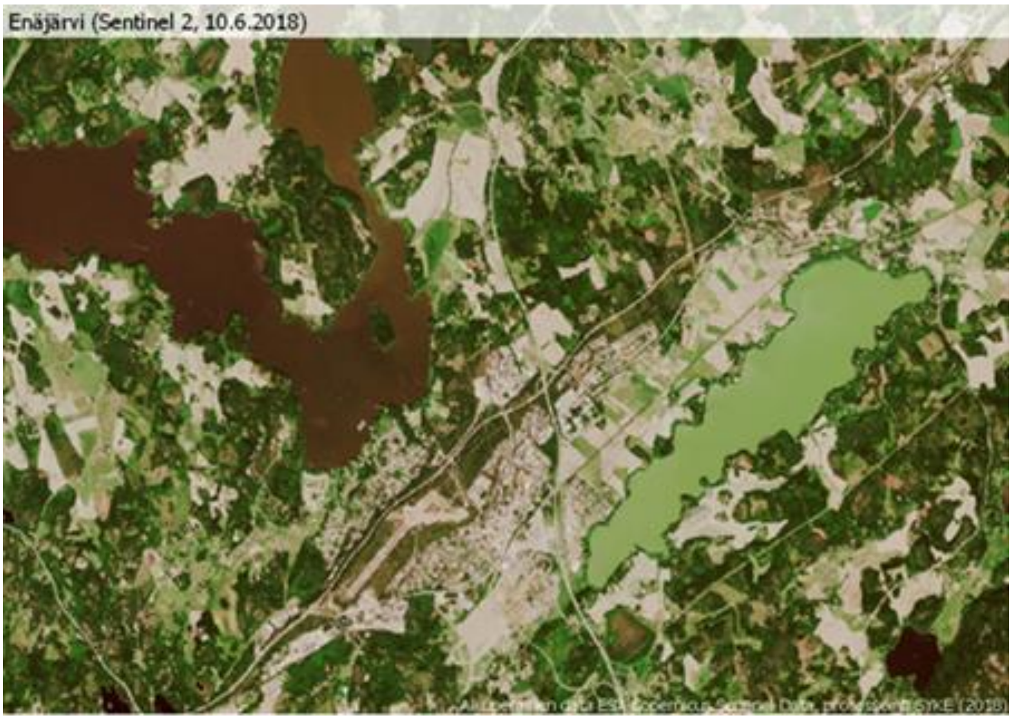
Modelling the water quality of lake Enäjärvi, Finland

Master thesis



M.Sc. Thesis by Harmen Knap

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Water Systems and Global Change Group

Frontpage image: Contains Copernicus Sentinel data (2018), (SYKE, 2018)

Modelling the water quality of lake Enäjärvi, Finland

Master thesis

Master Thesis Water Systems and Global Change Group in partial fulfilment of the degree of Master of Science in International Land and Water Management at Wageningen University, the Netherlands

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SUMMARY

The word eutrophic is composed of the Greek words *eu*, meaning “well”, and *trophe*, meaning “nourishment” and henceforth means, in the literal sense, “well-nourishment”, indicating a situation with a more than sufficient food supply. Eutrophication of lakes and other surface waters across the globe is thought to be mainly caused by excessive nutrient loading of nitrogen (N) and phosphorus (P) into the water system (Yang et al., 2008). Many lake systems have deteriorated over the past decade due to eutrophication, often caused by anthropogenic activities (Nielsen et al., 2014). In shallow lakes, eutrophication can trigger rather abrupt shifts from a clear and often desired state to a turbid and often less preferred state. The external nutrient load at which a shift can happen, or at which a threshold chlorophyll-a concentration is surpassed, is called the critical nutrient load. Lake Enäjärvi, a shallow lake in the south of Finland, is defined as “bad” according to the Water Framework Directive and algal blooms occur frequently (SYKE, n.d.).

To find out whether additional measures are needed to prevent the critical load from being exceeded to regularly the following question is answered in this thesis: *“How does the nutrient loading in lake Enäjärvi relate to the critical nutrient loading and what are lake restoration measures that can be taken to either keep the lake in a clear state or switch the lake from turbid to clear?”*

To this end, the current state of the water quality in Enäjärvi is analysed based on the historical water quality (2000-2017), critical nutrient load, and several lake restoration measures for Enäjärvi are analysed. By using the process-based models PCLake+ and VEMALA v.3 the chlorophyll-a and nutrient concentrations were simulated. The models were validated for these variables against observational data and compared with each other. Both models simulated concentrations in similar ranges as the observational data. R^2 and Mean Relative Absolute Error (RE) values fell within a range indicating a good model performance except for NH_4 and NO_3 (R^2: 0.17-0.22 and 0.5-0.57 and RE: 16.41-17.73 and 7.42-19.57 respectively for NH_4 and NO_3). For most variables, the models agree fairly well with each other. In general PCLake+ captures the higher values better then VEMALA V.3, in the lower ranges VEMALA V.3 performs better compared to PCLake+.

PCLake+ was used to determine the load response curve (chlorophyll-a concentration-response to external nutrient load) and the critical nutrient load in lake Enäjärvi for chlorophyll-a thresholds of 20, 30, 40, and 50 mg chlorophyll-a per m³. The critical nutrient load for the 20 and 30 mg chlorophyll-a/m³ thresholds are 0.29 and 0.37 mgP $m^{-2}d^{-1}$. These levels were exceeded 4 and 6 times in the period 2000-2017 respectively. The 40 and 50 mg chlorophyll-a/m³ thresholds were not reached in summer.

