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Master Thesis in Geoinformatics for Urbanized Society (30 ECTS)

Modelling the Nitrous Oxide Emissions from Vända Free Water Surface Constructed Wetland, Tartu, Estonia

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Abstract (Annotation)

Title: Modelling the Nitrous Oxide Emissions from Vända Free Water Surface Constructed Wetland, Tartu, Estonia.

Nitrous oxide (N_2O) is a potent greenhouse gas and the primary driver of stratospheric ozone depletion. Since constructed wetlands (CWs) have been identified as a possible source of N_2O, this study aimed to model the emission at an in-stream free surface flow (FSW) Vända CW in Southern Estonia from May 2018 till November 2020. The catchment area of Vända CW is 2.2 km^2, with approximately 62% of it under intensive agricultural management. The CW consists of two shallow-water wetlands planted with cattail (*Typha latifolia*). Nitrous oxide samples were collected from twelve sampling points in the wetland together with other parameters such as water level depth, conductivity, and oxygen concentration. From the inflow and outflow of the wetland, the total nitrogen, nitrate-nitrogen, and total organic carbon concentration were determined as well. Analysis showed that the N_2O emission from the wetland is influenced by various factors, including vegetation, flow rate, water depth, temperature, and the availability of carbon and nitrogen compounds. Water depth showed a negative correlation with N_2O. The highest emission of about 2600 $\mu g\ m^{-2}\ hr^{-1}$ was observed at a water level of less than 15cm, which corresponds to a region with high vegetation, higher temperature, and at some points, lower flow rate. Modeling predicted that Fluxes during the growing season would be reduced by 86% if the water level could be held to no less than 15 cm above the ground level. However, increasing it to 70 cm would reduce them by over 99%.

Keywords: global warming, constructed wetlands, nitrous oxide, denitrification, nitrification, nitrogen cycle.
CERCS code: T270 Environmental technology, pollution control

Abstrakt (Annotatsioon)

Pealkiri: Lämmastikdioksiidi emissioonide modelleerimine Vända avaveelise tehismärgala näitel

Dilämmastikoksiid (N_2O) ehk naerugaas on tugev kasvuhoonegaas ja üks peamisi tegureid osoonikihi kahanemises stratosfääris. Kuna tehismärgalasid peetakse potentsiaalseteks N_2O allikateks, oli antud uurimuse eesmärgiks modelleerida N_2O heitkoguseid voolusängile rajatud avaveelisest Vända tehismärgalast, mis asub Uhti külas, Tartumaal, ajaperioodil mai 2018–november 2020. Vända tehismärgala valgala pindala on ca 2,2 km^2, millest umbes 62% on intensiivse põllumajanduse all, mis tingib ka taimetoitainete rohkuse kraavivees. Tehismärgala koosneb kahest madalaveelisest märgalast, mis on taimestatud laialehise hundinuiaga (*Typha latifolia*). N_2O proovid koguti kaheteistkümnest märgalal asuvast proovivõtupunktist koos teiste parameetritega, nagu veetase, temperatuur, elektrijuhtivus ja hapniku kontsentratsioon. Märgalade sisse- ja väljavoolu põhjal määrati ka üldlämmastiku, nitraatse lämmastiku ja üldorgaanilise süsiniku kontsentratsioonid. N_2O proovide analüüs näitas, et märgalalt pärinevat emissiooni mõjutavad erinevad tegurid, sealhulgas taimestik, voolukiirus, vee sügavus, temperatuur ning süsiniku- ja lämmastikuühendite kättesaadavus. Vee sügavuse ja N_2O emissiooni vahel oli negatiivne korrelatsioon. Suurimat emissiooni, ligikaudu 2600 μg m^{-2} hr^{-1}, täheldati alla 15 cm veetasemega piirkondades, kus on kõikuv veetase, kõrgem temperatuur ja mõnes kohas ka madalam voolukiirus. Modelleerimise järgi prognoositi, et vegetatsiooniperioodil vähenevad N_2O vood 86% võrra, kui veesügavus on vähemalt 15 cm. Veetaseme kergitamine 70 cm-ni maapinnast vähendaks N_2O vooge üle 99%.

Võtmesõnad: kliima soojenemine, tehismärgalad, lämmastikdioksiid, denitrifikatsioon, nitrifikatsioon, lämmastikuringe

CERCS klassifikaator: T270 keskkonnatehnoloogia, reostuskontroll

Content

Introduction

Water pollution is a problem that affects not only the environment but also the economy and human health (Uuemaa et al., 2018). Recent population growth has resulted in a rise in food demand, affecting agriculture and other land management (Kill et al., 2018). This has led to changes in land use, thereby intensifying the use of fertilizers, which in turn serve as an increasing measure of the impact of diffuse pollution on water quality (Pärn et al., 2018). Kasak et al. (2018) noted that diffuse wetland pollution from agriculture had been a major cause of eutrophication, especially from intensely managed agricultural land; hence there is a need to reduce the diffuse load to surface water. Constructed wetlands (hereinafter referred to as CWs) are one of the most essential tools to reduce pollution, including that caused by nutrient losses from agricultural fields by capturing and eliminating sediments and nutrients in runoff before they reach surface waters (Kadlec, 2012). Previous studies (Koskiaho et al., 2005; Vymazal, 2010; Mustafa, 2013; Kasak et al., 2018; Khazaleh and Gopalan, 2018; Kumar and Choudhary, 2018; Charvan and Mutnuri, 2020) has also emphasized the use of CWs for the improvement of water quality using natural water treatment processes.

The treatment processes for agricultural runoff aim to reduce the content of inflow from agricultural nitrogen-rich organic soils. These contents include suspended solids, organics, NH_3^-, NO_2^-, NO_3^-, N_{org}, metals, pathogens, among others (Soosaar et al., 2009). The removal or retention of nitrogen, which is the primary cause of N_2O emission from CWs, entails various processes. Nitrification, denitrification, plant and microbial uptake, nitrogen fixation, mineralization (ammonification), nitrate reduction to ammonium (nitrate ammonification), fragmentation, sorption, anaerobic ammonia oxidation (Anammox), desorption, burial, and leaching are examples of these processes (Vymazal, 2007). Just a few processes, however, are effective in extracting total nitrogen (Vymazal, 2007). Organic materials and nitrogen are mainly separated in these processes by volatilization into various gaseous substances such as N_2, N_2O, NO, and NH_3. These gases are released from wetland plants culms or through diffusion through the water from waterlogged soil (Soosaar et al., 2009). Other factors that serve as a driving mechanism for the level of these emissions include but are not limited to water table depth, oxidation-reduction potential (ORP), hydraulic retention time, pH, and water temperature, among other factors (Wu et al., 2009). Pärn et al. (2018), for example, noted that increase in agricultural soil temperature correlated positively with the emission of N_2O while water table depth shows a negative correlation.

Besides the use of CWs as a vital tool for mitigating diffuse water pollution from farms, they also serve as potential habitats for conserving wildlife as such could reduce the negative effect of declining biodiversity by human activities, especially when applied as wildlife refuges due to the severe destruction of natural habitats (Hale et al., 2019). This has contributed to an increase in CWs worldwide, which is estimated to have risen by 5-50% since the 1970s and earlier, although accurate data cannot be accounted for since about 35% of wetlands data has been lost (Ramsar Convention on Wetlands, 2018; Rannap et al., 2020).

An exemplary type of CWs is the free water surface (FWS) CWs which contain areas of open water, emergent plant, and floating vegetation, either by design or as an unavoidable consequence of the design configuration (Kadlec and Wallace, 2008). There are two ways to create surface flow CWs based on the wetland position in relation to the stream: directly into the stream (in-stream) or adjacent to the stream (off-stream). Just a portion of the water is routed into the CWs for purification in off-stream wetlands, which are located outside of a stream or river channel (Kadlec and Wallace, 2008), while in-stream CWs are located directly in the flow direction, where all the water from the river/ditch flows through the wetland, where the flow rate is reduced, and nutrient removal processes can occur for more extended periods (Darwiche-Criado et al., 2017).

Koskiaho et al. (2003) noted that most of the FWS CWs utilized for the treatment of diffuse pollution to be off stream. This is due to the inability of in-stream CWs to completely treat storm or floodwater, as well as the lack of land availability in intensive agriculture areas to reach a sufficient wetland catchment ratio (Kasak et al., 2018). This does not limit the fact that in-stream wetlands are often much larger than off-stream and, as such, provide other environmental benefits such as increasing biodiversity (Hsu et al., 2011). Also, if properly designed and the wetland catchment ratio is achieved, in-stream wetlands can effectively treat water (Braskerund, 2002) because nutrient mitigation occurs across the stream, unlike off-stream wetlands that treat only part of the water of the water (Arheimer and Pers, 2017). In addition, in-stream wetlands have a higher permanent vegetation cover (50–90%) than off-stream wetlands (10–20%) due to normalized permanent flooding, which is essential for water treatment (Darwiche-Criado et al., 2017). Thus, it is expected that instream wetland would emit more N_2O gases compare to off-stream since nutrient mitigation occurs across the stream and the presence of more vegetation that aids and promote pollutant mitigation and removal through uptake, degradation, and sorption through the biofilm they help for microbial consortia activity, as well as oxygen supply through the root system (Stefanakis et al., 2014).

Søvik et al. (2007) noted that various studies conducted on the greenhouse gas (GHG) emission from CWs had indicated high N_2O and methane (CH_4) emissions. Generally, it has been agreed that the observed increase in the level of GHGs such as carbon dioxide (CO_2), N_2O, and CH_4 in the atmosphere has led to a warming of the Earth's surface (Søvik et al., 2007). In 2001, the Intergovernmental Panel on Climate Change (IPCC) noted the N_2O concentration in the atmosphere to be increasing at a rate of 0.3% per year and has an atmospheric lifetime of 120 years with a global warming potential of 296 in relation to CO_2 (over a 100-year time horizon) and was predicted to be the source of about 5% global warming. This prediction has been recently estimated to have increased between 2007 and 2016 and contributes to about 23% of global warming (IPCC, 2019).

Ravishankara et al. (2009) noted the importance of N_2O in the depletion of the atmosphere with a debate on the N_2O importance for ozone depletion stating it to be the most crucial anthropogenic ozone depletion substance. This was subsequently supported by Portmann et al. (2012), which discussed the importance of N_2O to ozone depletion through the 21st century. Thus, N_2O, among other chemical compounds or elements, has been recognized as an essential substance that affects the stratospheric ozone. Therefore, it is crucial to research further the various medium in which N_2O gas is emitted to the atmosphere. This thesis intends to model the level of N_2O gas emission from an in-stream FWS CWs established for the treatment of diffuse agricultural pollution located at Uhti village, Tartu, Estonia, with an aim in understanding the mechanism for this emission and, in turn, propose a possible model that could mimic the gas emission level. Hence, the research sought to answer the following questions.

1. What is the level of N_2O gas emission from the study location and how it changes in different wetland section?

2. What are the main regulating mechanisms for N_2O emissions?

3. What is the impact of Vända CWs N_2O emission on global warming?

4. Can N_2O emissions be further minimized?

This Master thesis has five chapters. The first provides a theoretical overview in general concerning CWs and GHGs emissions. The second chapter covers the data collected and the methods utilized. The third chapter focuses on the result and analysis done. The fourth chapter is a discussion of the methods and the results of the research. And finally, the fifth chapter will provide a conclusion to this research.

