range (Johannesson et al. 2015). The estimated theoretical P retention (37 kg ha$^{-1}$ yr$^{-1}$) based on theoretical P load with an equation from Weisner et al. (2016) was higher than the measured P accumulation (13 kg ha$^{-1}$ yr$^{-1}$) of the wetland Vilhelmsberg. This discrepancy is reasonable because the equation did not include other factors like the influence of the Baltic Sea and was only based on the theoretical P load. Johannesson et al. (2015) argues that often theoretical P loads were higher than actual P loads measured in the water.

Despite low P accumulation, the TP concentration per volume of the wetland Vilhelmsberg was higher than in six out of ten wetlands studied by Corbee (2021). Especially, the sediment at the inlet that showed a very high proportion of clay particles had a high TP concentration per volume despite low TP concentration per mass. Lannergård et al. (2020) explained that due to a higher share of clay particles in the sediment, the water content can be lower and TP concentrations per volume can be higher.

Overall, for improving accumulation of particles and P of the wetland Vilhelmsberg, it is necessary to consider design changes. The outlet of the wetland Vilhelmsberg is directly connected to the open Baltic Sea and rather broad (53 m), compared to the wetland's average width (20 m). The Baltic Sea underlies an annual cycle with high fluctuations in water level which is affected by wind and air pressure fields and linked to the water exchange with the North Sea, since the Baltic Sea is landlocked (Stramska et al. 2013). To limit the impact of the Baltic Sea on the wetland Vilhelmsberg, it is recommended to change the outlet design. Additionally, the wetland Vilhelmsberg had the same water depth (1 m) throughout the entire area. For increasing particle settling, it is recommended to construct a deeper part at the inlet or middle of the wetland. This should decrease water flow velocities and additionally, makes future maintenance like sediment removal easier. Often, wetlands with the aim of P retention are designed with a deeper initial area where most of the particles can settle followed by a longer shallow area with emergent vegetation that can act as a filter for particles (Kynkäänniemi et al. 2013). In the case of the wetland Vilhelmsberg a shallow section with emergent vegetation is hardly possible due to the direct transition to the Baltic Sea. However, it could be established together with a barrier at the outlet or in cross sections within the wetland after a deeper part.

## 4.2. Internal loading

For an effective wetland management it is also essential to estimate the risk of internal loading in order to prevent nutrient transport to downstream waterbodies (Pant 2020). It is known that sediment can act as sink or source of P. The release of P is a dynamic processes that changes in time and space and is regulated by biotic processes like the mineralization of organic matter and abiotic processes like adsorption and desorption (Pant 2020). Due to an enhanced release of P, sediments can be an internal P source and the removal of sediments is important to reduce internal loading. In particular, the internal P load is of major interest in lakes and can cause up to 80 % of the P load, where the external P load has already been curtailed (Palmer-Felgate et al. 2011).

The mobile P (Fe-P and PW-P) and the resulting maximum potential P release rate of Maren (5.2 mg $m^{-2}$ $d^{-1}$) and the wetland Vilhelmsberg (6.5 mg $m^{-2}$ $d^{-1}$) were small compared to 21 other Swedish wetlands (4.4 to 21.6 mg $m^{-2}$ $d^{-1}$) on clay soils around lake Mälaren (Michélsen 2021). While another investigation on coastal sediment of the Baltic Sea close to the wetland Vilhelmsberg estimated P release rates between 2.7 and 7.4 mg $m^{-2}$ $d^{-1}$ (Rydin et al. 2011). This corresponded with the wetland Vilhelmsberg and Maren, which means that the two waterbodies did not show extraordinary P release rates. The maximum potential P release rate represents the internal P load that could get released under anoxic conditions (Pilgrim et al. 2007).

It needs to be considered that the estimated P release rate was only based on mobile P and might be slightly higher, because over time Org-P can contribute to mobile P (ibid.). Nevertheless, the potentially available P (PW-P, Fe-P and labile Org-P) of the wetland Vilhelmsberg (0.33 g $kg^{-1}$ DS) and Maren (0.36 g $kg^{-1}$ DS) was much lower than in lakes (0.56 to 1.33 g $kg^{-1}$ DS) and lied in the lower range in the studied streams (0.02 to 0.99 g $kg^{-1}$ DS) investigated by Lannergård et al. (2020) in Sweden. Thus, the risk of internal loading of the two waterbodies Maren and the wetland Vilhelmsberg was considered to be small. Hence, there is no current need to remove the sediment regarding the risk of internal P loading.

Additionally, the proportion of the Al-P of both the wetland Vilhelmsberg (17 %) and Maren (28 %) was higher than in most wetlands (9 to 19 %) studied by Michélsen (2021). High proportions of Al-P minimize the risk of internal loading because Al-P is less likely to be released than other P fractions. Similarly, Ca-P is considered as stabile fraction. On average, Ca-P was the largest P fraction in the sediment with 30 mg per 100 g DS in the wetland and 36 mg per 100 g DS in Maren. This is probably influenced by agriculture in the catchment. Lannergård et al. (2020) found out that Ca-P in the sediment was the dominating P fraction in agricultural areas. The soil survey of arable land in the catchment showed on average 71 mg per 100 g DS for the parameter "P-HCl" which corresponds to Ca-P in this study. In particular, the field sampling point closest to the outlet of Maren was highly above average (90 mg per 100 g DS). This explains why the sediment at the outlet of Maren showed the highest absolute Ca-P concentration per mass of all six sediment sampling points (41 mg per 100 g DS).

The release of P stored in sediment is dominated by how mobile the prevailing P fraction potentially are and additionally influenced by the prevailing environmental conditions (Lannergård et al. 2020). Factors influencing the P exchange between sediment and water are pH, oxygen conditions, chemical conditions and P concentration in the sediment pore water (ibid.). Following conditions can lead to a P release of sediment, which can then be transported further downstream: (i) low oxygen content combined with Fe-P, (ii) degradation of organic material or (iii) lower P concentrations in the water column than in the sediment pore water (ibid.). Often, the risk of internal loading is controlled by a combination of Fe-P and labile Org-P (ibid).

(i) Sometimes redox induced mobilization of P bound to metal (hydr)oxides are considered as the most important reason for internal loading (Palmer-Felgate et al. 2011). Redox sensitive mobilization are generally likely to occur in the anoxic zones just below the surface of the sediments (Søndergaard et al. 2003). From all sampling points, the highest Fe-P concentration was analysed at the inlet of the wetland Vilhelmsberg, where also orange spots of precipitated iron could be observed while taking the samples. The average proportion of Fe-P fraction in the wetland Vilhelmsberg (15 %) is low compared to Fe-P found in lakes (15 to 37 %) studied by Lannergård et al. (2020). Maren even had slightly lower Fe-P concentrations per volume than the wetland Vilhelmsberg, which could be due to Maren's larger water depth where anoxic conditions were more likely, so that Fe-P had already been released.