Since the critical nutrient loadings have been surpassed several over the period 2000-2017, various lake restoration measures were explored and their effects on the critical nutrient loading tested. In this study I looked at in-lake system measures, these are measures aimed at increasing the critical nutrient load. The measures include adding a marsh zone, reducing the wind fetch, biomanipulation by fish catch, water level alterations, or combinations of these measures. All measures, except for a reduction of the water level in spring, resulted in a lower peak of the load response curve to under the 30 mg chlorophyll-a/m³ threshold, therefore the further mentioned results relate to the 20 mg chlorophyll-a/m³ threshold. Interesting, a reduction of the water level results in an increase of the load-response curve peak, likely due to the lack of alternative stable states, hampering a domination of macrophytes. Adding a marsh area of 30% of the lake's surface to the lake resulted in a 48% increase of the 20 mg chlorophyll-a threshold critical nutrient load to 0.43 mgP m^{-2} d^{-1}. A marsh zone of 10% results in a 28% increase of this critical nutrient load, while a reduction of the wind fetch by 50% results in a 13% increase. Regular fish catch and reducing the water depth in spring did not increase the critical nutrient load for the 20 mg chlorophyll-a/m³ threshold. Reducing the water depth in spring by one meter did however increase the critical nutrient load of the 30 mg chlorophyll-a threshold by 13% (from 0.37 to 0.42 mgP $m^{-2}d^{-1}$). It should be noted that implementation possibilities of these measures is dependent on a lot more factors which are not researched in this thesis, results merely show their simulated effect on the load response curve and critical nutrient loading.

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1. INTRODUCTION

The word eutrophic is composed of the Greek words *eu*, meaning “well”, and *trophe*, meaning “nourishment” and henceforth means, in the literal sense, “well-nourishment”, indicating a situation with a more than sufficient food supply. Eutrophication of lakes and other surface waters is thought to be mainly caused by excessive nutrient loading, nitrogen (N) and phosphorus (P), into the water system (Yang et al., 2008). Many lake systems have deteriorated over the past decade due to eutrophication, often caused by anthropogenic activities (Nielsen et al., 2014). In shallow lakes, eutrophication can trigger rather abrupt shifts from a clear and “pristine” state to a turbid and often less preferred state, because it is accompanied by a loss in biodiversity and dominance by phytoplankton (Scheffer, 2009). The state of the ecosystem has next to the influence on the biodiversity effect on various ecosystem services such as drinking water extraction, recreation and fisheries since algal blooms (high chlorophyll-a concentrations) often lead to oxygen depletion or toxin production (Janssen et al., 2019). To prevent such shift or to return to the clear state, it is important to assess the critical nutrient load at which the ecosystem state changes.

1.1. Theoretical background

In his influential article, Forbes (1887) described lakes as microcosms, reflecting what happens in the world itself. He painted an image of the web of interactions present in (lake) ecosystems which shape the struggle for survival of the present species. Hereby, according to Scheffer (2009, pg 109), Forbes article is “often considered the start of ecology as a scientific discipline”. The article characterizes the steady balance of organism in relation to each other as one of the most remarkable phenomena in life. In respect to this, if this balance between two or more species gets disrupted, the most well-adjusted species to the change causing this disbalance will “in time crowd out his poorly-adjusted competitors” (Forbes, 1887 pg 550). Increased external nutrient loading can be an example of such a disruption, causing catastrophic shifts in lake ecosystems. Scheffer (2009), in his book, and also earlier articles (e.g. Scheffer, 1993), argues that such big shifts in natural- but also in social- systems can be explained by critical transitions. Understanding critical transitions in the system of interest, in this case lakes, can help in developing ways to manage change.

The predominant feedback in shallow lakes that causes such shifts is shown in Figure 1. Shallow lakes can have two contrasting states between which such a transition can take place. Overly simplified, the two main dominant states are a turbid and a clear state (Janse et al., 2010)(Figure 2).

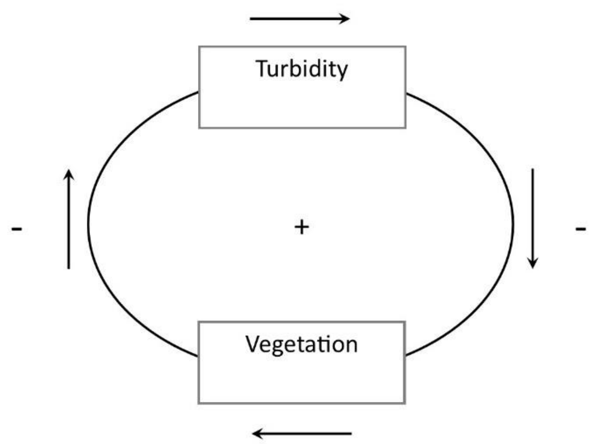


Figure 1: Turbidity - vegetation feedback loop. – indicates a negative feedback, meaning that an increase of a component leads to a decrease of the next one (in this example; an increase in turbidity leads to a decrease of vegetation). A + indicates a positive feedback, in this case an increase of one components stimulates and increase in the next one. The total sum of + and – determines weather the sum is positive or negative. In this case 2 minuses become a plus, henceforth, the feedback loop is positive (more turbidity leads to more turbidity and the other way around).

The basic mechanism of turbidity and vegetation is that an enhancement of turbidity, caused by for example eutrophication and excessive algal growth, causes a decrease in vegetation, which in turn increases the turbidity again. The cycle can also be in reverse where more vegetation leads to less turbidity, which would increase the vegetation again. Figure 2 shows this feedback in more detail and in relation to eutrophication.