1. Theoretical background

1.1 Constructed wetland and types

CWs are artificial ecosystems that utilize biogeochemical processes observed to be taking place within the natural wetland to treat different kinds of contaminated waters (Zhang et al., 2020). Vymazal (2010), defined CWs as engineered systems that have been designed and constructed to use natural processes utilizing soils, wetland vegetation, and their associated microbial assemblages. Although CWs can be classified based on different design criteria, the three most common criteria that have been used for the categorization of CWs include hydrology which can be either surface flow and subsurface flow, macrophyte growth form, i.e., emergent, submerged, free-floating, and floating leaved plants and the flow path which could be either vertical or horizontal (Vymazal, 2011). There is a possibility to combine different types of CWs, to create hybrid or integrated systems. For example, in the 1990s and 2000s, an enhanced design was done combining vertical and horizontal flow to achieve higher treatment efficiency for the effective removal of ammonia and total nitrogen (N) (Kadlec and Wallace, 2008). Although the various types of wetlands have been classified (Figure 1) and discussed in various research, this thesis only considers the surface flow CWs.

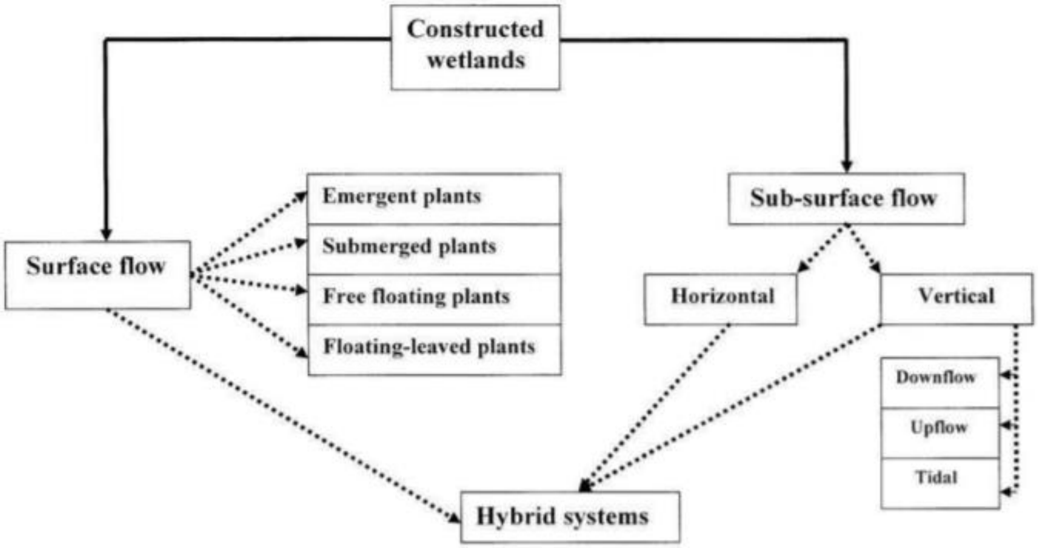

Figure 1: Classification of constructed wetland for different types of water treatment (source: Vymazal 2011).

1.1.2 Constructed wetlands with surface flow

The surface flow CW is also known as free water surface (FWS) CWs, which contains areas of open water, basically submerged with the presence of emergent plants (Kadlec and Wallace, 2008). This type of CW is most efficient in removing organics through the settling and colloidal of particles and microbial degradation (Vymazal, 2010). The water flows particularly in constructed wide channels to increase the water resident time (Hsu et al., 2011). Although FWS CWs with emergent vegetation have been used to treat various types of contaminated waters (Vymazal, 2011), it has basically been applied for tertiary treatment of municipal wastewater alongside stormwater runoff and mine drainage waters (Kadlec and Wallace, 2008). This wetland has been stated by Mander et al. (2018) to be suitable in all climates, including the far north. Vymazal (2008) classify the various types of FWS CWs (Figure 2), ranging from the types with emergent macrophytes (A) to those with floating mats of emergent macrophytes (B), with free-floating macrophytes (C), with submerged macrophytes (D), and with floating leaved macrophytes (E).

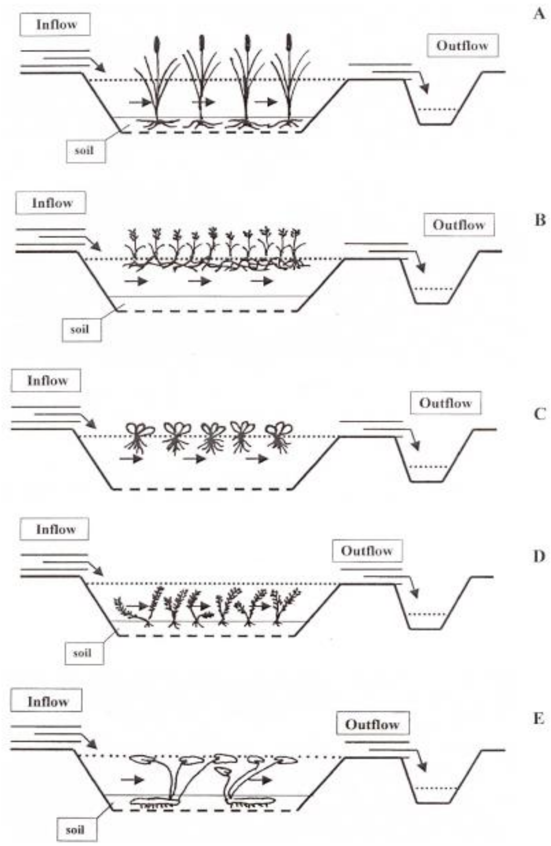

Figure 2: Types of FWS CWs (source: Vymazal, 2008).

1.2 Greenhouse gas emission and global warming

Jain et al. (2015) defined the global warming as the current increase in the average temperature of the Earth's atmosphere and oceans because of the transmission of incoming short-wave radiation from the sun and the absorbance of the outgoing long-wave radiation from the Earth. This process has resulted from the increase of some gases in the atmosphere, such as N_2O, CH_4, and CO_2, which are collectively called GHGs. These gases inhibit the outgoing radiations from the Earth and upset the Earth's heat balance (Ibemere et al., 2015).

From the standpoint of the world agency, Inter-Governmental Panel on Climate Change (IPCC, 2013), an increase of 0.40–0.76°C in the 20th-century annual global mean temperature than as it were at the end of the 19th century was recorded with about two-third of this rise occurring over the last few years (Jain et al., 2015). Researchers (Craig, 2012; Canadell et al., 2010; Acousta-Navarro et al., 2014; D'Andrea et al., 2015; Mohanraj et al., 2019; Moreno-Garcia et al., 2020) have mentioned that most of this rise is caused by the increasing concentration of GHGs by anthropogenic activities among which is attributed to agriculture. Smith et al. (2007) noted Agriculture to be contributing approximately 10%–12% of the total GHG emissions, thereby accounting for about 59 to 60% of the total CH_4 and N_2O anthropogenic emissions. An increasing amount of these GHGs, such as N_2O in the atmosphere, is becoming a very serious thing to the destruction of human livelihood since its global warming potential is 298 times higher (Rees et al., 2013) than that of CO_2. Thus, mitigation of N_2O is needed to combat global warming (Parameswari et al., 2017).

1.2.1 Greenhouse gas emission from constructed wetland

1.2.1.1 Agricultural Runoff and Associated Pollutants

Agricultural runoff is the discharge of agrochemicals, such as fertilizers and pesticides, into surface waters and is the most significant cause of nonpoint source pollution (Ioannidou and Stefanakis, 2020). The leading cause of water quality issues is surface water pollution, which is spread through agricultural non-point source runoff (EPA, 2016). Significant amounts of fertilizers, pesticides, and suspended solids are discharged downstream agricultural catchments or into marine environments adjacent to rural areas because of diffuse agricultural runoff (Ioannidou and Stefanakis, 2020). Agricultural runoff has a significant negative impact on biodiversity, aquatic habitats, and human health through drinking water supply (EPA, 2016), leading to the deterioration of surface and groundwater, with severe environmental and economic consequences (Wu et al., 2013b). As a result, agricultural non-point source pollution research has gotten a lot of attention in the last few decades (Yanhua et al., 2012).

1.2.1.2 N_2O production and driving mechanism

The use of CWs to treat pollutants associated with agricultural diffuse runoff results in GHG emissions (Tao et al., 2015). The use of nitrogenous fertilizers, for example, was identified by Rajkishore et al. (2015) as a potential factor related to the GHGs in wetlands. Because of leaching, volatilization, and immobilization, nitrogenous fertilizers are vulnerable to the loss of N as such can collectively contribute up to 60% (Mohanraj et al., 2019). Adsorption, denitrification and nitrification, sediment retention, and plant absorption are all CW processes related to reducing eutrophication and toxicity from agricultural runoff (Huang et al., 2012). Nitrification, the conversion of ammonium to nitrate, occurs in aerobic areas of FWS CWs (Vymazal, 2013b), while denitrification, or the conversion of NO_3-N to N_2O and N_2 (Mander et al., 2014), occurs in water zones with lower oxygen concentrations (Tournebize et al., 2017).

In general, NO_3^- removal is principally achieved via denitrification (Huang et al., 2012), making this process the major player of N_2O emission in agricultural designed CWs. However, under most conditions, nitrification makes a considerable contribution to the emission in aerobic conditions (Wang et al., 2008). Thus, the product of nitrification works as a substrate for denitrification. As such, controlling the first process will certainly help regulate the second process to an extent (Parameswari, 2017).

Denitrification as a biological process is affected by dissolved oxygen (DO) levels, temperature, vegetation, and pH, among others (Stefanakis et al., 2014). In CWs, plants, for example, provide a range of services. They promote pollutant mitigation and removal through absorption, sorption, and degradation through the biofilms they help for microbial consortia function, as well as through the oxygen supply provided by the root system (Stefanakis et al., 2014). In the presence of vegetation, N removal has also been shown to be positively affected (Stefanakis and Tsihrintzis, 2012). Vymazal showed in 2007 that denitrification rates in CWs increase with temperature. In 2014, Manda et al., demonstrated the impact of the water depth and forms of built wetland. Lower water depths resulted in higher N_2O emissions, according to the findings.

2. Data and Methods

2.1 Data

2.1.1 Site description

The Vända FWS CW (58 17N, 26 43 E) is in the Vända ditch sub-catchment that is part of the Porijõgi river catchment located at the Uhti village in the south-eastern Estonia**.** Approximately 62% of the Vända FWS CW catchment is arable land, 32% is natural land, and around 8% is for other land-use forms (Kasak et al., 2018). The area experiences a temperate climate, with four seasons of almost equal duration. Usually, the growing season ranges from late April to September. During 168 days of rainfall per year, this region has an annual precipitation of 726 mm, whereas the annual average air temperature is 6.3^{O}C, as reported at the nearby Tartu Observatory weather station in Tõravere.

Vända FWS CW comprises two shallow-water wetlands (Figure 3) with a total area of about $4500m^2$. During the construction of the wetland in 2015, several baffles were created to improve hydraulic efficiency and water retention time. The baffles and riparian areas of both wetlands were covered with stones and geotextile (Kasak et al., 2018). Right after the construction works, the first wetland was planted with cattail (*Typha Latifolia*) with four plants per m^2, while the second wetland was left to naturally colonize. In a few years, the first wetland shows a significant development in the growth of approximately 51%, while the second wetland was only 10.5% (Kill et al., 2018).