(ii) The degradation of organic matter is generally influenced by characteristics of the organic matter itself, the microbial community and the external environmental conditions, for instance temperature, oxygen and salinity (Stagg et al. 2018). For instance, the degradation decreases with increasing salinity and water depth (ibid.). Even though the proportion of Org-P was very high at one sampling point of the wetland, it was still considered a typical value since Lannergård et al. (2020) found proportions of Org-P up to 75 % in lakes and up to 59 % in rivers.

(iii) Exchange processes of PW-P are influenced by the concentration of PW-P in sediment and the P concentration in the water column (Lannergård et al. 2020). Taking equilibrium P concentrations into consideration, stored P in sediment could be released, if the sediment has higher P concentrations than in the water column above (ibid.). In the wetland Vilhelmsberg, the PW-P concentration was smaller than all other P fractions. Therefore, it is considered that PW-P only has minor impacts on internal loading of the wetland Vilhelmsberg.

## 4.3. Maintenance need

Proper maintenance is essential for an optimal functioning of FWS wetlands (Chen 2011). Maintenance includes amongst others the inspection of inlet and outlet structures and where appropriate the removal of debris from inlets to ensure particle and water inflow (Verbyla 2016). Although no disturbances could be observed at the sampling dates, it is important to regularly inspect whether undisturbed water flow is possible. In particular, regular inspections are important after the outlet of Maren at a pipe where the water flows through under a road to ensure water flow.

In the long term, erosion should be inspected and stabilized if needed (ibid.). At the sampling date only slight signs of erosion were observed at the shores of the ditches in the areas before Maren as well as between Maren and the wetland. Whereas in the wetland itself no signs of erosion could be detected at the shores. Likewise, in Maren no erosion could be observed because of vegetation along the shores.

Sedimentation of particles is not everlasting and settled particles are prone to resuspension. Hence, after a certain time, sediments should be removed from wetlands to ensure further functioning of the wetland. The sludge of wetlands should be removed, when one quarter of the useful volume is full in order to ensure settling of suspended particles (Verbyla 2016). How long this takes usually depends on the type of wastewater and usually it will take longer the better the quality of the water is (ibid.). Considering the past sediment accumulation rate of 0.5 cm $yr^{-1}$ and a typical water depth of 1 m, one quarter of the wetland Vilhelmsberg (25 cm) would be reached 50 years after the wetland's construction. However, the relative water level can be lower due to influence of the Baltic Sea. Considering the lowest water level measured at Landsort Norra since the wetland's construction (-52 cm referred to the relative water level), one quarter of the wetland (12 cm) would be reached after 24 years since the wetland's construction. However, it needs to be considered that the wetland Vilhelmsberg is exposed to turbulence by fish, wind and ocean waves. In particular, extreme weather conditions or water fluctuations affect the functioning of FWS wetlands and decrease its long-term sustainability (Chen 2011). Often, after one or two decades accumulated sediments in a FWS wetland might become problematic (ibid.). Hence, it is not recommended to wait more than two decades since the wetland's construction for sediment removal. Moreover, increased age of wetlands negatively affects the retention of dissolved P but does not affect the retention of particulate P as much (Braskerud et al. 2005). Therefore, it is reasonable to wait further for sediment removal of the wetland Vilhelmsberg since larger parts of P load in the study area are in particulate form.

Options for the disposal of the sediments are incineration, land filling or recycling (Eriksson 2013). Sludge recycling in agriculture supports a transition towards circular economy. It is a great option for using less chemical fertilizers, because sludge is mostly nutrient rich. However, it needs to be considered that sludge can also contain harmful substances like heavy metals that can have negative consequences on plants, soil, animals or humans. Hence, the EU encourages the use of sludge for agriculture when it is appropriate and save, while on the same time regulating the use to protect the environment (Eriksson 2013).

### 4.3.1. Metal content

The European Sewage Sludge Directive (86/278/EEC) gives limit values for the annual amount of certain metals in sewage sludge added to agricultural soils as well as limit values on the metal content in soils and in sludge that is used for agriculture. The directive is over 30 years old and has not been revised or updated (Dagerskog & Olsson 2020). Hence, some countries inclusive Sweden implemented stricter limit values, which are referred to in this thesis. For instance, the Swedish ordinance on prohibition (SFS 1998:944) specified limit values for the metal content in sewage sludge that might be transferred or marketed for agricultural purposes. Another regulation from the Swedish EPA on the protection of the environment, in particular the soil when sewage sludge is used in agriculture (SNFS 1994:2, amendment regulations SNFS 1998:4, SNFS 2001:5) specified the maximum allowed amount of heavy metals in sewage sludge that is added to agricultural soils as well as limit values for the metal content in soils.

The metal analysis package from ALS Scandinavia covered all of the metals (Cd, Cr, Cu, Hg, Ni, Pb, Zn) that are specified in the mentioned European and Swedish legislations on sewage sludge use for agricultural purposes. The remaining analysed heavy metals (As, Ba, Co, V) that were part of the analysis package from ALS Scandinavia got compared with reference values that were developed by the Swedish EPA for remediation of contaminated soil (Swedish EPA 2009). Below these reference values the risk of negative effects on environment and humans is usually acceptable in the context of soil remediation (ibid.). It is important to mention that these reference values are not legally binding and are only used in this thesis for a rough estimation and interpretation of the analysed metal contents. The report by the Swedish EPA (2009) distinguished sensitive and less sensitive soils, whereof the master thesis only compared reference values for sensitive soils, which includes amongst others arable land.

In general, the maximum permitted metal contents defined in ordinance (1998:944) for sewage sludge are less strict than the reference values for sensitive soil that was remediated from soil contamination. This is probably because the sewage sludge gets applied in low amounts and mixed with uncontaminated soil. For example, although the nickel and cadmium content in the sediment samples from the wetland Vilhelmsberg meet the maximum allowed limits according to the ordinance (1998:944), the reference value for sensitive soil were exceeded at most sampling points. Hence, for comparison in this study the legally binding limits given in ordinance (1998:944) were considered more important than the non-legally binding recommendations for soil remediation.

In Maren, nickel was the only metal that was above the legally permitted limit. Even though in the wetland Vilhelmsberg, the nickel content was below the legally permitted limit, it was very close to it, especially at the first curve (49 mg $kg^{-1}$ DS). Many deposit sites of nickel were found in Sweden due to investigations of SGU and the Swedish Geological AB (Hallberg & Reginiussen 2019). The study site showed a high mining potential for some base metals including nickel, cobalt and iron according to SGU (2021b). This also gives a reason for the high cobalt contents in the sediment samples.