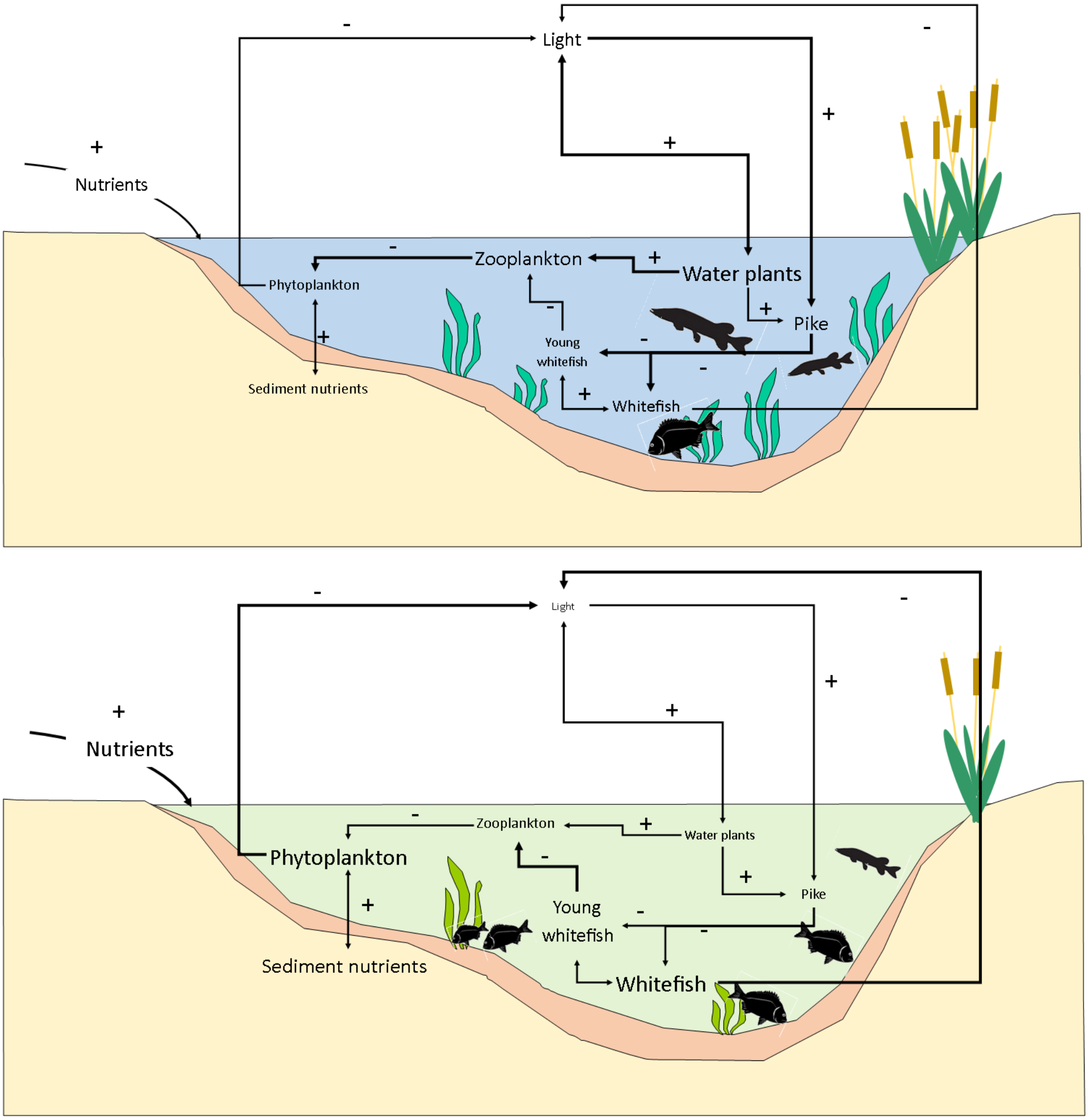


Figure 2: A clear (above) and turbid (under) lake system and the dominant feedback processes The thicker the lines and the larger the text, the stronger the interaction (source: own creation based on Scheffer, 2009)).

Figure 2 provides a graphical overview of the interactions in a clear and in a turbid lake system in more detail. In a clear system, abundant (submerged) macrophytes can grow and cover large parts of the lake. These water plants provide shelter for young pike from adult pike, and shelter zooplankton, hereby limiting predation by young planktivorous fish. The presence of macrophytes limits resuspension due to waves, hampering turbidity and favouring conditions for sight hunting fish such as pike. Pike, or similar species, in turn, prey on (young) whitefish, providing protection for the zooplankton and controlling the whitefish population. Zooplankton grazes on the phytoplankton and by doing this limiting turbidity by extreme algal growth. Adult whitefish or bottom-feeding fish do not hunt on sight and resuspend the sediment while hunting for prey, enhancing turbidity. A turbid system is characterized by a lack of aquatic vegetation. Direct effects are more resuspension of sediments due to waves and less uptake of nutrients, providing the phytoplankton with extra food and enhancing their growth. Young pike here have a lack of shelter and the pike population is decreased substantially as compared to the clear state. With a lack of predation, the young whitefish population increases considerably and graze on zooplankton. The system gets dominated by bottom-feeding fish, further enhancing turbidity (Scheffer, 2009).

The nutrient loading at which a system shifts from one of the respective state to the other is defined as the critical nutrient load (Scheffer, 2009). The critical nutrient load is measured as the amount of nutrients under which a certain concentration of chlorophyll-a becomes apparent or where the chlorophyll-a concentrations show a very steep increase if seen from an initial oligotrophic state (low phytoplankton and nutrient concentrations), or steep decrease if seen from an initial eutrophic state (high phytoplankton and nutrient concentrations). For the estimation of the critical nutrient load, it is important to have an understanding of the load-response curve in the lake, describing the response of chlorophyll-a concentrations to certain nutrient loads. Figure 3 provides a representation of possible load-response curve types (based on Janssen et al., 2017). An assessment on the load-response curve, the critical nutrient load, and the present nutrient loading and related ecological state indicates how the nutrient load in a lake is positioned with respect to the critical nutrient loading. This distance between the critical nutrient load and the current nutrient load suggests how much increase in nutrient load the system can still absorb in case of an oligotrophic state- or how much nutrient load need to be reduced in case of a eutrophic state- before tipping to another state. This is of use for the design of water quality measures. If the shift from a turbid state to a clear state happens at a substantial lower nutrient loading relative to the opposite shift, the system shows hysteresis (e.g. figure 3C). I deem the peak of the load-response curve important because high chlorophyll-a might cause, even if not permanent, damage to the ecosystem.