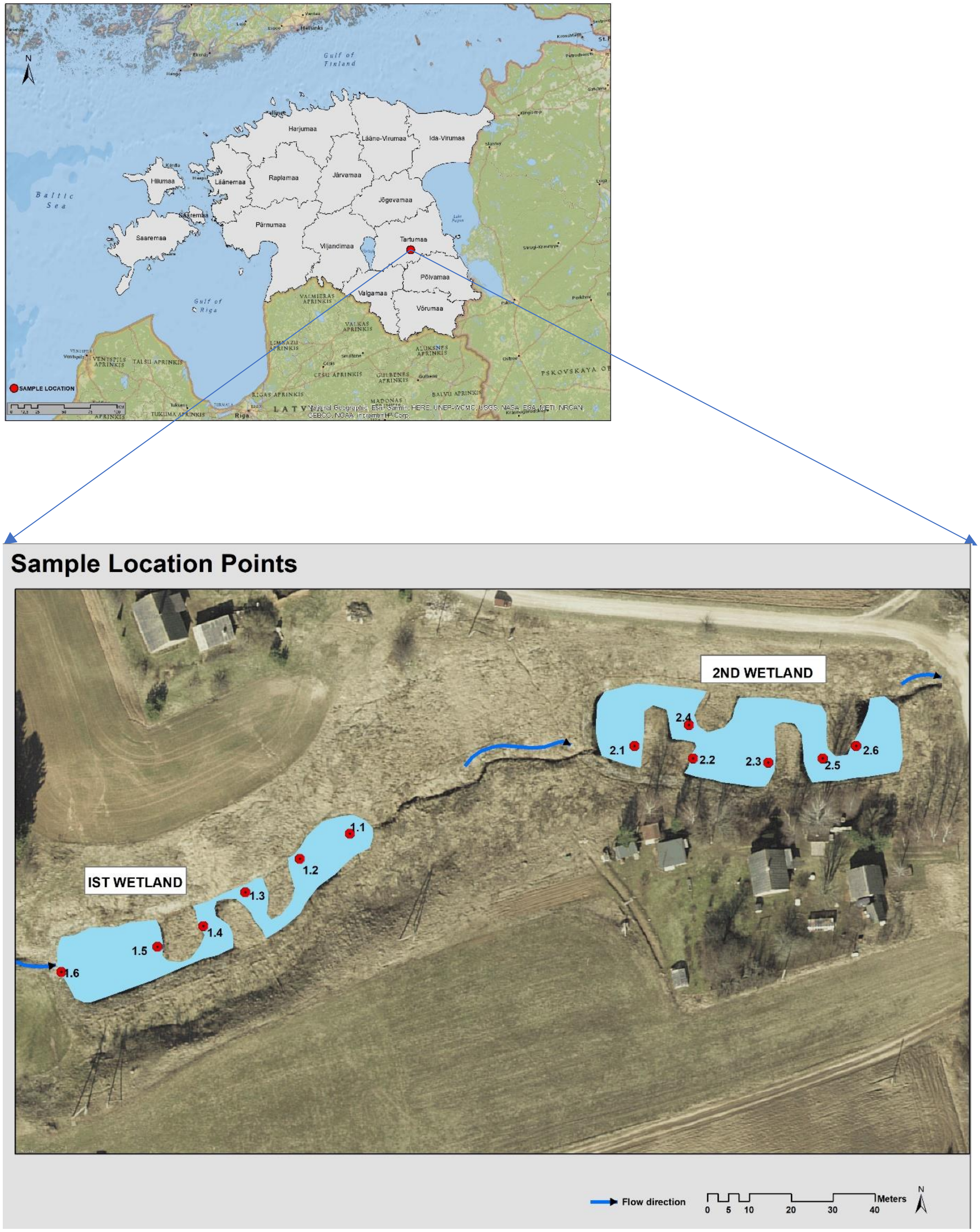

Figure 3: Location map of study area showing sampling points (1.1 – 2.6) for N_2O.
(basemap source: https://geoportaal.maaamet.ee)

2.1.2 Sample collection and preparation process

2.1.2.1 N_2O sample collection

The samples were collected biweekly using closed chambers of height 40 cm, diameter 50 cm, and volume 651cm^3, painted white to avoid heating during the application. The chambers were installed on hollow-foam swimming-pool noodles and anchored for stability (Figure 4). Samples were taken from 12 points on both wetlands (Figure 3 and Appendix Table 1). The Gas samples were taken immediately after the chambers' enclosure (0 moments) and after 20 minutes, 40 minutes, and 60 minutes using pre-evacuated (0.3 mbar) 50-mL gas bottles to determine emission rates. Gas concentration in collected samples was analyzed using the Shimadzu GC-2014 gas chromatography system using an electron capture detector (ECD).

Figure 4: Installed chambers on site.

2.1.2.2 Water sampling

Water samples were collected biweekly during the study period from the inlet (VM 1.6) and outlet (VM 2.6) of both wetlands (Figure 2). Portable device (YSI ProDSS) (YSI Inc., Yellow Springs, OH, USA) was used to measure five parameters on the site, which includes pH, turbidity, temperature, redox potential, and electrical conductivity (EC) while SonTek FlowTracker (YSI Inc., Yellow Springs, OH, USA) handheld acoustic Doppler velocimeter was used to measure the flow rate (Figure 5). To determine other parameters such as total nitrogen (TN), total organic carbon (TOC), and nitrate (NO_3), the water samples collected were first stored in a thermal box before being transported to the laboratory. At the laboratory, the concentration of TN and TOC were analysed with a Vario TOC cube (Elementar GmbH, Germany), while ion chromatography was used to determine concentrations of nitrate-nitrogen (NO_3-N) following standard methods (APHA-AWWA-WEF, 2005).

Figure 5: Author measuring the flow rate on site.

2.1.3 Data structure

The data available for analysis includes:

- Twenty-seven (27) months of N_2O gas emission measurements.
- Twenty-seven (27) months of water table depth from sampling points.
- Twenty-two (22) months of other wetlands parameters such as turbidity, electrical conductivity, dissolved oxygen concentration, flow rates and nitrate, among others.

2.1.4 Data source

Data used for this thesis analysis and visualization were gotten from two sources:

1. Dr. Kuno Kasak, Keit Kill and Isaac Okiti (author): Institute of Ecology and Earth Sciences, Department of Geography, University of Tartu, 50410 Tartu, Estonia.

2. https://geoportaal.maaamet.ee.

2.2 Methods

Two aspects are considered in this study.

1) First was to model the N_2O gas emission from different sections in the wetland.

2) Second was to understand the relationship between the N_2O gas and other wetland parameters such as water table depth, temperature, and oxygen reduction potentials, among others. Figure 6 shows a simplified workflow diagram employed in this study.

2.2.1 Modelling the N_2O gas emission

In most cases, it is typically impossible or costly to visit every location in a study area to measure the gas emission concentration or severity of a phenomenon while attempting to carry out a sample from a study area. Instead, at strategically scattered sampling points, you can determine the phenomenon, and expected values can be assigned to all other locations. The predicted values could be attained using various geostatistical tools, among which is the ArcGIS tool interpolation. Spatial interpolation operates with Waldo Tobler's theory of the first law of geography, which states that while everything is linked to everything else, close things are more connected than distant things. The Inverse distance weighting (IDW) interpolation tool was used for this analysis to visualize the emission level of N_2O gas because it works fine with my data.

2.2.1.1 Working procedure

- The Collection of data was carried out twice a month. As such, the average value for the N_2O gas emission was determined for each month with the use of Excel.
- The Excel file was added to Arc GIS using the add X-Y data because each point was coordinated.
- WMS layer was added from https://geoportaal.maaamet.ee to aid the digitalization of the study site.
- The digitalized wetland was placed in a newly created empty shapefile and exported. After this, the shapefile was imported, and the coordinate system was set to be the same as the data frame using the project tool.
- IDW Interpolation was then carried out at 10metres to show the difference in the level of N_2O gas emission at the various sample points of the wetland. This process was also repeated using the wetland water table depth data.

2.2.2 Wetland parameters and N_2O gas emission

To assess how wetland parameters such as water depth influence gas emissions, the simple excel Pearson Correlation coefficient (r) was utilized using the CORREL function because this Correlation analysis is suitable for easy determination of the linear relationship between two or more variables (Mondal and Mondal, 2016). The correlation results can either be positive or negative. Zero r values indicate no relationship between any two variables, and $r = 1$ or $r = -1$ indicates a perfect relationship. Thus, the strength can be anywhere between 0 and ±1 (Khanal, 2016). Further chats were produced with the use of Excel to verify the relationship between the wetland parameters. The statistical calculation, which includes the determination of average values for N_2O emission alongside percentages, was also done with the use of Excel.

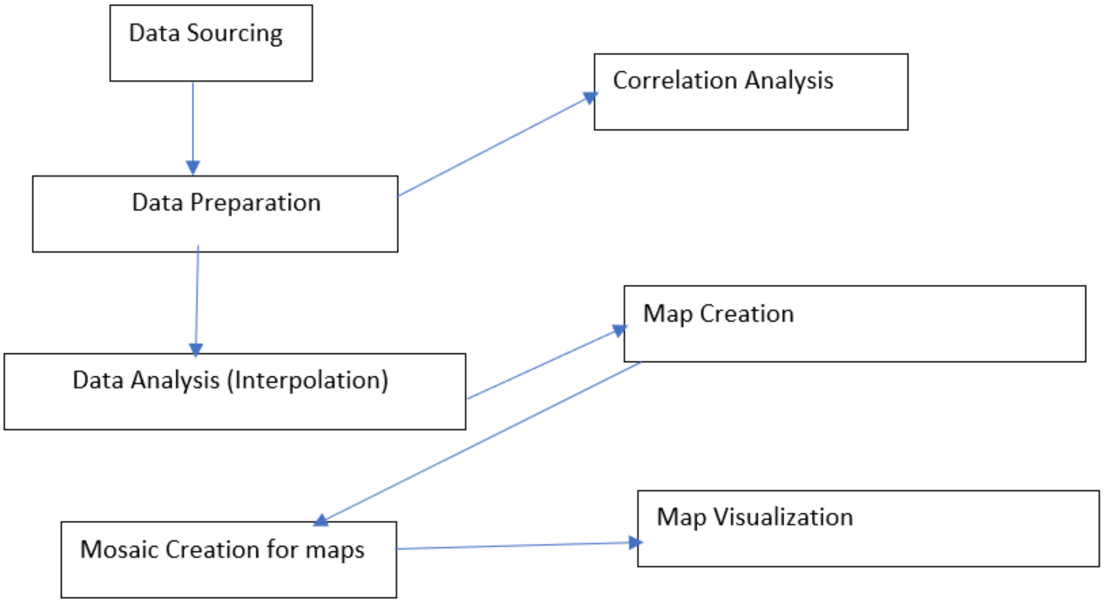

Figure 6: Simplified workflow diagram.

3. Results

This section presents the results obtained from the spatial analysis carried out to suit the aim of this master thesis.