In one sampling point at the wetland Vilhelmsberg, the arsenic content was above the reference value for soil remediation. In soils, the arsenic content varies a lot, for instance ranging from 0.2 to 40 mg $kg^{-1}$ DS in Swedish non-contaminated soils (Forslund & Barregård n.d.). Arsenic is often naturally occurring in the bedrock, from where it can be dissolved and wander in the surrounding environment and water (ibid.). Hence, the measured arsenic contents at the sampling points were considered to be typical.

The barium content was 5 to 9 times higher at the inlet of the wetland Vilhelmsberg than at the other sampling points. Generally, barium can get released into the environment due to atmospheric deposition or the weathering of minerals and rocks (Liguori et al. 2016). In water, barium can be dissolved, attached to suspended particulate matter or precipitated in insoluble compounds (e.g. barium sulphate, BaSO4) (Gad 2014; Liguori et al. 2016). Barium that is attached to suspended particulate matter tends to migrate downwards in drainage basins in rivers or runoff water (Liguori et al. 2016). Due to sedimentation of suspended particulate matter a large proportion of barium from surface water gets stored in sediments (Gad 2014). Moreover, when a river enters an ocean, barium is very prone to precipitate due to higher contents of sulphate in marine water (Gad 2014; Liguori et al. 2016). The high barium content at the wetland's inlet indicated that the Sea water probably reached until this point. Typically, the barium content in anaerobic sewage sludge can range from 100 to 9000 mg $kg^{-1}$ DS (Nogueira et al. 2010). Due to application of sewage sludge, heavy metals like barium can accumulate in soils and potentially be available to plants (Abreu et al. 2012). Due to the fact that the ingestion of soluble barium is harmful to humans and animals, for instance the National Environment Council of Brazil published maximum barium levels of 1300 mg $kg^{-1}$ DS in sewage sludge that gets spread on agricultural soils (Nogueira et al. 2010). Despite the high barium content in the sediments at the inlet of the wetland Vilhelmsberg, it was still quite below that limit.

### 4.3.2. Soil amendment

Beneficial aspects of sludge application to agricultural land are on the one side nutrients that can improve soil fertility and on the other side organic matter that can improve soil structure.

The status of P-AL in agricultural soil gets divided into five classes ranging from very low P-AL (class I) to very high P-AL (class V) contents (Swedish EPA 1994). The analysed sediment of both waterbodies had a high P-AL content (class IV), while the arable top soil in the catchment had a low P-AL content (class II) according to the landowner's soil survey (Eurofins 2012). Thus, the sediment will probably enhance soil fertility regarding P-AL if it is returned to agricultural fields in the catchment. Based on the P-AL class of the catchment soil, there are limit values for the amount of P that is allowed to be added to agricultural fields via sewage sludge (Swedish EPA 1994). The amount of P in the wetland's sediment will be well below the legal limit if the suggested amounts of sludge regarding the metal contents get applied.

The organic matter of the wetland Vilhelmsberg was 18 % whereas in agricultural soils often it is between 3 and 6 % (Fenton et al. 2008). Organic matter plays a key role in to the soil productivity, some benefits are for instance that it improves water holding capacity and nutrient availability, enhances soil microbial biodiversity and reduces crusting of the soil surface and stickiness of clay soils (ibid.). The sediment of the wetland Vilhelmsberg will be a very good soil amendment with regards to its organic matter content.

Maren had higher TN and TC contents than the wetland Vilhelmsberg which is most likely influenced by the nutrient rich excreta of ducklings that are reared close to Maren and emergent vegetation along Maren's shores. The low TC:TN of the sediment of Maren (14) and the wetland (10) would be beneficial for the soil fertility if the sludge gets applied to agricultural fields. This is because soil microbes need adequate levels of C and N to live, with an optimum intake ratio of 24 (United States Department of Agriculture 2011). If applied material has a lower ratio, like the sediments from the wetland Vilhelmsberg, microbes will not use the entire N and leave it in the soil available for plants (ibid.). Contrary, it is not advisable to apply material with a very high ratio to soils because microbes will use additional N from the soil making it unavailable for plants (ibid.). The added sediments of the wetland Vilhelmsberg will also support the decomposition of plant residuals that have a high ratio (e.g. cereal straw or corn stover) by providing additional N (Fenton et al. 2008).

The possibility to recycle the sediments mainly focussed on the metal content and some quality factors like P-AL, TC:TN and organic matter. Other factors might also need to be considered before returning the sediments. For instance, the salt content was not analysed in this thesis, but it could increase the soil salt content when the sludge gets applied. However, the Baltic Sea is known for its brackish water nature due to high discharges of freshwater rivers and a small inflow from salty water from the North Sea. Hence, with a practical salinity unit of 7 close to the study area, the Baltic Sea has a much lower salt content than oceans typically have (Bryhn & Håkanson 2010).

Furthermore, there are some restrictions where sewage sludge is allowed to be applied on agricultural land according to SNFS 1994:2 that should be considered (Swedish EPA 1994). For instance, it is not allowed to spread sewage sludge on pastures; on land that currently cultivates vegetables, berries, potatoes or fruits; on arable land that cultivates forage crops that are harvested within ten months; and on arable land that will be used for grazing within the next ten months (ibid.). Also on land that is intended for future cultivation of food that is in direct contact with soil or consumed raw, sludge is not allowed to be applied ten months prior to harvesting.

## 4.4. Impact of domestic wastewater

In the general advice on small-scale wastewater system (HVMFS 2016:17), SwAM recommends P removal efficiencies of 70 % respectively 90 % for a normal respectively high level of protection, which corresponds to 3.0 respectively 1.0 g P $l^{-1}$ in the outflowing treated wastewater (SwAM 2016). The limit that was set for the outflowing water of the WWTP in the study area was even lower (0.3 mg P $l^{-1}$). Although the measured outflow P concentrations exceeded this limit in 2019 and 2020 (6.3 mg P $l^{-1}$ and 1.1 mg P $l^{-1}$), it was estimated that the WWTP had a negligible impact on the P load of Maren and the wetland Vilhelmsberg, accounting for only 1 % of the entire P load on average. Furthermore, the retention in the field ditch and Maren before the treated water of the WWTP reaches the wetland Vilhelmsberg should be mentioned. In case of high runoff more treated wastewater from the WWTP will flow towards Maren and the wetland Vilhelmsberg than in case of low runoff. This is because in the case of low runoff the treated wastewater will rather seep into the ground of the field ditch at the discharge location as it was observed at the sampling in middle of April (Figure 3).