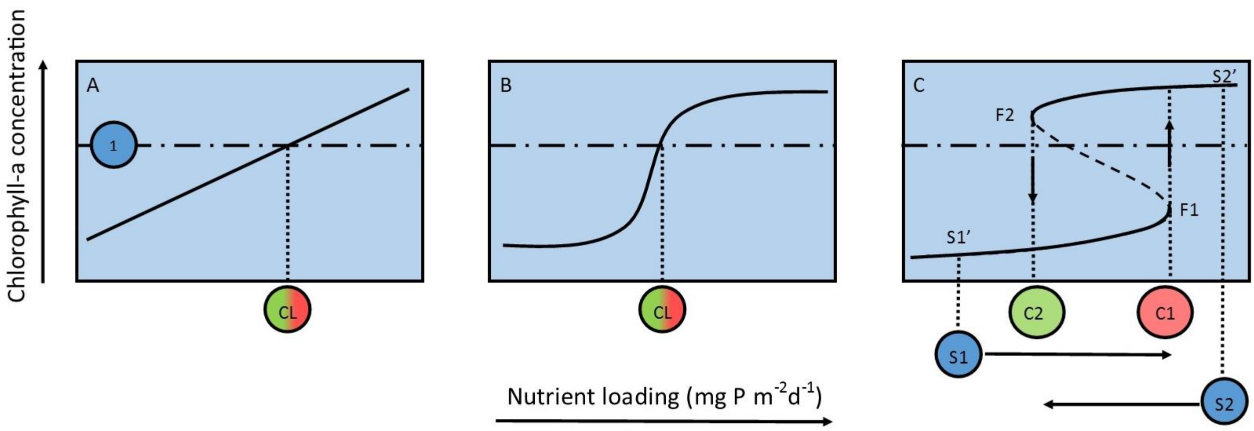


Figure 3: Load-response curves with hypothetical, human-defined, chlorophyll-a thresholds (dotted line indicated with number 1). A) a linear load-response curve where the chlorophyll-a concentration grows or declines proportionally with the nutrient loading. In this case, the critical nutrient load is sensitive to the human-defined chlorophyll- a threshold. B) a non-linear load-response without hysteresis, often typical for deeper lakes. Around a certain nutrient amount, the chlorophyll-a concentration changes rapidly until it stabilises again. The critical load here is less sensitive to the human-defined maximum allowable chlorophyll-a concentration, if the threshold moves up or down the critical load does not change much. C) depicts a non-linear load-response with hysteresis, typical for shallow lakes. At the bifurcation point F1 the system quickly tips into a turbid stable state (following the C1 line to the upper solid line, it passes the chlorophyll threshold). Reduction of the amount of nutrients to the same critical nutrient load as where the lake shifted does not help to flip the system back to the previous state. To reach a clear- oligotrophic- state the nutrient loading needs to be reduced to F2 and the critical load for oligotraphication (C2) (following the C2 line to the lower solid line). C shows two hypothetical initial states (S1 and S2), the distance of S1 to the C1 show how much nutrients the system can increase before becoming eutrophic, the distance of S2 to the C2 is the reduction needed for the system to become oligotrophic (Scheffer, 2009; Janssen et al., 2017).

1.2. STUDY SETTING

This thesis focusses on lake Enäjärvi, of which the ecological status is defined as poor according to the Water Framework Directive (WFD) (SYKE, n.d.). Monitoring of water quality started in 1961 and abundant data for the lake is available (Lake & Sea Wiki, 2014). The lake is located approximately 40 km northwest of Helsinki in the upstream region of the Siuntionjoki river, meaning that the water quality has an impact on the downstream water system (Figure 4). The lake is considered shallow with an average water depth of 3.22 m with a maximum depth of 9.1 meters. Still, the deeper regions of the lake occasionally stratify. The lake is approximately 492 ha in size with a catchment area of approximately 34 km² (Pellikka & Sammalkorpi, 2020; Lake & Sea Wiki, 2014). Around the lake, there are 17 wetlands or settling ponds to enhance the water quality. However, only three wetlands function well and retain approximately 20% of the P load they receive annually. The other wetlands are thought to be too small in relation to the catchment area to substantially retain nutrients. The wetlands are with one side connected to the lake (Pellikka & Sammalkorpi, 2020). According to Salonen et al. (1993) the water quality of the lake has deteriorated in 1928 due to a lowering of the water level. Furthermore, the nearby village Nummela discharged its sewage waters for 25 years into the lake until 1976, substantially fuelling the P content of the lake. Even though the sewage water was no longer directed towards the lake after 1976, sedimentary P had grown to such extent that algal blooms remained characteristic for the lake (Salonen et al., 1993). Therefore, lake Enäjärvi is still defined as eutrophic and ecological poor (Tuominen, Schultz, & Sillanpää, 2013; Kallio et al., 2001; SYKE, n.d.).

This thesis will contribute to the Blue Adapt project in Finland. The Blue Adapt project aims at securing the sustainability and resilience of water ecosystems, economics and social systems. Within Blue Adapt the first step is to identify the potential tipping points in (aquatic) ecosystems that can lead to catastrophic changes. One of the identified tasks in this step is to model the ecosystem state and response to amongst others the external nutrient loading (Blue Adapt, n.d.). By modelling water quality and assessing the resilience of the ecosystem the Blue Adapt project contributes to the objective of the Water Framework Directive (WFD) to prevent further deterioration of surface water (Blöch, 1999).

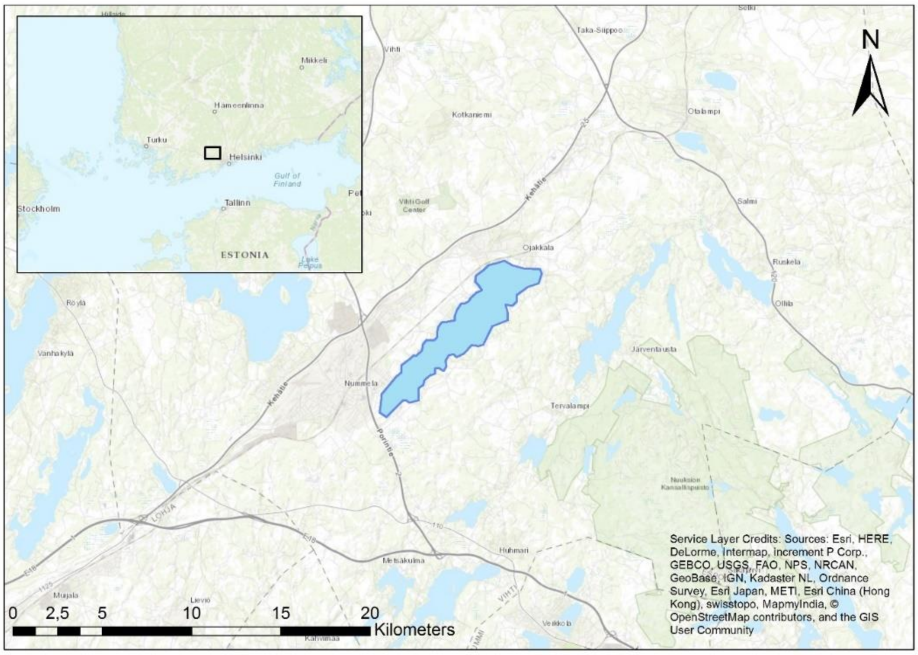


Figure 4: Location of lake Enäjärvi (own production, made with World Topo Map. For source layer credits see right bottom of the figure).