3.1 N_2O gas emission

To model the level of N_2O gas emission in the study area, there was a need to carry out spatial analysis with IDW interpolation. A total of 27 maps (Appendix Figure 1-3) were obtained from May 2018 to November 2020, from which a visualization was made (*https://photos.app.goo.gl/t3oLRAygg68tRTD38*). These maps and visualization show that the Vända CWs is emitting N_2O gas that ranges from low (0-30 µg m^{-2} hr^{-1}) to high (above 400 µg m^{-2} hr^{-1}) at various sampling locations. N_2O emission also had a clear seasonal dynamics as the emissions were much higher during summer months (June, July, and August) ranging to as high as 1600 µg m^{-2} hr^{-1} compared to the autumn, spring, and winter months where the emission ranges from 0 - 200 µg m^{-2} hr^{-1} (Figure 7, Figure 8, Supplementary Figures 1-3).

Concerning the sample points, VM1.1 shows the highest average emission level (above 400 µg m^{-2} hr^{-1}.) in most of the months throughout the study period (Figures 7 - 8 and the visualization). This emission can be observed to be almost eight times higher than from the other points. VM1.1 is followed by VM1.3, which at some points, although in few cases (mostly in the winter months), shows emission slightly equal to VM1.2, VM1.4, and VM2.4. In general, the least average emission (0-30 µg m^{-2} hr^{-1}) occurs from other points such as VM 1.5, VM 1.6, and VM 2.6. In contrast, the maximum emissions were observed at VM 1.1, having a value of up to 2600 µg m^{-2} hr^{-1} (Figure 13).

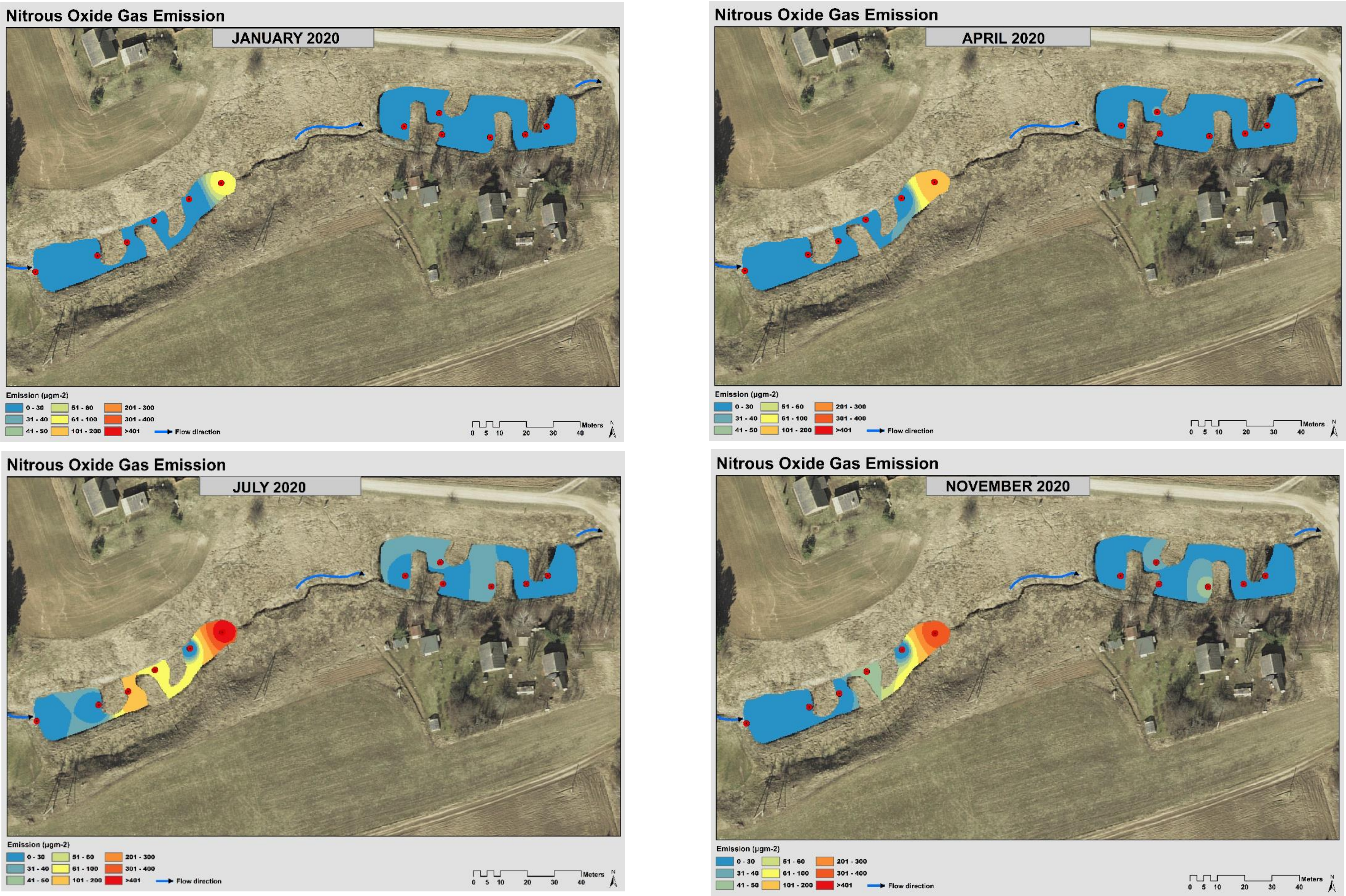

Figure 7: Maps for N_2O gas emission in winter (January), spring (April), summer (July), and autumn (November) 2020. Red dots indicate sampling spots.

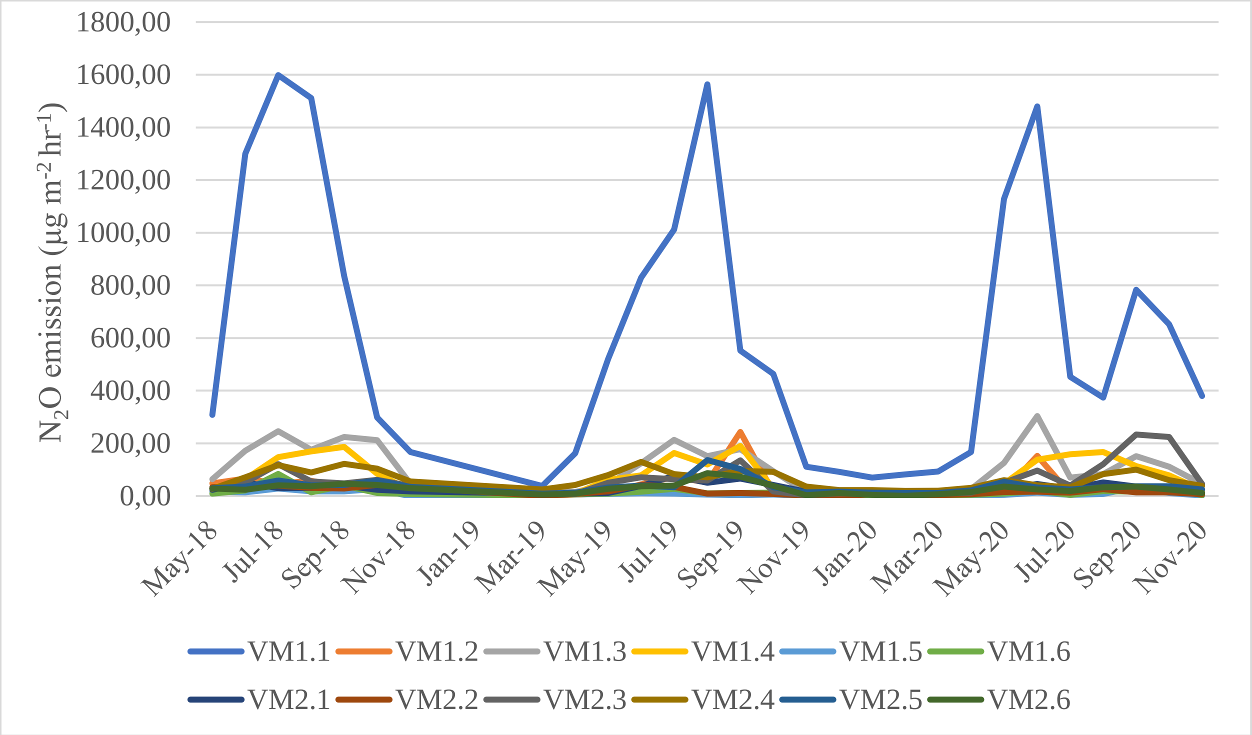

Figure 8: Line graph showing mean N_2O monthly gas emission for study period from different sampling points.

3.2 Nitrogen compounds, dissolved oxygen, and total organic carbon

Figure 9 shows boxplots of the inflow and outflow concentration of TN (a), NO_3-N (b), oxygen (c) and, TOC (d) at the inlet (IN) and outlet (OUT) of the wetland. From the figure 9, it is seen that the inflow and outflow levels seem not to be the same for almost all the compounds. It was observable that the average concentration of NO_3-N and TN increased at the outlet. For example, the NO_3-N (Figure 9a) shows a mean average value of 2.7 mgL^{-1} at the inlet and 4.2 mgL^{-1} at the outlet with a minimum and maximum value ranging from 2.17 mgL^{-1} to 7.0mgL^{-1} at the inlet to 2.40 mgL^{-1} and 9.60 mgL^{-1} at the outlet.

TN (Figure 9b) shows a minimum value ranging from just above 2 mgL^{-1} at both the inlet and outlet to a maximum value of about 6.42 mgL^{-1} and 9.12 mgL^{-1} at the inlet and outlet, respectively, while an average mean value of 4.7 mgL^{-1} at the inlet increases to 6.34 mgL^{-1} at the outlet. TOC, on the other hand, shows a decreasing mean value from 44 mgL^{-1} at the inlet to 36 mgL^{-1} at the outlet (9c).

Both the inlet and outlet Dissolved oxygen concentration mean values were similar (9d). The inlet shows a concentration of 9.51 mgL^{-1} which is almost the same as the outlet, that shows a value of 9.56 mgL^{-1}. However, dissolved oxygen minimum and maximum value range from just above 2 mgL^{-1} and 12 mgL^{-1} respectively at both locations.