When comparing the measured outflow P concentrations of the WWTP with the estimated inflow P concentration, a treatment reduction of 97 % would be necessary in order to be below the limit of 0.3 mg P $l^{-1}$. Treatment reductions of P for sludge separation followed by a soil bed are estimated to be between 25 and 75 % (Swedish EPA 2008). In order to ensure a 70 % P reduction soil-based systems can for instance be supplemented with chemical P precipitation or P filters (Swedish EPA 2008). According to Olshammar et al. (2015) a treatment systems consisting of sludge separator, soil bed and a supplemented P filter have an expected P treatment reduction of 75 to 95 %. This means even with advanced P treatment a reduction of 97 % can hardly be achieved.

Around one quarter of the 700,000 households in Sweden that have a small onsite wastewater treatment system do not comply with the legal requirements (André et al. 2016). The supervisory authority SwAM for small-scale wastewater systems for less than 200 people and gives advice on treatment requirements to municipalities that are the responsible authorities for its oversight (SwAM 2016). For the Nynäshamn municipality, where the study area is located, the local environment and health authority SMHOF (in Swedish: Södertörns miljö- och hälsoskyddsförbund) is responsible and for instance decides on which level of protection is necessary concerning the environmental and public health (Swedish EPA 2006; SwAM 2016). One criteria that a high level of protection should be applied is for instance that the treated wastewater get discharged directly in a ditch (Swedish EPA 2006), which is the case for the WWTP in the study area (Figure 3).

An additional approach to lower P concentrations from domestic wastewater is to use P free detergents and washing agents (Karlsson 2005). This is a sustainable way to decrease P concentrations in the inflowing and hence outflowing water of the WWTP.

## 4.5. Method uncertainties

Despite the low number of sampling points, regression lines between the six sampling points were established in order to detect whether there are relationships in the sediment of the study area. For statistical analyses it would be better to have a higher number of sampling points to establish more meaningful relationships.

### 4.5.1. Sediment depth

The average sediment depth at a certain sampling point could be calculated by using either (i) the measured sediment depth from outside of the collected sediment core tubes or (ii) the sediment depth received from slicing. The sediment depth measurements from the outside of the tubes were often uncertain because mostly there was no clear distinctive bottom layer. Additionally, the sediment depth that was visible from the outside of the sediment core tube could often not be measured exactly because it varied within one tube. Reasons therefore could be that sediment got stuck on the tube's wall when inserting it into the ground or that sediment flowed downwards along the tube's wall. Hence, using the measurements from the outside could overestimate the sediment depth. Calculating with the outside measurements, the wetland Vilhelmsberg would have a mean sediment depth of 4 cm and a sediment accumulation rate of 0.6 cm $yr^{-1}$. A better picture of the sediment structure could be attained by looking inside the tube while slicing the sediment core. However, this could lead also to an underestimation of the sediment accumulation rate due to compaction during slicing. Calculating with the sediment depth received from slicing, the wetland Vilhelmsberg had a mean sediment depth of 3.5 cm and a sediment accumulation rate of 0.5 cm $yr^{-1}$. Overall, those values were considered to be more precise and reliable and were used in this study.

Another point that should be mentioned is that the calculated particle accumulation within the wetland Vilhelmsberg was affected by the sediment depth that was used for analysing the dry fraction. The used sediment depth was 3 cm at each sampling point, whereas the sediment depth based on the collected tubes varied within the wetland and was 3.5 cm on average. Comparing the sampling points, the sediment depth was highest at the outlet (4 cm) and lowest at the second curve (3 cm), while the inlet and first curve had the same sediment depth (3.5 cm). Hence, using these sediment depths for calculating the particle accumulation would lead to a higher particle accumulation at the outlet (114 t $ha^{-1}$) than at the second curve (100 t $ha^{-1}$). While the first curve would still show the lowest value (78 t $ha^{-1}$) and the inlet would still show the highest particle accumulation (190 t $ha^{-1}$), however, with a greater distance to all other point. The ranking order of sampling points in terms of P accumulation in the wetland Vilhelmsberg would not change when using the average sediment depth of the collected tubes instead of the sediment depth from the P analysis.

### 4.5.2. Catchment area and P load

The national ground cover used in GIS in this study were from the year 2018. Therefore, changes in land use since 2018 have not been covered in this study. In the previous years, new wetlands were constructed in the catchment area, which might will lead to a lower P load to the wetland Vilhelmsberg in future.

The soil survey in the catchment area was performed in 2012 and the measured values might have changed over time. Especially, fields with a low nutrient status could have received a higher fertilizer input after the soil analysis was conducted. Nevertheless, these results were the only available ones about the soil status and still gave an insight into the soil nutrient status.

The P load and HL coming from the Baltic Sea were uncertain and not taken into account in this thesis. For calculating the P load of the upstream catchment, geographical locations of the land areas got neglected and only the proportion of each land cover area got considered. This approach is similar to the nutrient flow simulations of a model about hydrological predictions for the environment (HYPE)

(SMHI 2021b). Likewise, the geographical location of the WWTP and the individual houses within the catchment was not taken into account, however, they were considered to be diffuse sources within the catchment. This was done because the treated water is not directly discharged into the wetland but mixes with runoff water or seeps into the ground. The actual amount of treated water in the WWTP and actual inflowing P concentrations were unknown. Therefore, in this study estimations about the treatment efficiency were made with referring to standard values for wastewater production combined with assumed PE.

# 5. Conclusion

The study investigated on the wetland Vilhelmsberg and an upstream located natural waterbody Maren. Results showed that the wetland Vilhelmsberg functions as a P trap by decreasing the P load from agriculture to the Baltic Sea. However, the accumulation of particles and associated P was low compared to other wetlands in Sweden. This was probably due to the wetland's non-optimal position downstream of Maren, which received most particles of the upstream catchment in combination with the wetland's open outlet which allowed additional water inflow from the Baltic Sea.

The particle size distribution showed that silt was the dominating particle size in the sediment in both waterbodies. Furthermore, at the inlet of the wetland Vilhelmsberg the clay fraction was eight times higher than at the outlet and three times higher than at the curves, probably indicating resuspension caused by the Baltic Sea and efficient particle settling at the wetland's inlet. The dominating P fraction in the sediment of the wetland Vilhelmsberg and Maren was Ca-P which positively correlated with the TP concentration. The mobile P concentration (PW-P and Fe-P) and the maximum potential P release rate of both waterbodies were low, indicating a small risk of internal loading.

Due to a low sediment accumulation rate and a small risk of internal loading there is no current need for sediment removal of the wetland Vilhelmsberg. Though, the sediment will be a good soil amendment in the catchment regarding P-AL. Furthermore, it will be allowed to return it to agricultural fields regarding the metal contents. Contrary, Maren's sediment will not be allowed to be transferred for agricultural purposes because it exceeded the limit for nickel content.