1.3. KNOWLEDGE GAPS AND RESEARCH QUESTIONS

The critical nutrient load of lake Enäjärvi is unknown, therefore at this moment comparison between the current nutrient loading in the lake and the critical nutrient load is still lacking. It is important to know how the nutrient loading relates to this critical nutrient load threshold to develop sufficient water quality management strategies to either keep the lake clear, in case of an oligotrophic situation or switch the lake from a turbid to a clear state in case of a eutrophic situation.

To assess how the nutrient loading in the lake relates to the critical nutrient load and what management strategies might be beneficial the following main research question is developed: *"How does the nutrient loading in lake Enäjärvi relate to the critical nutrient loading and what are lake restoration measures that can be taken to either keep the lake in a clear state or switch the lake from turbid to clear?"*

The main research question will be answered by answering the sub-questions defined in table 1.

Table 1: Sub-research questions

Research questions
1. What is the current state of the water quality of lake Enäjärvi and how do PCLake+ and VEMALA v.3 perform in relation to observational data?
2. What is the critical nutrient load of lake Enäjärvi and how does this relate to the past external nutrient loading to the lake?[1]
3. What are the effects of lake restoration measures on the critical nutrient load?

In this study, the PCLake+ and VEMALA v.3 models are used. PCLake+ is used to assess the critical nutrient loading and the VEMALA v.3 model output is used as input to PCLake+. Both models are also validated for their performance in simulating the historical water quality (2000-2017) of lake Enäjärvi

Figure 5 is an overview of the steps taken in this research. Before any of the steps can be taken to answer the sub-research questions the input and validation data needs to be gathered. Additional, the structure of VEMALA V.3 and PCLake+ will be anlysed to understand the models better and determine their input data needs, and how they produce their output. The first step, aimed to answer the first sub-research question, consist of the validation of both models and an assessment of the current state (Chapter 2.) In the second step, the critical nutrient load of lake Enäjärvi is calculated by PCLake+. The critical nutrient load is hereafter compared to the historical water quality of Enäjärvi to determine the need for lake restoration measures and of what order of magnitude (chapter 3.). The third step consists of modelling several identified possible lake restoration measures in PCLake+ (chapter 4.).

[1] By answering this question I can state whether or not I deem it necessary to analyse possible lake restoration measures