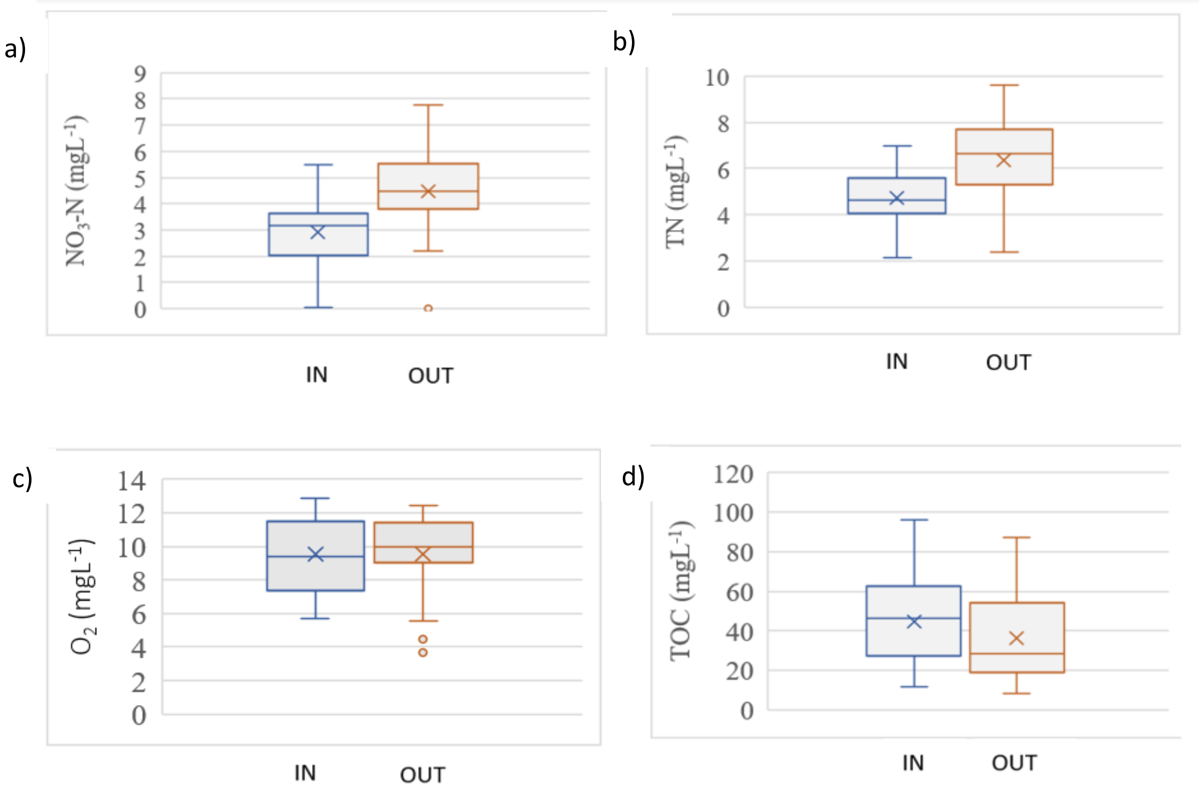

Figure 9: NO$_3$-N (a), TN (b), O$_2$ (c), and TOC (d) concentration in the inflow (IN) and outflow (OUT) of the study site. Horizontal lines show medians, crosses show means, boxes are the interquartile range and whiskers are the min-max values.

3.3 Vegetation and flow rate

The vegetation of the study area was estimated using information from GPS mapping and the MapInfo profession 10.5 software. The result (Table 1) shows a gradual increase over the years since the construction of the wetland. The first wetland and second wetland increased by approximately 52% and 11% respectively in 2018, and by 2020, this growth had increased to approximately 83% for the first wetland and 64% for the second wetland representing an increase above 100% from its initial state in 2016 (Table 1).

Table 1: Study location vegetation cover (source: Author and Supervisors)

Year	1st wetland %	2nd Wetland %
2016	30	1
2017	41	2,50
2018	52,00	10,50
2019	66,30	49,50
2020	83	63,7

The average flow rate value was observed to be approximately 5L/s with a minimum and maximum value of about 3L/s and 8 L/s, respectively (Figure 10). The maximum average flow rates correspond to months in the autumn (October and November 2020, for example) and spring seasons (May 2019 and March 2020), while the minimum average value was recorded in the summer months (July 2019 and 2020, for example). However, there was slight increase observed in the month of June 2018 (Figure 10).

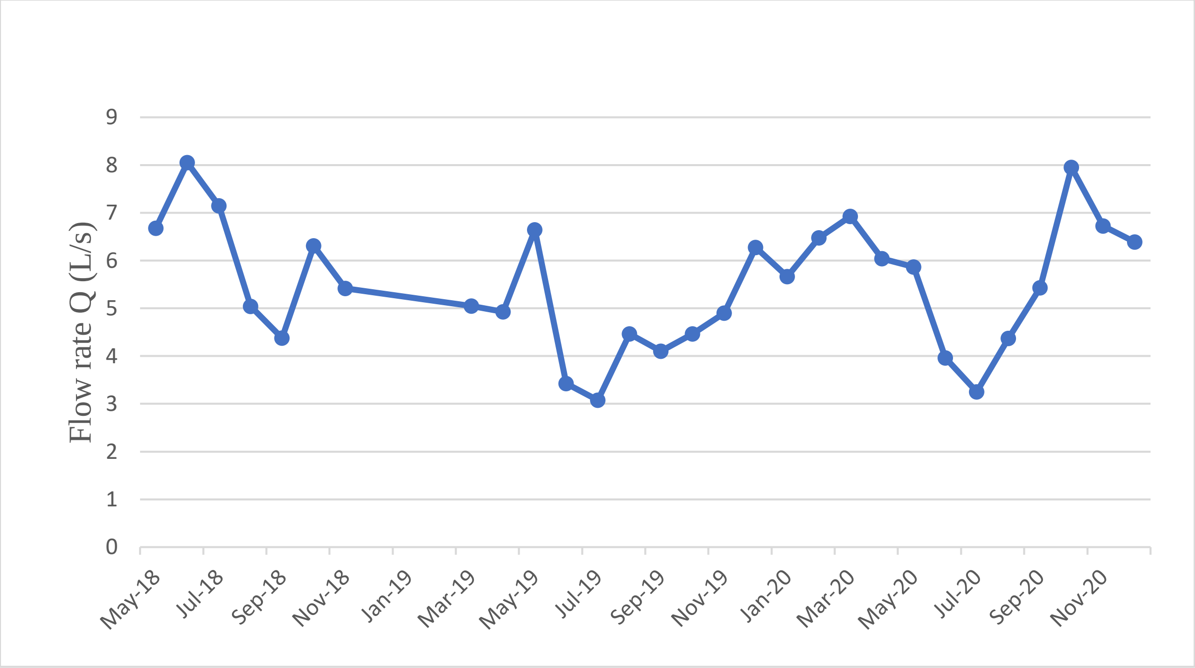

Figure 10: Average monthly flow rate

3.4 Other wetland parameters.

Since the aim of this thesis, as earlier stated, was not only to model the N_2O gas emissions but rather to delineate the possible mechanisms or parameters that are controlling this emission, a simple Pearson correlation coefficient for some parameters measured from the study location was performed with the use of Excel. Table 2 shows the results from the correlation analysis conducted. Water depth shows a correlation value of -0.66 while turbidity, ORP, pH, and conductivity show a correlation value of -0.09, 0.11, 0.22, and 0.02 rounded to 2 decimal places, respectively.

Table 2: Pearson coefficient correlation result for other measured wetland parameters (1 ≥ r ≥ -1)

	N_2O Emission	Water Table Depth	Conductivity	ORP	Turbidity	pH
N_2O Emission	1					
Water Table Depth	-0,66	1				
Conductivity	0,02	-0,22	1			
ORP	0,11	-0,06	0,35	1		
Turbidity	-0,09	0,07	-0,28	0,33	1	
pH	0,23	-0,09	0,66	0,53	0,03	1

When comparing the results obtained for the various parameters, to the Pearson correlation standard coefficient r-value that ranges from -1 to +1, only water dept shows a moderate to strong linear negative correlation (Table 2) with the emission of N_2O gas. However, it should be noted that the data measured from the sample location for pH value cannot be completely trusted due to some observed malfunctioning of the pH measuring instrument during sample collection.

3.4.1 Water depth

Because of the observed correlation results, it was further necessary to model the changes in the water depth within the various wetland sections. The findings (Appendix Figure 4-6 and visualization) show that water depth changes within the study location. Water depth values range from about 0cm to above 80cm at the various sample points. Concerning the different wetlands, the second wetland shows a value that mainly lies within 21cm to 30 cm. In comparison, wetland one displays a more extensive variation between the various points with a value that ranges from as low as 0cm to above 70 cm for virtually all the season (Figure 11). VM 1.5 and VM 1.6 have the maximum water level above 70cm, while the minimum value (below 20cm) is observed at VM 1.3 and VM 1.1(Figure 3, 11 and visualization).

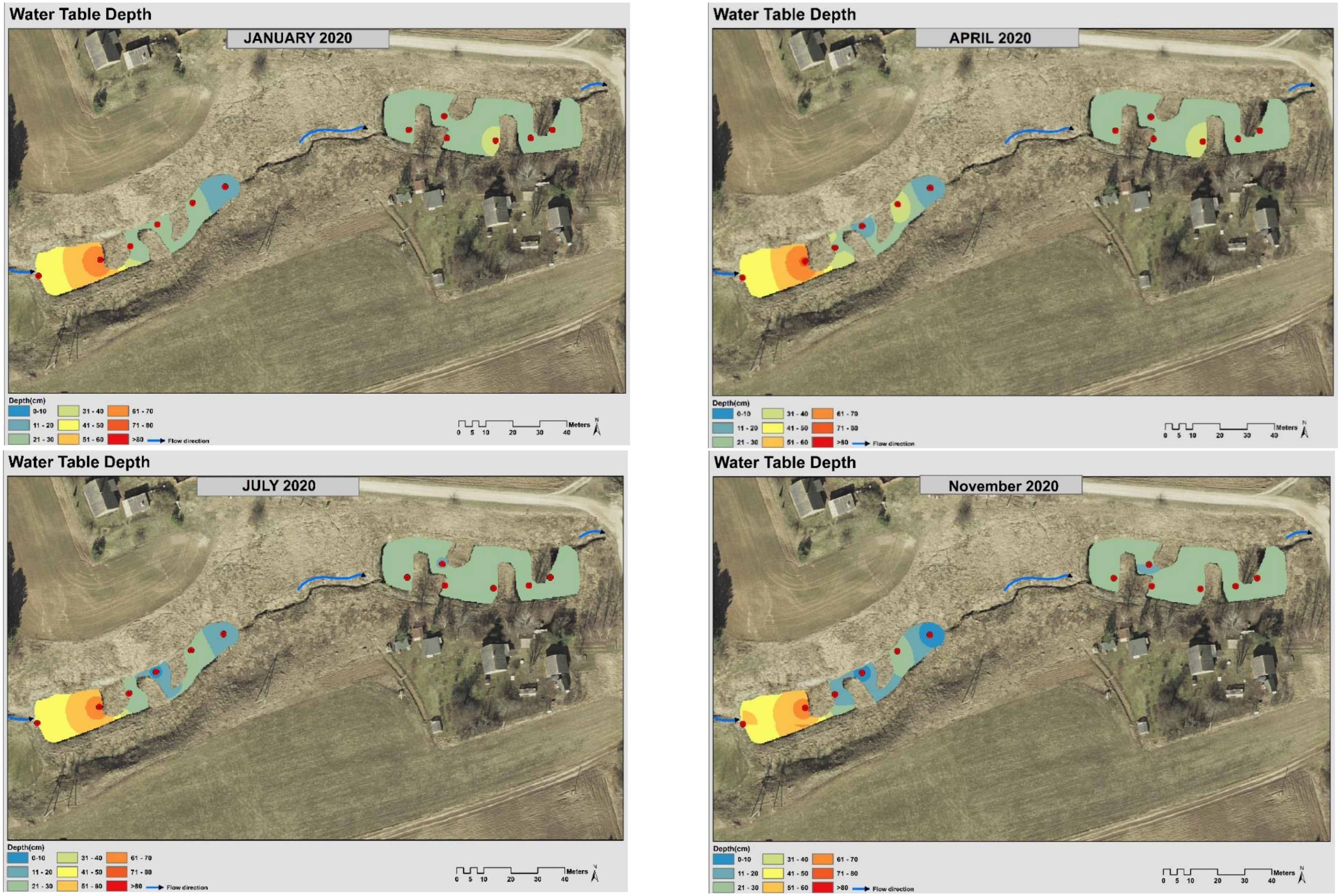

Figure 11: Maps for water table depth in winter (January), spring (April), summer (July), and autumn (November) 2020. Red dots indicate sampling spots.