Maren had a huge impact on retaining P because 98 % of the theoretical estimated P load that the wetland Vilhelmsberg would have received from the upstream catchment, probably passed through Maren. Based on a theoretical model, Maren retained 43 % of the estimated P load that would have reached the wetland Vilhelmsberg. The largest proportion of the estimated P load for both waterbodies came from agriculture, followed by forests. However, the loads from the Baltic Sea were not estimated in this study. The impact of the treated wastewater from the WWTP on the P load is negligible accounting for less than 1 %.

# 6. Recommendations and further research

It is recommended to establish a barrier at the outlet of the wetland to limit the influence of the Baltic Sea. Additionally, design considerations for improving particle accumulation should be made, for instance constructing a deeper part at the inlet or middle, followed by a shallow zone with emergent vegetation.

Extended research of the wetland Vilhelmsberg could evaluate the influence of the Baltic Sea more precisely. P bindings can potentially be affected due to the higher contents of sulphate in Sea water and a research question could for instance be “How does the Sea water influence the risk of internal P loading?”

Before returning the sediment back to the fields, it is recommended to analyse the salt content and compare it with values that negatively affect the soil status and plant growth. It is also recommended to investigate on how save the sediment recycling of the wetland’s inlet will be regarding the high barium contents for human and animal health.

The overall impact of wetland Vilhelmsberg on the ecosystem was not assessed in this thesis. Though other positive aspects like biodiversity enrichment or N treatment and negative aspects like emission of greenhouse gases could be investigated.

# References


Abreu, C.A., Cantoni, M., Coscione, A.R. & Paz-Ferreiro, J. (2012). Organic Matter and Barium Absorption by Plant Species Grown in an Area Polluted with Scrap Metal Residue. *Applied and Environmental Soil Science,*. https://doi.org/10.1155/2012/476821

Andersen, J.H., Carstensen, J., Conley, D.J., Dromph, K., Fleming-Lehtinen, V., Gustafsson, B.G., Josefson, A.B., Norkko, A., Villnäs, A. & Murray, C. (2017). Long-term temporal and spatial trends in eutrophication status of the Baltic Sea. *Biological Reviews*, 92, 135–149. https://doi.org/10.1111/brv.12221

André, A., Sundin, A.M., Linderholm, L., Borbas, I. & Svinhufvud, K. (2016). *Wastewater treatment in Sweden 2016*. Stockholm: Swedish EPA.

Boesch, D., Hecky, R., Chair, C.O., Schindler, D. & Seitzinger, S. (2006). *Eutrophication of Swedish Seas*. (5509). Stockholm: Swedish Environmental Protection Agency. http://www.naturvardsverket.se/Documents/publikationer/620-5509-7.pdf [2021-02-25]

Braskerud, B.C. (2001). The Influence of Vegetation on Sedimentation and Resuspension of Soil Particles in Small Constructed Wetlands. *Journal of Environmental Quality*, 30, 1447–1457. https://doi.org/10.2134/jeq2001.3041447x

Braskerud, B.C. (2003). Clay particle retention in small constructed wetlands. *Water Research*, 37 (16), 3793–3802. https://doi.org/10.1016/S0043-1354(02)00484-0

Braskerud, B.C., Lundekvam, H. & Krogstad, T. (2000). The Impact of Hydraulic Load and Aggregation on Sedimentation of Soil Particles in Small Constructed Wetlands. *Journal of Environmental Quality*, 29, 2013–2020. https://doi.org/10.2134/jeq2000.00472425002900060039x

Braskerud, B.C., Tonderski, K.S., Wedding, B., Bakke, R., Blankenberg, A.-G.B., Ulén, B. & Koskiaho, J. (2005). Can Constructed Wetlands Reduce the Diffuse Phosphorus Loads to Eutrophic Water in Cold Temperate Regions? *Journal of Environmental Quality*, 34, 2145–2155. https://doi.org/10.2134/jeq2004.0466

Bryhn, A.C. & Håkanson, L. (2010). Land uplift effects on the phosphorus cycle of the Baltic Sea. *Environmental Earth Sciences*, 62 (8). https://doi.org/10.1007/s12665-010-0656-6

Chen, H. (2011). Surface-Flow Constructed Treatment Wetlands for Pollutant Removal: Applications and Perspectives. *Wetlands*, 31, 805–814. https://doi.org/10.1007/s13157-011-0186-3

Chislock, M.F., Doster, E., Zitomer, R.A. & Wilson, A.E. (2013). Eutrophication: Causes, Consequences, and Controls in Aquatic Ecosystems. *Nature Education Knowledge*, 4 (4)

Conley, D.J., Paerl, H.W., Howarth, R.W., Boesch, D.F., Seitzinger, S.P., Havens, K.E., Lancelot, C. & Likens, G.E. (2009). Controlling Eutrophication: Nitrogen and Phosphorus. *Science*, 323, 1014–1015. https://doi.org/10.1126/science.1167755

Corbee, G. (2021). *Phosphorus accumulation in constructed wetlands: A study of 10 wetlands constructed on agricultural clay soils in Södermanland*. Uppsala University and Swedish University of Agricultural Sciences.