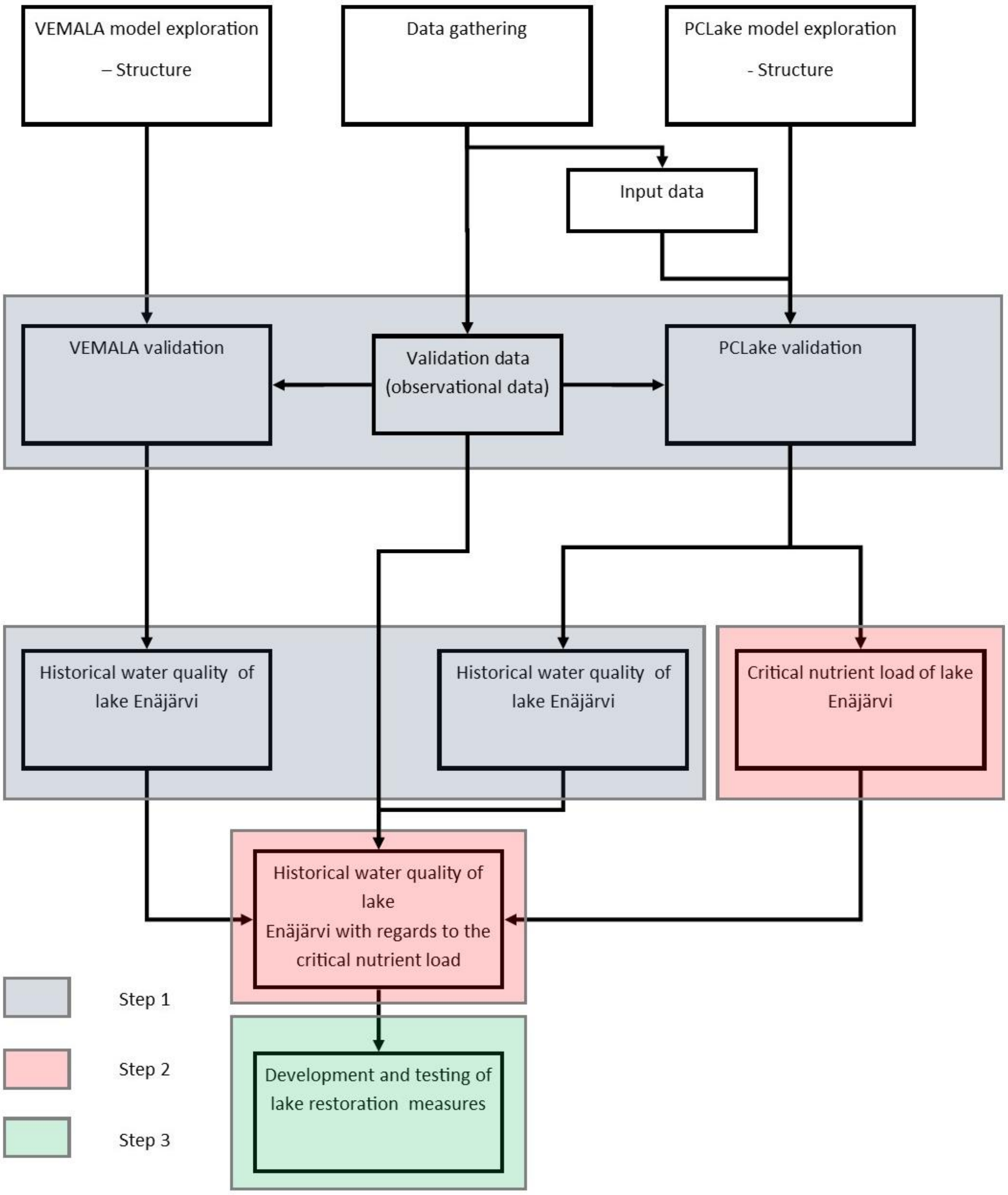


Figure 5: Research steps

The aim of this study is ultimately to assess the water quality of lake Enäjärvi, if lake restoration measures are needed, and which measures can be effective. With this, I aim to contribute to the field of modelling aquatic ecosystems. In the process of the water quality assessment, the performance of PCLake+ and VEMALA V.3 is analysed. The critical nutrient loading of the lake will be determined and the uncertainties around the results recognized. Lake restoration measures and their effectiveness in lake Enäjärvi will be explored, however, this analysis will be a mere indication of their relative effectiveness and are by no means absolute recommendations. The construction of a lake management or restoration plan is outside of the scope of this research.

2. VALIDATION AND CURRENT STATE

In this chapter I will answer sub-question 1: "*What is the current state of the water quality of lake Enäjärvi and how do PCLake+ and VEMALA V.3 perform in relation to observational data?*". Data on the current and historical water quality of a lake system, both in chemical- and ecological quality, provides information on the lake's historical dynamics. A general understanding of the lake system of interest provides insights into the current state and whether the lake switched once or periodically to an alternative state. Consequently, it provides insight into the starting point for water quality management. The current state and historical behaviour of the lake can be determined sole with observational data, however, in this study, I additionally chose to use PCLake+ and VEMALA to bridge gaps in observations. Moreover, the data serves as a validation of the models so they can be used for the assessment of the critical load and scenario runs for management strategies in the next chapters. This chapter contains a description of VEMALA V.3 and PCLake+ and the methodology used to run PCLake+ and validate the models. Furthermore, there is a results section with the validation and sensitivity results, and a discussion and conclusion regarded to the used methodology and found results.

2.1. MODEL DESCRIPTION

Modelling of water quality and the current water quality state, can be done by using Aquatic Ecosystems Models (AEMs). AEMs are frequently used for the quantification of ecosystem state and to develop management strategies (Janssen et al., 2019). Application of models allows scientists and managers to undertake virtual experiments which would in the "real" world be too risky, expensive, or impactful. AEMs can be statistical (including generalised additive models (GAMs)), or process-based. Statistical models have their relative simplicity to their advantage. However, statistical relations do not necessarily provide insight into the biochemical processes underlying such relation, implying that it is easy to miss steps or misjudge relations. All statistically-based models are found on data gathered in the past, henceforth, future predictions are hard to make. Process-based models are relatively complex as compared to statistical models. They, however, include a general theoretical grasp of relevant ecological, hydrological and chemical processes making them suitable for larger-scale usages (if calibrated) and future predictions. The data input, on the other hand, is more demanding than that of statistical models (Janssen et al., 2019).

Because of the benefits of process-based models such as their broad application possibilities, a basic understanding of processes, and the possibility to simulate water quality restoration measures this research will make use of the VEMALA v3. and PCLake+ models, developed in Finland and the Netherlands respectively. Both models are process-based models which simulate water quality variables and are used to assess the ecosystem states of lakes (Korpoo et al., 2017; Janssen et al., 2019; Janssen et al., 2019a).

2.1.1. EXTENDED DESCRIPTION PCLAKE+

PCLake is a process-based Aquatic Ecosystem Model which describes phytoplankton, zooplankton, fish, and macrophyte food web interactions and their response to external nutrient loading (Janssen et al., 2019). One of the main aims of the model is to analyse when a lake system shifts from a clear- vegetation dominated system to a turbid phytoplankton dominated system, and the other way around. The main water quality parameters the model calculates are chlorophyll-a concentration, transparency (Secchi), phytoplankton, and the density of submerged macrophytes (Janse, 1997). PCLake can quantitatively evaluate the effects of climatic change, aquatic management options, changes in the hydrology and morphology, or a combination of the aforementioned on a lake system (Janse, 2005; Nielsen et al., 2017). PCLake was originally developed to describe a fully mixed- non-stratifying- water body and the top sediment layer. This assumption, along with certain other assumptions, does not hold in situations with greater depths and different climatological situations (Janssen et al., 2019). To this end, PCLake+ is updated to account for different climate zones, and in case of thermal stratification, a hypolimnion can be switched on (Janssen et al., 2019). Spatial differences within the lake are not included in the model (Janse, 2005).

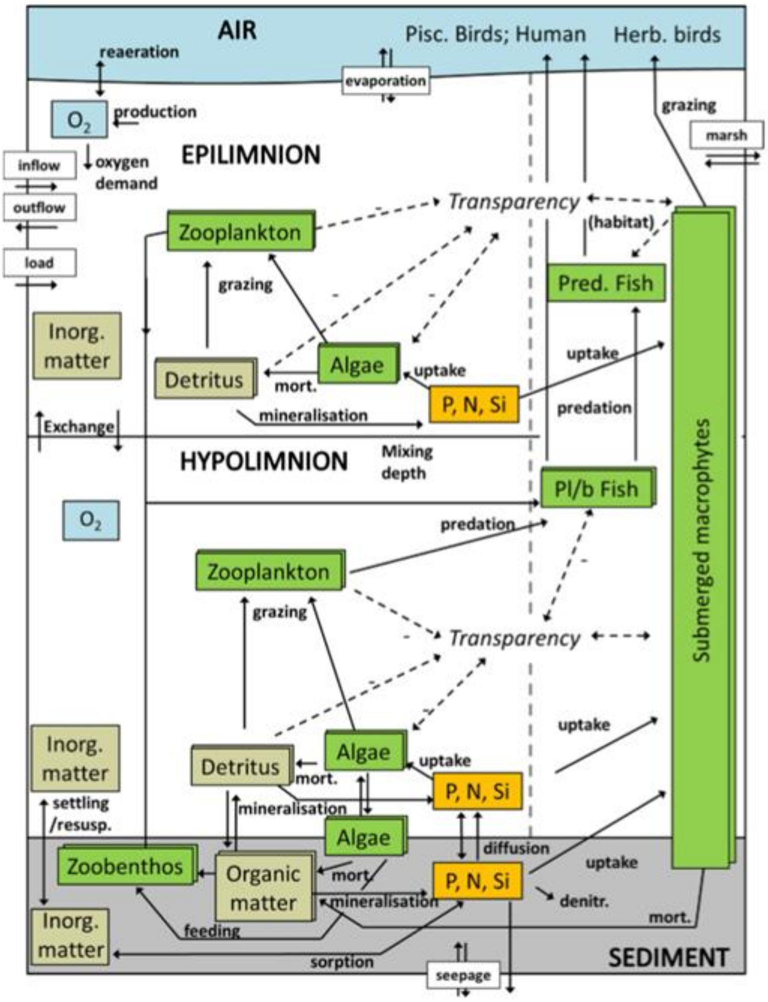


Figure 6 : Model framework of PCLake+ (Janssen et al., 2019a)