A scattered plot was further driven to show the water level at which N_2O emissions tend to be highest within the study area. Figure 12 shows that the highest emission (between 400 and 2600 µg m^{-2} hr^{-1}) occurred at locations where water depth is lesser than 15cm. The lower the water depth, the higher the emission (Figure 12), supporting the negative correlation result obtained earlier (Table 2). It was further, necessary to create a map to clarify the region with the lowest average water table depth, particularly considering any sections that are less than 15cm in depth since most emissions were observed to be at this region. The findings (Figure 13) depict VM1.1 as the only points with an average water level depth below 15cm. Thus, the emission of N_2O from VM1.1 is almost eight times higher than from other sampling points, which will make this section as a hot spot for the N_2O emission (Figure 12 and 13).

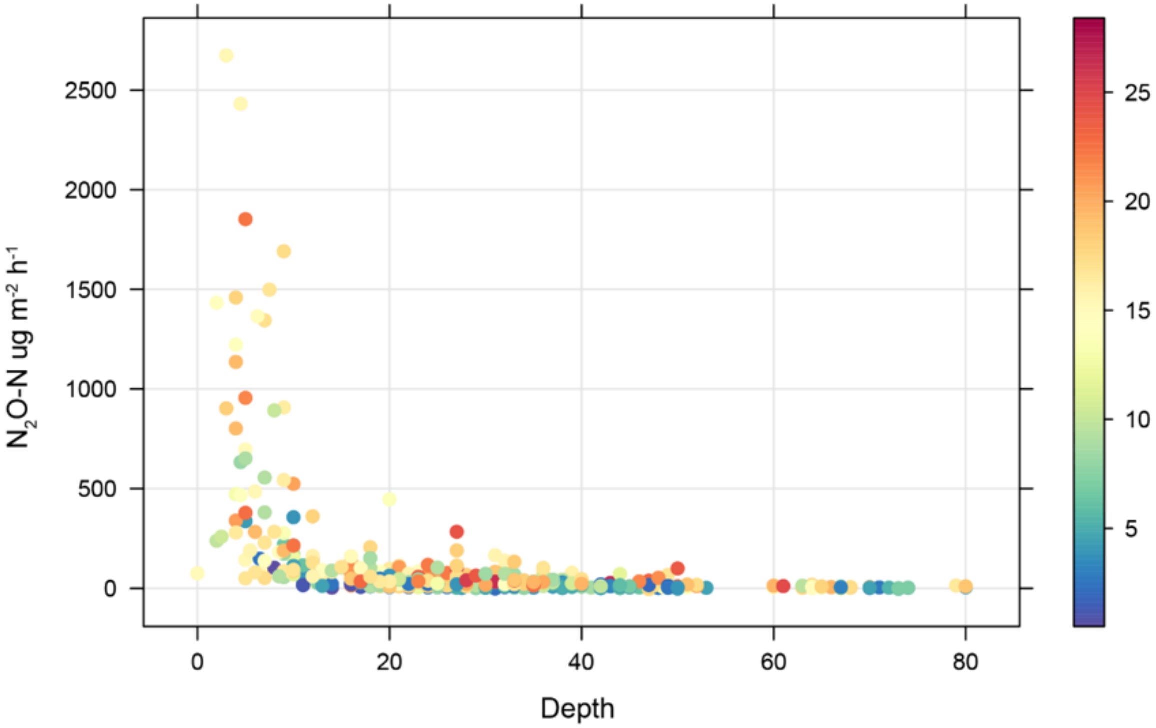

Figure 12: Scatter plot for N_2O emission against water table depth (R^2=0.36). Scale on the right shows water temperature at the sampling spot.

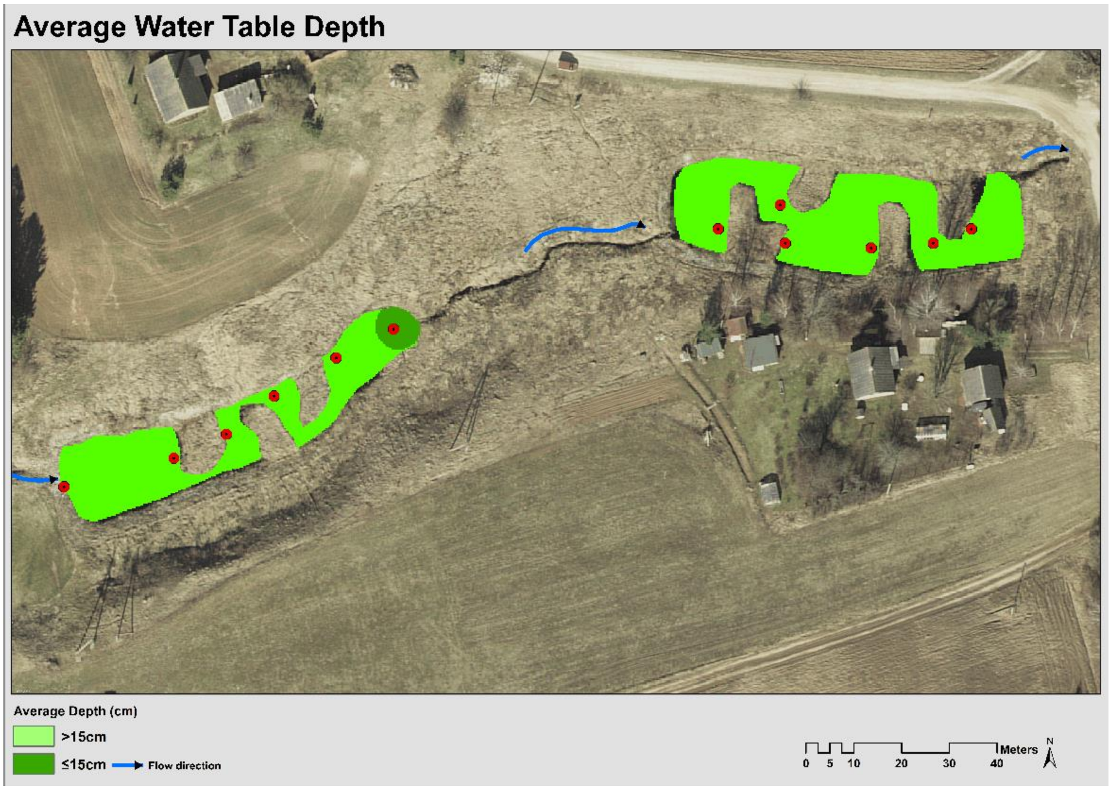

Figure 13: The map showing a region where the average water table depth is less than 15cm. Red dots indicate sampling locations.

To provide a possible wetland design model, it was, therefore, necessary to understand how large VM1.1 is in terms of area to be emitting such quantity of N_2O taking also into consideration the growing and non-growing season. The findings (Table 3) show that VM1.1 has an area of about $300m^2$, which is only approximately 7% of the total wetland area. However, during the growing season the relatively small area contributed almost about 86% of the total N_2O emission (Table 3). Even at the cold season, the small shallow water area was significantly higher emitter than other areas.

Table 3: Statistical data for the level of emission from the study location taking some factors into consideration.

Total wetland area	4400m^2
Total area with water table depth <15cm	300 m^2
Total area with water table depth >15cm	4100m^2
% of area with water table depth < 15cm.	6.8%
Total emission from water level >15 cm (g m^{-2} month^{-1}) from May 2018 to November 2020.	1g gm^{-2}month^{-1}
Total emission from water level <15 cm (g m^{-2} month^{-1}) from May 2018 to November 2020.	13g gm^{-2}month^{-1}
Emission from water table depth <15cm for growing season (May-October for 2018, 2019 and 2020)	12gm^{-2}month^{-1}
% emission from water table depth <15cm for growing season (May-October for 2018, 2019 and 2020).	85.7%

4. Discussion

4.1 N_2O emission

With the application of geospatial data analysis, 27 maps were obtained for N_2O gas emission to depict the level of emission from May 2018 to December 2020 in the study location. From the maps (Figure 7 and 8) and visualization (https://photos.app.goo.gl/t3oLRAygg68tRTD38), one can depict the fact that the Vända CW is emitting N_2O at all points of the wetlands, although at a different level. This is thus, in line with previous research (Sovik et al., 2007; Wang et al., 2008; Tallec et al., 2008; Huang et al., 2012; Manda et al., 2014; IPCC 2019, among others) which have all in one way or the other, described CWs as a source of GHG.

4.2 N_2O gas emission and driving mechanism.

N_2O gas emission from the standpoint of previous research (Wang et al., 2008; Wu et al., 2009; Parameswari et al., 2017) was described to be controlled by several factors. These factors include water table depth, temperature, microbial activities (nitrification and denitrification), and vegetation. The denitrification process involves the reduction of NO_3-N to N_2O and N_2 (Mander et al., 2014) and has been identified as the major process that produces N_2O in an anoxic/anaerobic environment, where nitrate is transformed into a removable gaseous product, unlike nitrification that only involves the conversion of nitrate in the presence of oxygen (Shi et al., 2018). Thus, the action of denitrifiers in the production of N_2O depends primarily not only on factors such as water table depth and vegetation but on the availability of N compounds and carbon (Vymazal, 2007).

The availability of these nitrogen compounds in the study location (Figure 9) supports the fact that denitrification is probably taking place in the wetland and, as such, is playing a role in the emission of N_2O. Nitrous oxide emissions increase with NO_3^- concentrations (Pärn et al., 2018). This is also supported by the decreasing value of the TOC from 44 mgL^{-1} at the inlet to 36 mgL^{-1} at the outlet (Figure 9) indicating its utilization by microorganism. Denitrification will not occur or will occur too slowly to remove enough nitrogen if there is not a ready supply of biodegradable carbon available (EPA 2013).

Generally, the increase of N compounds in the outlet is surprising because previous research (Koskiaho et al., 2005; Borin & Tocchetto, 2007) has shown high removal efficiency of these compounds, although few studies have shown quite a low removal during the cold season

(Grinberga & Lagzdins, 2017). Thus, one would expect the value of the N compounds to reduce at the outlet since denitrification and other processes should aid their removal or reduction (Vymazal, 2007). This increase might be because the built wetland is situated in a valley (44–47 m a.s.l.) surrounded by intensively controlled agricultural land (48–55 m a.s.l.) and therefore experiences groundwater seepage, which can be a significant factor in increasing NO_3-N concentrations (Kasak et al., 2018; Kill et al., 2018). The presence of oxygen could also be playing a role since both the inlet and outlet show very high mean oxygen values of about 9.51mgL^{-1} and 9.56mgL^{-1}, respectively. If enough NO_3^- is present but oxygen is low, denitrification will be completed, resulting in emission of N_2 (Wang et al., 2014). Oxygen is, thus, a limiting factor to denitrification (Shcherbak et al., 2014), which only occurs in water areas with low oxygen concentrations (Tournebize et al., 2017). Kasak et al. (2018) also suggested oxygen as a limiting factor to denitrification within the Vända constructed wetland. Other factors that could be playing a role include water depth, vegetation, and flow rate, among others. For example, a study conducted by Kill et al. (2018) shows that increased flow rate reduces water residence time, which in turn reduces the nutrients removal processes.