Dagerskog, L. & Olsson, O. (2020). *Swedish sludge management at the cross-roads*. Stockholm: Stockholm Environmental Institute.
Di Luca, G.A., Maine, M.A., Mufarrege, M.M., Hadad, H.R., Pedro, M.C., Sánchez, G.C. & Caffaratti, S.E. (2017). Phosphorus distribution pattern in sediments of natural and constructed wetlands. *Ecological Engineering*, 108, 227–233. https://doi.org/10.1016/j.ecoleng.2017.08.038
Djodjic, F., Elmquist, H. & Collentine, D. (2018). Targeting critical source areas for phosphorus losses: Evaluation with soil testing, farmers' assessment and modelling. *Ambio*, 47, 45–56. https://doi.org/10.1007/s13280-017-0935-5
Dotro, G., Langergraber, G., Molle, P., Nivala, J., Puigagut, J., Stein, O. & von Sperling, M. (2017). Treatment Wetlands. *IWA Publishing*,. https://doi.org/10.2166/9781780408774
Ellis, J.B., Shutes, R.B.E. & Revitt, D.M. (2003). *Guidance Manual for Constructed Wetlands*. (P2-159). Bristol: Environment Agency.
Eriksson, E. (2013). *Organic substances in sewage sludge intended for agricultural land – what do we know and where are the knowledge gaps?* Denmark. http://www.naturvardsverket.se/Documents/publikationer/620-5214-4.pdf [2021-05-19]
ESRI (2021). *How To: Create centerlines from polygon features in ArcMap*. https://support.esri.com/en/technical-article/000012414 [2021-04-07]
Eurofins (2012). *Soil survey (in Swedish: Analysrapport)*. Lidköping: Eurofins Food & Agro Sweden AB.
Fenton, M., Albers, C. & Ketterings, Q. (2008). *Soil Organic Matter*. (Agricultural Fact Sheet Series, Factsheet 41). Cornell University.
Fonder, N. & Headley, T. (2010). Systematic Classification, Nomenclature and Reporting for Constructed Treatment Wetlands. In: Vymazal, J. (ed.) *Water and Nutrient Management in Natural and Constructed Wetlands*. Praha: Springer, 191–219. https://doi.org/10.1007/978-90-481-9585-5_15
Forslund, J. & Barregård, L. (n.d.). Remediation of Sites Contaminated by Arsenic. *National Institute of Economic Research*, 30
Gad, S.C. (2014). Barium. *Encyclopedia of Toxicology*. Elsevier, 368–370. https://doi.org/10.1016/B978-0-12-386454-3.00819-8
Geranmayeh, P., Johannesson, K.M., Ulén, B. & Tonderski, K.S. (2018). Particle deposition, resuspension and phosphorus accumulation in small constructed wetlands. *Ambio*, 47, 134–145. https://doi.org/10.1007/s13280-017-0992-9
Gunes, K., Tuncsiper, B., Ayaz, S. & Drizo, A. (2012). The ability of free water surface constructed wetland system to treat high strength domestic wastewater: A case study for the Mediterranean. *Ecological Engineering*, 44, 278–284. https://doi.org/10.1016/j.ecoleng.2012.04.008
Gustafsson, E., Savchuk, O.P., Gustafsson, B.G. & Müller-Karulis, B. (2017). Key processes in the coupled carbon, nitrogen, and phosphorus cycling of the Baltic Sea. *Biogeochemistry*, 134 (3), 301–317. https://doi.org/10.1007/s10533-017-0361-6
Hallberg, A. & Reginiussen, H. (2019). *Mapping of innovation-critical metals and minerals*. (20). Uppsala: Geological Survey of Sweden.
Hansson, K., Ejhed, H., Widén-Nilsson, E., Johnsson, H., Brunell, J.T., Gustavsson, H., Hytteborn, J. & Åkerblom, S. (2019). *The nutrient load on the Baltic Sea and the North Sea 2017 - Sweden's data for HELCOM's seventh Pollution Load Compilation (In Swedish: Näringsbelastningen på Östersjön och Västerhavet 2017 - Sveriges underlag till HELCOM:s sjunde Pollution Load Compilation)*. (20). Swedish Agency for Marine and Water Management.
HELCOM (2007). *HELCOM Baltic Sea Action Plan*. Krakow: HELCOM Ministerial Meeting.
HELCOM (2021). *BSAP update 2021 – HELCOM*. https://helcom.fi/baltic-sea-action-plan/bsap-update-2021/ [2021-03-09]

HELCOM (2021). *Helcom Map And Data Service*. http://maps.helcom.fi/website/mapservice/ [2021-04-10]
HELCOM (2018a). HELCOM Thematic assessment of eutrophication 2011-2016. *Baltic Sea Environment Proceedings*, 156
HELCOM (2018b). Sources and pathways of nutrients to the Baltic Sea. *Baltic Sea Environmental Proceedings*, 153
Johannesson, K., Linköpings universitet & Institutionen för fysik, kemi och biologi (2011). *Analysis of phosphorus retention variations in constructed wetlands receiving variable loads from arable land*. Department of Physics, Chemistry and Biology, Linköpings universitet.
Johannesson, K.M., Kynkäänniemi, P., Ulén, B., Weisner, S.E.B. & Tonderski, K.S. (2015). Phosphorus and particle retention in constructed wetlands - A catchment comparison. *Ecological Engineering*, 80, 20–31. https://doi.org/10.1016/j.ecoleng.2014.08.014
Johnsson, H., Mårtensson, K., Lindsjö, A., Persson, K., Rangel, Y.A. & Blombäck, K. (2019). *Leakage of nutrients from Swedish arable land (in Swedish: Läckage av näringsämnen från svensk åkermark)*. (5). SMED.
Jordan, T.E., Whigham, D.F., Hofmockel, K.H. & Pittek, M.A. (2003). Nutrient and Sediment Removal by a Restored Wetland Receiving Agricultural Runoff. *Journal of Environment Quality*, 32 (4), 1534. https://doi.org/10.2134/jeq2003.1534
Kadlec, R.H. (2005). Phosphorus Removal in Emergent Free Surface Wetlands. *Journal of Environmental Science and Health, Part A*, 40 (6–7), 1293–1306. https://doi.org/10.1081/ESE-200055832
Kadlec, R.H. & Reddy, K.R. (2001). Temperature Effects in Treatment Wetlands. *Water Environment Research*, 73 (5), 557. https://doi.org/10.2175/106143001X139614
Karlsson, M. (2005). *Nutritional load on a landscaped wetland from fields, forests and individual sewers (in Swedish: Näringsbelastning på en anlagd våtmark från åker, skog och enskilda avlopp)*. Swedish University of Agricultural Sciences.
Koskiaho, J., Ekholm, P., Räty, M., Riihimäki, J. & Puustinen, M. (2003). Retaining agricultural nutrients in constructed wetlands - experiences under boreal conditions. *Ecological Engineering*, 20, 89–103. https://doi.org/10.1016/S0925-8574(03)00006-5
Kynkäänniemi, P. (2014). *Small Wetlands Designed for Phosphorus Retention in Swedish Agricultural Areas*. Swedish University of Agricultural Sciences.
Kynkäänniemi, P., Ulén, B., Torstensson, G. & Tonderski, K.S. (2013). Phosphorus Retention in a Newly Constructed Wetland Receiving Agricultural Tile Drainage Water. *Journal of Environmental Quality*, 42, 596–605. https://doi.org/10.2134/jeq2012.0266
Land, M., Granéli, W., Grimvall, A., Hoffmann, C.C., Mitsch, W.J., Tonderski, K.S. & Verhoeven, J.T.A. (2016). How effective are created or restored freshwater wetlands for nitrogen and phosphorus removal? A systematic review. *Environmental Evidence*, 5 (9), 1–26. https://doi.org/10.1186/s13750-016-0060-0
Lannergård, E.E., Agstam-Norlin, O., Huser, B.J., Sandström, S., Rakovic, J. & Futter, M.N. (2020). New Insights Into Legacy Phosphorus From Fractionation of Streambed Sediment. *Journal of Geophysical Research: Biogeosciences*, 125. https://doi.org/10.1029/2020JG005763
Liguori, B.T.P., De Almeida, M.G. & De Rezende, C.E. (2016). Barium and its importance as an indicator of (paleo)productivity. *Anais da Academia Brasileira de Ciências*, 88 (4), 2093–2103. https://doi.org/10.1590/0001-3765201620140592
Michélsen, S.K. (2021). *Risk of internal loading in the sediments of constructed wetlands (in Swedish: Internbelastningsrisk från våtmarkssediment)*. Swedish University of Agricultural Sciences.