The model structure of PCLake+ is depicted in figure 6. The model uses for each state one coupled differential equation and consists of mass fluxes (e.g. food web relations) and some observed relations, mathematically translated into empirical formulas (e.g resuspension due to fish or macrophyte presence) (Janse, 2005). All biota represented in the model are modelled as functional groups, hereby the effects of the food web on the nutrient cycles are rationalized. The main groups in the water column are the zooplankton, fish (benthivorous/planktivorous and piscivorous), and the phytoplankton (algae: consisting of green algae, diatoms, and cyanobacteria). The groups in the sediment are the three kinds of phytoplankton, represented as a settled fraction, and the zoobenthos (Janse, 1997; Janse et al., 2008). The model furthermore includes submerged macrophytes. The sediment layer is incorporated in the model to account for the lakes “history” since historical nutrient pollution is often partly deposited in the sediment, which in turn releases it back into the lake. In addition to these functional groups, the nutrient cycles are present in the model as closed, except for external fluxes as the nutrient in- and outflow (Janse, 2005). The input data that is needed is shown in table 2

The entire set of processes in PCLake (not including the PCLake+ additional processes) can be found in the Appendixes of Janse (2005). Since Janse (2005) gives the most elaborate description of the processes in PCLake I will largely follow his work and only give a short description of the most important processes in the model. Furthermore, based predominantly on Janssen et al. (2019), I will include the main additions of PCLake+ to this description.

The processes in PCLake Janse (2005) identifies are 1) abiotic and microbial processes, 2) processes related to the phytoplankton module, 3) aquatic vegetation processes, 4) the food web interactions, and 5) the wetland module. In addition to this, PCLake+ has as main extra function the inclusion stratification processes. To this end, the water column is split into a hypolimnion layer and an epilimnion layer, separated at mixing depth. River in- and outflow are connected to the epilimnion where seepage and infiltration interact with the hypolimnion. Both the hypolimnion and epilimnion have individual state variables for the algae groups, inorganic matter, nutrients, zooplankton, oxygen and detritus. The fish and macrophyte groups are not physically restrained by stratification (Janssen et al., 2019).

The internal P loading is not standard included in PCLake+. To quantify the internal P load in lake Enäjärvi, and compare this to the output of VEMALA V.3, an extra formula was added to PCLake+.

InternalLoadEpi = (_tPDifPO4Epi_ + _tPResusDet_ + _tPResusPO4_ + _tPResusAIM_) / _uDepthWEpi_ (**Eq. 1**)

The internal P load consists of the dissolved PO_4 together with the P resuspension of detritus, PO_4, and inert matter, all divided by the water depth to come to a concentration in g P $*m^{-3}*d^{-1}$.

There are ample examples available where the PCLake(+) model is used to simulate the water quality in mostly shallow lakes, both for research purposes (e.g. Janse, 2005; Janse, 2010; Mooij et al., 2007; Schep et al., 2007; Nielsen et al., 2014; Mellios et al., 2015; Janssen et al., 2017(a)) and as a base for the design and appliance of water quality measures (Stowa, n.d.).

2.1.2. EXTENDED DESCRIPTION VEMALA V.3

VEMALA v.3 is similarly to PCLake(+) a process-based model, used for the estimation of water quality (Figure 7). The spatial extension of the model consists of roughly 58000 lakes in Finland (Janssen et al., 2019). The model simulates the transport of nutrients on land, lakes and rivers, hydrology, and nutrient processes (Huttunen et al., 2016). VEMALA v.3 includes a hydrological sub-model (WSFS), a field-scale terrestrial nutrient loading sub-model for the simulation of the nutrient loading from agricultural fields (ICECREAM), a terrestrial catchment-scale forest nutrient loading sub-model and a biogeochemical sub-model for the nutrient cycling in lakes and rivers (Korppoo et al., 2017). The biogeochemical sub-model (Figure 8) utilizes a simplification of the RIVE biochemical model to simulate the transport and processes affecting nutrients in the freshwater ecosystem, diatoms and cyanobacteria (phytoplankton), suspended sediments, and total organic carbon. The phytoplankton module is based on the AQUAPHY model (Korppoo et al., 2017). The WSFS sub-model is used for the modelling of the hydrological processes and simulates subsurface and baseflow, snow melt and accumulation, evapotranspiration, soil moisture and infiltration. The hydrological processes are needed to determine the discharge and load concentration.