4.2.1 Temperature and vegetation response

Shi et al. (2018) described temperature and plant uptake to be among the factors supporting the main treatment process like denitrification. This has been supported by Stefanakis et al. (2014) who noted nitrogen removal to be positively influenced in the presence of vegetation. Plants can influence denitrification through a variety of mechanisms, including nitrate and water competition, oxygen consumption, and so on (Malique et al., 2019). They promote nutrients mitigation and removal through absorption, sorption, and degradation through the biofilms they help for microbial consortia function (Stefanakis et al., 2014). Vända CW shows growth in the level of vegetation (Table 2) over the years since construction in 2015 (Kasak et al., 2018), which could have influenced the level of N_2O emission throughout the CWs since denitrification takes place in the vicinity of the substrate (Fennel et al., 2009). Taking a closer look at the visualization, one would observe more N_2O emission from the first wetland compared to the second in almost all the months in the study period. This was, however, expected because during construction, in 2015, the first wetland (Figure 3) was planted with cattail (*Typha latifolia*) (4 plants per m^2) while the second wetland was left to colonize naturally (Kasak et al., 2018). Thus, since the first wetland has more vegetation, it is only expected to have more emission considering the role of plants in denitrification (Fennel et al. 2009; Stefanakis et al. 2014; Malique et al. 2019; Ioannidou & Stefanakis 2020). However, Plants have also been shown in some studies to acidify the environment through the absorption of CO_2 that can alter the pH level (Saleh et al., 2009), thus, affecting some denitrifying bacteria (Kasak et al., 2018) that cannot live under such acidify environment (Saleh et al., 2009). Therefore, vegetation alone cannot account for the emission in this study but must be considered with other factors such as temperature.

The effect of temperature, on the other hand, could be observed from figure 7, 8 and the visualization. Higher emissions are occurring in the summer months (June, July, and August) compared to autumn, spring, and winter months, except in few cases such as October 2020, where emission was like in July 2020. An increase in temperature leads to an increase in the microbial activities that favor denitrification (Zhang et al., 2020). As such, we do not see much N_2O gas emissions in early spring, late autumn, and in winter due to the possibility of lower temperatures that could fall below the temperature where denitrifiers are more effective (Jauhiainen et al., 2014). Denitrifiers become more effective at a temperature ranging from $7^{O}C$ to about $24^{O}C$ (Saleh et al., 2009). Above this temperature, N_2O efflux decreases (Pärn et al., 2018). However, Denitrifier populations, on the other hand, may adjust to higher temperatures, resulting in increased N_2O emissions (Farquharson et al., 2008). And below this temperature, the effect of denitrifiers might reduce gradually, and at $< 4°C$ degrees, denitrification might be put to a stop because low temperature poses a challenge to denitrifying populations (Saleh et al., 2009). Thus, more emission is seen in the CWs in summer months compared to other months at all points (VM 1.1- VM 2.6; Figure 3 and visualization).

In addition, Zhang et al. (2020) stated that a drop in temperature causes a rise in dissolved oxygen (DO) concentration in the effluent of a nitrifying biofilter, which may be another explanation for poor denitrification output at low temperatures leading to low N_2O emission. This could also be a reason why we observed almost zero-emission (visualization and maps (appendix)) specifically in the second wetland at some points for winter months (December, January, and February). However, previous researches (Teh et al., 2011; Dijkstra et al., 2012) have found contradictory relationships between temperature and N_2O emissions, ranging from strongly positive to strongly negative, demonstrating that temperature alone cannot explain N_2O fluxes and must be considered alongside other factors (Pärn et al., 2018), such as water depth and flow rate, among others.

4.2.2 Water level depth and flow rate

The production of N_2O in CWs, designed to treat agricultural runoffs, has been attributed to nitrification and denitrification, which can be influenced by the water level depth (Wang et al., 2008). A study conducted by Parit et al. (2020), shows that nitrification occurs at water depth where oxygen levels are slightly higher, and thus nitrifying bacteria oxidize ammonium to nitrate, which is then translocated to a reduced layer, or deeper layer, where it is susceptible to denitrification. VM1.1 shows an average water level depth of less than 15cm (Figure 13 and visualization) throughout the entire study period with an extreme value for N_2O emission compared to other points (Table 3). This is followed by VM1.3, which also shows a high level of N_2O gas emission compared to other points. Although at some very few points (August 2020, for example), VM1.4 shows almost the same water depth and thus similar N_2O emission level to VM1.3 while other points (VM2.6, for example) with water level rising to above 60cm at some months shows little or no emission (visualization). This is because, at a higher water table, there is an increase in the water-filled pore spaces of the CWs topsoil's where the mineral N substrate is largely located (Dobbie & Smith 2006), resulting in a decrease in dissolved oxygen concentration, which can cause N_2O production (Pärn et al. 2018), but not enough to complete denitrification (Liang et al., 2016).

In general, the emission of gases from all study points in this study corresponds negatively to the water table depth (Figure 12 and Table 2). This observation corresponds with research such as those carried out by Dobbie and Smith (2006) and Manda et al. (2014), which shows a negative relationship between water table depth and N_2O emission. Although VM1.1 is only approximately 7% of the total wetland area (Table 3), it contributes 13g m^{-2} $month^{-1}$ out of the total observable emissions of about 14 g m^{-2} $month^{-1}$ (Table 3), making this point to be the major source of N_2O in the CWs throughout the study period. The decrease in emissions with increasing water table depth is most likely due to improved aeration of the uppermost CWs soil horizons, which reduces the number of anaerobic zones where denitrification is most likely to occur (Dobbie & Smith 2006). As a result, we see almost zero or no emission at depths greater than 70cm (Figure 12).

However, the level of N_2O gas emission could also be influenced by flow rate since a higher flow rate indicates lesser retention time and thus lesser nutrient removal (Kill et al., 2018), which in turn slows down denitrification (Zhang et al., 2020). It is observable from figure 10 that higher emissions in most cases corresponded with the summer months when flowrate was lower compared to autumn and spring, which has higher runoff and snowmelt, respectively (Kasak et al., 2018).

4.3 Growing and non-growing season

Since the purpose of the Vända CWs is to treat agricultural pollutants, there was the need to understand the level of depth variation over the growing seasons and the non-growing season. The visualization shows that during the growing season (May to October of each year), water level seems to be much lower especially at VM1.1 and VM1.3 and as such more emission is seen in these months compared to the non-growing season. The possible reason could be due to the high inflow of agricultural runoffs due to the hilly nature of the CWs (Kasak et al., 2018), which can lead to more deposition of sediments, thereby contributing to the growth of shallow water table (Ioannidou & Stefanakis, 2020). As a result of this higher water level depth in the growing season, about 86% of the total monthly emission was seen during this period (Table 3), leaving just about 14% to the non-growing season.

5. Conclusion and recommendation

Over the years, the constructed wetland has been utilized for the treatment of agricultural runoffs, but it is a source of greenhouse gases. To effectively track the release of these gases, it is essential to conduct objective, independent research to identify patterns and evaluate the effectiveness of mitigation strategies and initiatives. Thus, the importance of this thesis which aims to model the N_2O emissions from Vända in-stream free water surface constructed wetlands located at Uhti village, Tartu, Estonia.

This study documented the importance of water table depth as a controlling factor for the emission of N_2O in a free water surface constructed wetland. Observed changes in N_2O emissions correlated negatively with water table depths. The effects in the variation of water depth were attributed to the fact that a decrease in emissions with increasing water table depth is most likely due to improved aeration of the uppermost CWs soil horizons, which reduces the number of anaerobic zones where denitrification is most likely to occur.

However, temperature exudation also affects microbial processes. In this way, more emission was observed in the summer months compared to autumn and winter months when microbial activity ranges from high to medium to low, respectively. Other factors which show a controlling effect in the emission include vegetation and flow rates. Higher vegetation denotes higher emission in the first wetland compared to the second, while flow rate was seen to play a role by affecting the nutrient retention time, which is important for denitrification and, as such, affecting the emission. In general, the highest N_2O emission of up to 2600 $\mu g\ m^{-2}\ hr^{-1}$ was observed at a water level of less than 15cm, which corresponds to a region with high vegetation, higher temperature, and at some points, lower flow rate.

Modeling predicted that fluxes during the growing season would be reduced by 86% if the water level could be held to no less than 15 cm above the ground level. However, increasing it to 70 cm would reduce them by over 99%.

Although the emission level can be said to be low compared to higher emissions from anthropogenic activities that can trigger climate change, it could still be reduced if the water level can fluctuate and, if possible, increase at all points to above 15cm. Thus, as a recommendation, several possible wetlands around the country should be studied using this kind of modeling technique employed in this study, which has proven to be good to visualize the N_2O emissions (but also the emission of other GHG´s) and, thus, point out problem areas and potential solutions. However, if this type of CW is to be built, it should have a water depth of at least 15cm.

.

Lämmastikdioksiidi emissioonide modelleerimine Vända avaveelise tehismärgala näitel

Isaac Okiti

Kokkuvõte

Kasvuhoonegaase eraldub pidevalt atmosfääri, kujutades ohtu nii inimeste tervisele kui ka keskkonnale. Üheks oluliseks kasvuhoonegaasiks on dilämmastikoksiid (N_2O) ehk naerugaas. N_2O gaasimolekulide atmosfääri sattumisel kahaneb osoonikiht, mis kaitseb maapinda kahjulike ultraviolettkiirte eest. Selle protsessi tagajärjeks on kliima soojenemine.

Tehismärgalasid on aastaid kasutatud põllumajandusliku äravoolu puhastamiseks. Sellegipoolest on tegu potentsiaalsete kasvuhoonegaaside allikatega. Kasvuhoonegaaside eraldumise tõhusaks jälgimiseks on tarvis objektiivseid ja sõltumatuid uuringuid, tuvastamaks gaasivoogude dünaamikat ning hindamaks leevendusstrateegiate ja -algatuste tõhusust. Sellest lähtuvalt on antud uurimuse eesmärk modelleerida N_2O heitkoguseid Vända voolusängile rajatud avaveelisest tehismärgalast, mis asub Uhti külas, Tartu maakonnas.

Antud töös püütakse vastata järgmistele küsimustele:

1. Millised on N_2O emissioonid uurimisalalt ja kuidas need varieeruvad erinevates märgala sektsioonides?
2. Millised on peamised N_2O emissiooni reguleerivad mehhanismid?
3. Milline võiks olla Vända tehismärgala mõju kliima soojenemisele?
4. Kas ja kuidas saaks N_2O emissioone uuritaval märgalal täiendavalt vähendada?