Microsoft (2021). *Use the Analysis ToolPak to perform complex data analysis*. https://support.microsoft.com/en-us/office/use-the-analysis-toolpak-to-perform-complex-data-analysis-6c67ccf0-f4a9-487c-8dec-bdb5a2cefab6 [2021-06-02]
Nogueira, T.A.R., deMelo, W.J., Fonseca, I.M., Marques, M.O. & He, Z. (2010). Barium uptake by maize plants as affected by sewage sludge in a long-term field study. *Journal of Hazardous Materials*, 181, 1148–1157. https://doi.org/10.1016/j.jhazmat.2010.05.138
Olshammar, M., Ek, M., Rosenquist, L., Ejhed, H., Sidvall, A. & Svanström, S. (2015). *Update of the state of knowledge and statistics for small sewage treatment plants (in Swedish: Uppdatering av kunskapsläget och statistik för små avloppsanläggningar)*. (166). SMHI.
Pajunen, H. (ed.) (2000). *Carbon in Finnish lake sediments*. Espoo: Geological Survey of Finland. (Special paper; 29)
Palmer-Felgate, E.J., Mortimer, R.J.G., Krom, M.D., Jarvie, H.P., Williams, R.J., Spraggs, R.E. & Stratford, C.J. (2011). Internal loading of phosphorus in a sedimentation pond of a treatment wetland: Effect of a phytoplankton crash. *Science of the Total Environment*, 409, 2222–2232. https://doi.org/10.1016/j.scitotenv.2011.02.034
Pant, H.K. (2020). Estimation of Internal Loading of Phosphorus in Freshwater Wetlands. *Current Pollution Reports*, 6, 28–35. https://doi.org/10.1007/s40726-020-00136-6
Piikki, K. & Söderström, M. (2019). Digital soil mapping of arable land in Sweden – Validation of performance at multiple scales. *Geoderma*, 352, 342–350. https://doi.org/10.1016/j.geoderma.2017.10.049
Pilgrim, K.M., Huser, B.J. & Brezonik, P.L. (2007). A method for comparative evaluation of whole-lake and inflow alum treatment. *Water Research*, 41 (6), 1215–1224. https://doi.org/10.1016/j.watres.2006.12.025
Rathore, S.S., Chandravanshi, P., Chandravanshi, A. & Jaiswal, K. (2016). Eutrophication: Impacts of Excess Nutrient Inputs on Aquatic Ecosystem. *IOSR Journal of Agriculture and Veterinary Science*, 09 (10), 89–96. https://doi.org/10.9790/2380-0910018996
Rydin, E., Malmaeus, J.M., Karlsson, O.M. & Jonsson, P. (2011). Phosphorus release from coastal Baltic Sea sediments as estimated from sediment profiles. *Estuarine, Coastal and Shelf Science*, 92, 111–117. https://doi.org/10.1016/j.ecss.2010.12.020
SGU (2021). *Marine geology map viewer (in Swedish: Maringeologi-kartvisare)*. https://www.sgu.se/produkter/kartor/kartvisaren/maringeologikartvisare/ [2021-05-12]
SGU (2021). *Ore and minerals (in Swedish: Malm och mineral)*. https://www.sgu.se/produkter/kartor/kartvisaren/bergkartvisare/malm-och-mineral/ [2021-05-12]
Simpson, G. & Wu, Y. (2014). Accuracy and Effort of Interpolation and Sampling: Can GIS Help Lower Field Costs? *ISPRS International Journal of Geo-Information*, 3 (4), 1317–1333. https://doi.org/10.3390/ijgi3041317
SMED (2021). *Swedish Environmental Emission Data TBV, PLC7*. https://tbv.smhi.se/tbv/overview/ [2021-03-03]
SMHI (2021). *Download oceanographic observations (in Swedish: Ladda ner oceanografiska observationer)*. https://www.smhi.se/data/oceanografi/ladda-ner-oceanografiska-observationer/#param=sealevelrw,stations=all,stationid=2507 [2021-04-23]
SMHI (2021). *HYPE: Our Hydrological Model*. https://www.smhi.se/en/research/research-departments/hydrology/hype-our-hydrological-model-1.7994 [2021-05-17]
SMHI (2021). *Water web (in Swedish: Vattenwebb)*. https://www.smhi.se/data/hydrologi/vattenwebb [2021-04-22]
de Smith, M.J., Goodchild, M.F. & Longley, P.A. (2018). *Geospatial Analysis: A Comprehensive Guide to Principles, Techniques and Software Tools*. 6.

ed. https://www.spatialanalysisonline.com/HTML/index.html [2021-03-17]
Søndergaard, M., Jensen, J.P. & Jeppesen, E. (2003). Role of sediment and internal loading of phosphorus in shallow lakes. *Hydrobiologia*, 506–509 (1–3), 135–145. https://doi.org/10.1023/B:HYDR.0000008611.12704.dd
Stagg, C.L., Baustian, M.M., Perry, C.L., Carruthers, T.J.B. & Hall, C.T. (2018). Direct and indirect controls on organic matter decomposition in four coastal wetland communities along a landscape salinity gradient. *Journal of Ecology*, 106, 655–670. https://doi.org/10.1111/1365-2745.12901
Stephan, U., Hengl, M. & Schmid, B.H. (2005). Sediment Retention in Constructed Wetland Ponds - A Laboratory Study. *Journal of Environmental Science and Health*, 40 (6–7), 1415–1430. https://doi.org/10.1081/ESE-200055879
Stramska, M., Kowalewska-Kalkowska, H. & Świrgoń, M. (2013). Seasonal variability in the Baltic Sea level. *Oceanologia*, 55 (4), 787–807. https://doi.org/10.5697/oc.55-4.787
Strand, J.A. & Weisner, S.E.B. (2011). Combating eutrophication of the sea and enhancing biodiversity of the agricultural landscape: experiences from wetland creation in Sweden. *Proceedings of 15th International Conference of the IWA Diffuse Pollution Specialist Group on: Diffuse Pollution and Eutrophication*, New Zealand, 2011. New Zealand
Sveistrup, T.E., Marcelino, V. & Braskerud, B.C. (2008). Aggregates explain the high clay retention of small constructed wetlands: a micromorphological study. *Boreal Environment Research*, 13, 275–284
SwAM (2016). *The Swedish Agency for Marine and Water Management's general advice on small sewage devices for domestic wastewater (in Swedish: Havs- och vattenmyndighetens allmänna råd om små avloppsanordningar för hushållsspillvatten). HVMFS 2016:17*
Swedish EPA (1994). *Proclamation with regulations on protection of the environment, especially the soil, when sewage sludge is used in agriculture (in Swedish: Kungörelse med föreskrifter om skydd för miljön, särskilt marken, när avloppsslam används i jordbruket). SNFS 1994:2 (with two amendment regulations SNFS 1998:4 , SNFS 2001:5)*
Swedish EPA (2006). *The Swedish Environmental Protection Agency's general advice on small sewage devices for domestic wastewater (in Swedish: Naturvårdsverkets allmänna råd om små avloppsanordningar för hushållsspillvatten). NFS 2006:7*
Swedish EPA (2008). *Appendices to the manual 'Small sewer systems' (in Swedish: Bilagor till handboken 'Små avloppsanläggningar')*. Stockholm: Naturvårdsverket.
Swedish EPA (2009). *Guide values for contaminated soil (in Swedish: Riktvärden för förorenad mark)*. (5976). Stockholm: Swedish Environmental Protection Agency.
Ulén, B. (2004). Size and settling velocities of phosphorus-containing particles in water from agricultural drains. *Water, Air and Soil Pollution*, 157, 331–343. https://doi.org/10.1023/B:WATE.0000038906.18517.e2
Ulén, B., Bechmann, M., Fölster, J., Jarvie, H.P. & Tunney, H. (2007). Agriculture as a phosphorus source for eutrophication in the north-west European countries, Norway, Sweden, United Kingdom and Ireland: a review. *Soil Use and Management*, 23, 5–15. https://doi.org/10.1111/j.1475-2743.2007.00115.x
United States Department of Agriculture (2011). Carbon to Nitrogen Ratios in Cropping Systems
Verbyla, M.E. (2016). *Ponds, Lagoons, and Wetlands for Wastewater Management*. New York: Momentum Press.

Weisner, S.E.B., Johannesson, K., Thiere, G., Svengren, H., Ehde, P.M. & Tonderski, K.S. (2016). National Large-Scale Wetland Creation in Agricultural Areas - Potential versus Realized Effects on Nutrient Transports. *Water*, 8 (544). https://doi.org/10.3390/w8110544

Withers, P.J.A., van Dijk, K.C., Neset, T.-S.S., Nesme, T., Oenema, O., Rubæk, G.H., Schoumans, O.F., Smit, B. & Pellerin, S. (2015). Stewardship to tackle global phosphorus inefficiency: The case of Europe. *Ambio*, 44 (2), 193–206. https://doi.org/10.1007/s13280-014-0614-8

Wörman, A. & Kronnäs, V. (2005). Effect of pond shape and vegetation heterogeneity on flow and treatment performance of constructed wetlands. *Journal of Hydrology*, 301, 123–138. https://doi.org/10.1016/j.jhydrol.2004.06.038

# Appendix

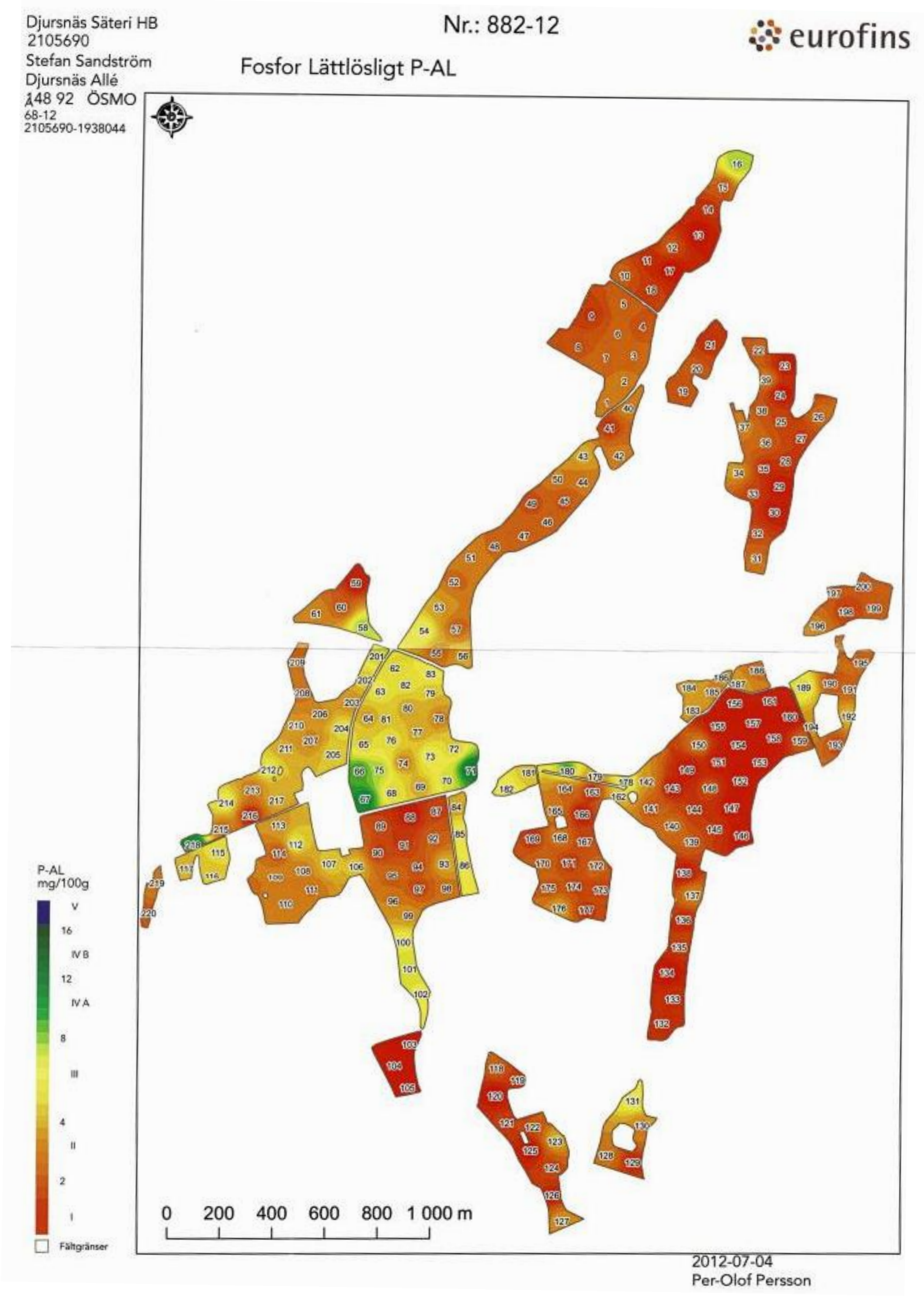


*Figure 15. P-AL measurement results of a soil survey conducted by Eurofins in 2012 (sampling numbers 1 to 117 and 132 to 218 are located within the catchment of the wetland Vilhelmsberg)*