N_2O kontsentratsioone mõõdeti kaks korda nädalas kaheteistkümnest proovipunktist. Lisaks koguti samadest punktidest veeproove. Proove analüüsiti Tartu Ülikooli geograafia osakonnas, kasutades Shimadzu GC-2014 gaasikromatograafi. Üldlämmastiku ja üldorgaanilise süsiniku sisaldust veeproovides analüüsiti Vario TOC/TN analüsaatoriga (Elementar GmbH, Saksamaa). Nitraatse lämmastiku (NO_3-N) kontsentratsioonid määrati ioonkromatograafia abil, kasutades standardiseeritud meetodeid (APHA-AWWA-WEF, 2005). Muud parameetrid, nagu hapniku kontsentratsioon, vee temperatuur ja elektrijuhtivus , mõõdeti kohapeal YSI PRO kaasaskantava analüsaatori abil.

Saadud andmeid analüüsiti interpoleerimise ja Exceli abil. Koostati 28 graafikut N_2O emissioonide ja vee sügavuse kohta. Uurimisperioodil aastatel 2018–2020 ulatusid N_2O emissioonid Vända uurimisalal keskmiselt kuni 400 $\mu g\ m^{-2}\ h^{-1}$ madala veega aladel (veesügavus <15 cm) ja keskmiselt kuni 30 $\mu g\ m^{-2}\ h^{-1}$ sügavama veega (veesügavus >15 cm) alades. Kusjuures madala veega aladel olid kõrgeimad emissioonid kuni 2600 $\mu g\ m^{-2}\ h^{-1}$. Seega madala veega aladel olid N_2O emissioonid võrreldes teiste proovipunktidega 92% kõrgemad.

Antud uurimuse tulemused näitasid vee sügavuse tähtsust N_2O emissiooni kontrolliva tegurina. N_2O heitkoguste muutused korreleerusid negatiivselt veetaseme sügavusega. Emissioonide suurenemine veetaseme langemisega on tõenäoliselt tingitud tehismärgala ülemiste settekihtide paremast õhustamisest, mis vähendab denitrifikatsiooniks sobilikku anaeroobset keskkonda.

Märgala põhjasette mikroobiprotsesse mõjutab omakorda ka temperatuur, mis avaldab omakorda mõju N_2O emissioonidele. Mikroobide aktiivsus suvekuudel on kõrge, sügiskuudel keskmine ja talvekuudel madal. Seetõttu on N_2O emissioonid soojematel suvekuudel võrreldes sügis- ja talvekuudega kõrgemad. Lisaks mõjutavad N_2O heitkoguseid taimestik ja vee voolukiirus. Tihedama taimestikuga aladel on N_2O heitkogused mõnevõrra suuremad. Voolukiirus mõjutab toitainete kinnipidamisaega, mis on oluline denitrifikatsiooni toimimiseks, mõjutades seeläbi emissioone.

N_2O emissioonid tehismärgaladelt on võrreldes kliimamuutusi põhjustavate antropogeensete heitkogustega väikesed. Sellegipoolest on oluline vähendada emissioone ka tehismärgaladelt. Käesoleva magistritöö üks olulisemaid tulemusi oli see, et tehismärgalade rajamisel ja hooldamisel tuleb arvestada sellega, et veesügavus märgalas oleks vähemalt 15 cm või rohkem.

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Appendix

Table 1: Sample location points with coordinates.

Location symbols	X-coordinates (Estonia)	Y-coordinates (Estonia)
VM1.1	6463460	659673
VM1.2	6463454	659661
VM1.3	6463446	659648
VM1.4	6463438	659638
VM1.5	6463433	659627
VM1.6	6463427	659604
VM2.1	6463481	659741
VM2.2	6463478	659755
VM2.3	6463477	659773
VM2.4	6463486	659754
VM2.5	6463478	659786
VM2.6	6463481	659794

Figure 1: Maps for N_2O gas emission for some months (Summer) in the study location.

Figure 2: Maps for N_2O gas emission for some months (Autumn) in the study location.

Figure 3: Maps for N_2O gas emission for months in spring and winter.

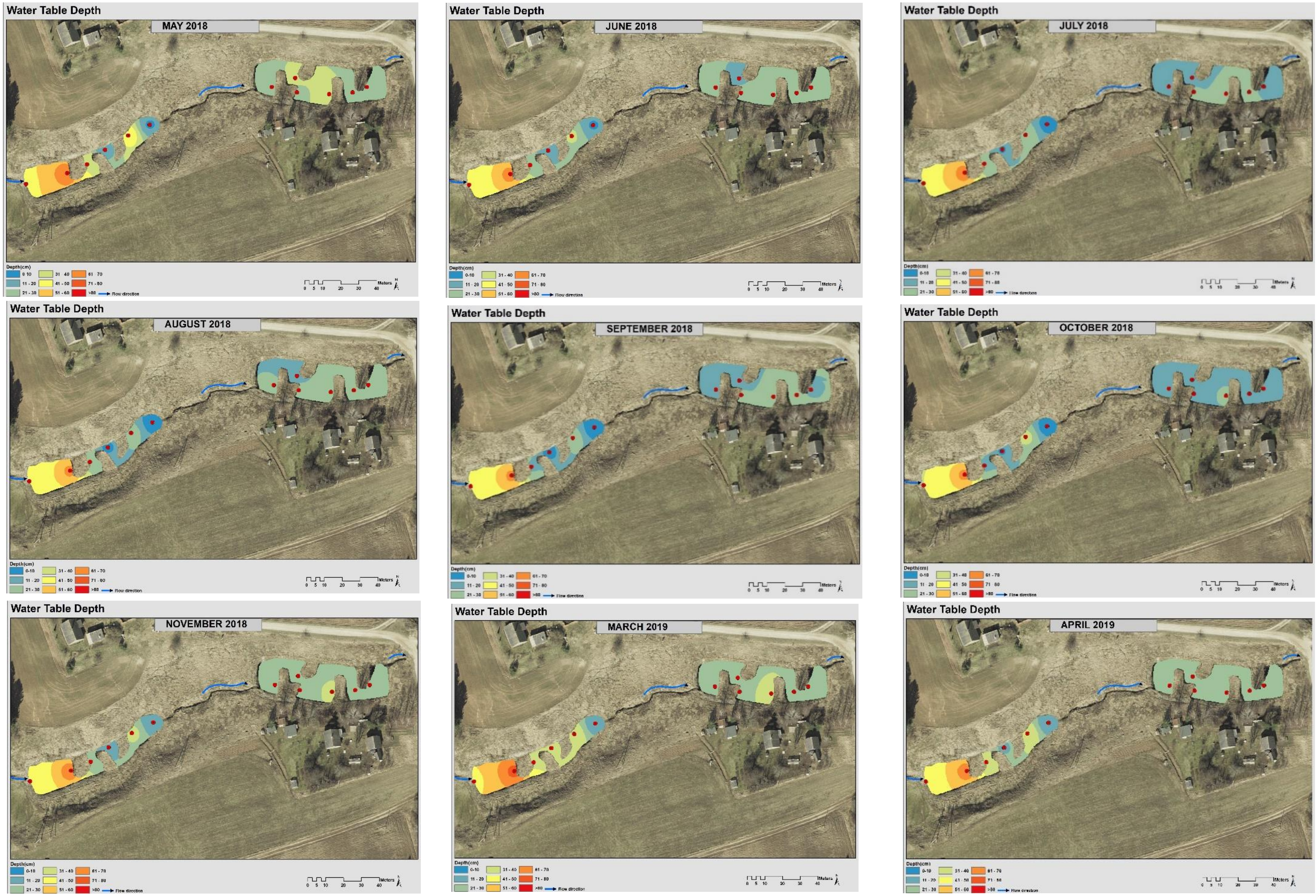

Figure 4: Maps for water table depth from May 2018 to April 2019 in the study location.

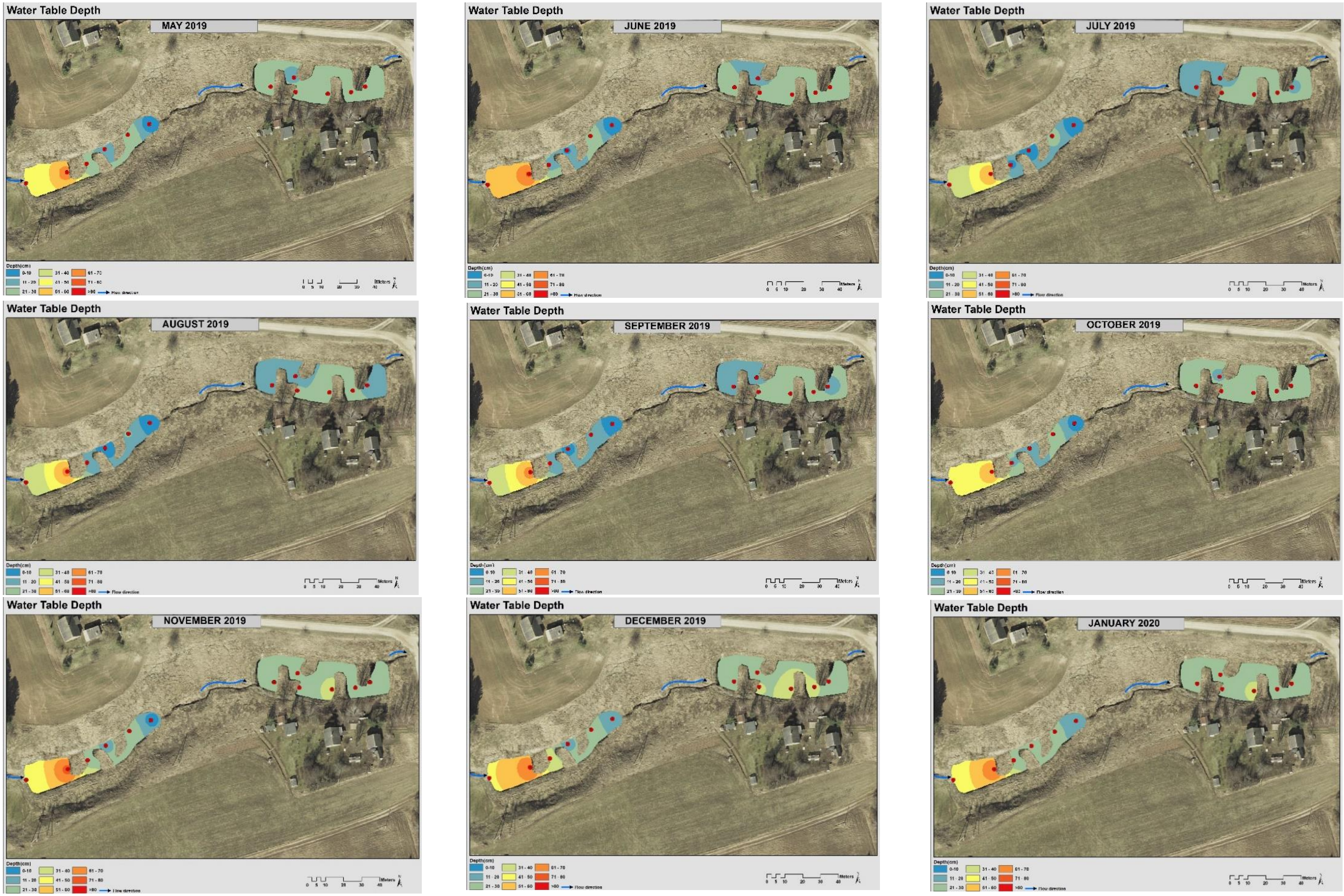

Figure 5: Maps for water table depth from May 2019 to January 2020 in the study location.

Figure 6: Maps for water table depth from February 2020 to October 2020 in the study location.

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