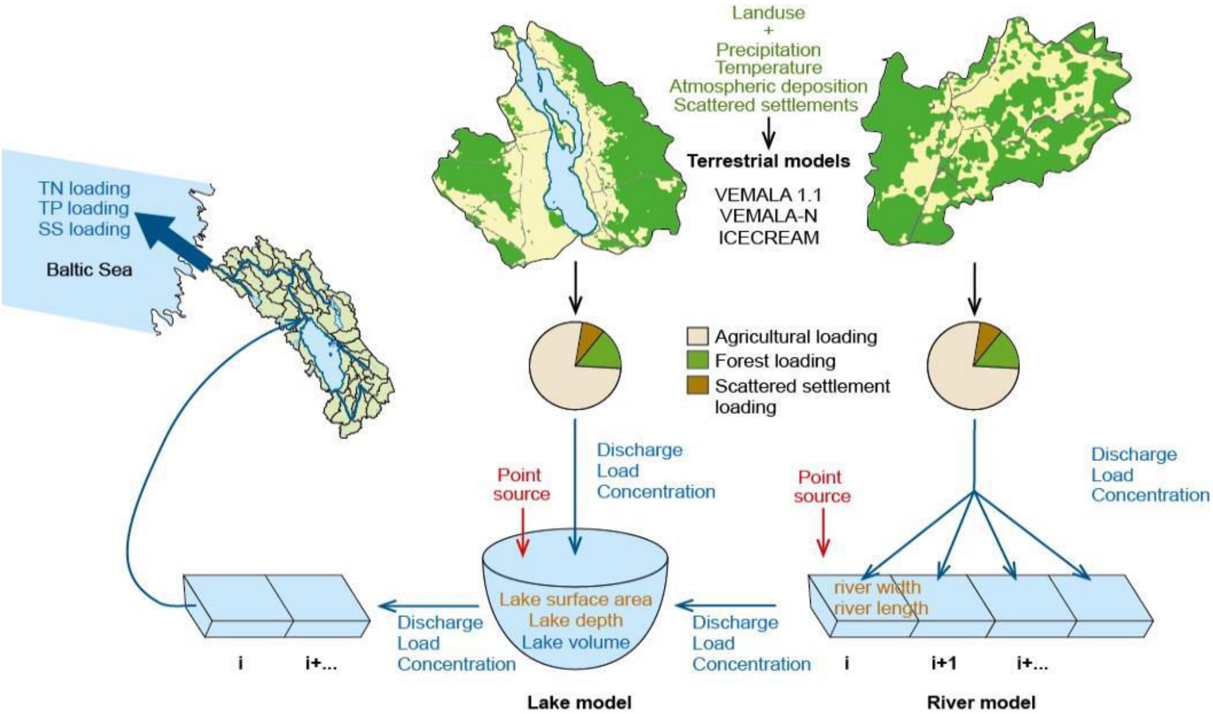


Figure 7: Model framework of VEMALA v.3 (Huttunen et al., 2016))

VEMALA v3. uses meteorological data, hydrological data water quality monitoring data, agricultural field data, and annual point loading information on a national (Finnish) scale. The model simulates hydrology, leaching, transport, nutrient processes, retention and the net load from Finnish water systems to the Baltic sea. Additionally it simulates phytoplankton, TOC, particulate phosphorus (PP), organic nitrogen (Norg), suspended sediments (SS), phosphate (PO_4^{3-}), organic phosphorus (Porg), nitrate (NO_3^-), ammonium (NH_4^+), and oxygen (O_2) daily concentrations for lakes and rivers (Korppoo et al., 2017). So far VEMALA has only been used in Finnish watersheds. The model results have been compared to observations in several Finnish catchments (Huttunen et al., 2016; Korppoo et al., 2017). Korppoo et al. (2017) states that the results of the model for simulating daily nutrient loads were “satisfactory” for the Aurajoki river basin.

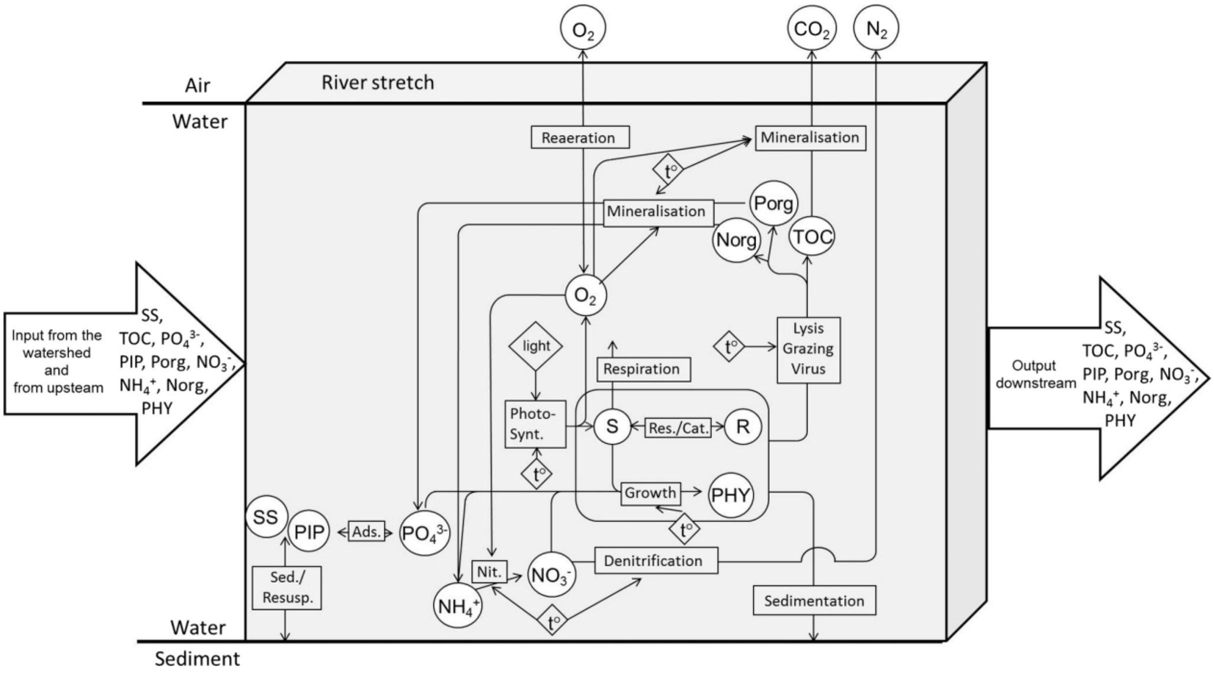


Figure 8: Biogeochemical sub-model included in VEMALA v.3 with Sed. is the sedimentation process, Resusp. is the resuspension process of the bottom sediments, Ads. is the adsorption/desorption process of phosphate onto the sediments, Nit. is the nitrification process, Photosynt. refers to photosynthesis, Res., the building up of reserves to allow growth in the dark and Cat. the catabolism of the reserves (Korppoo et al., 2017).

2.2. METHODOLOGY VALIDATION, SENSITIVITY ANALYSIS, AND ASSESSMENT OF THE CURRENT STATE

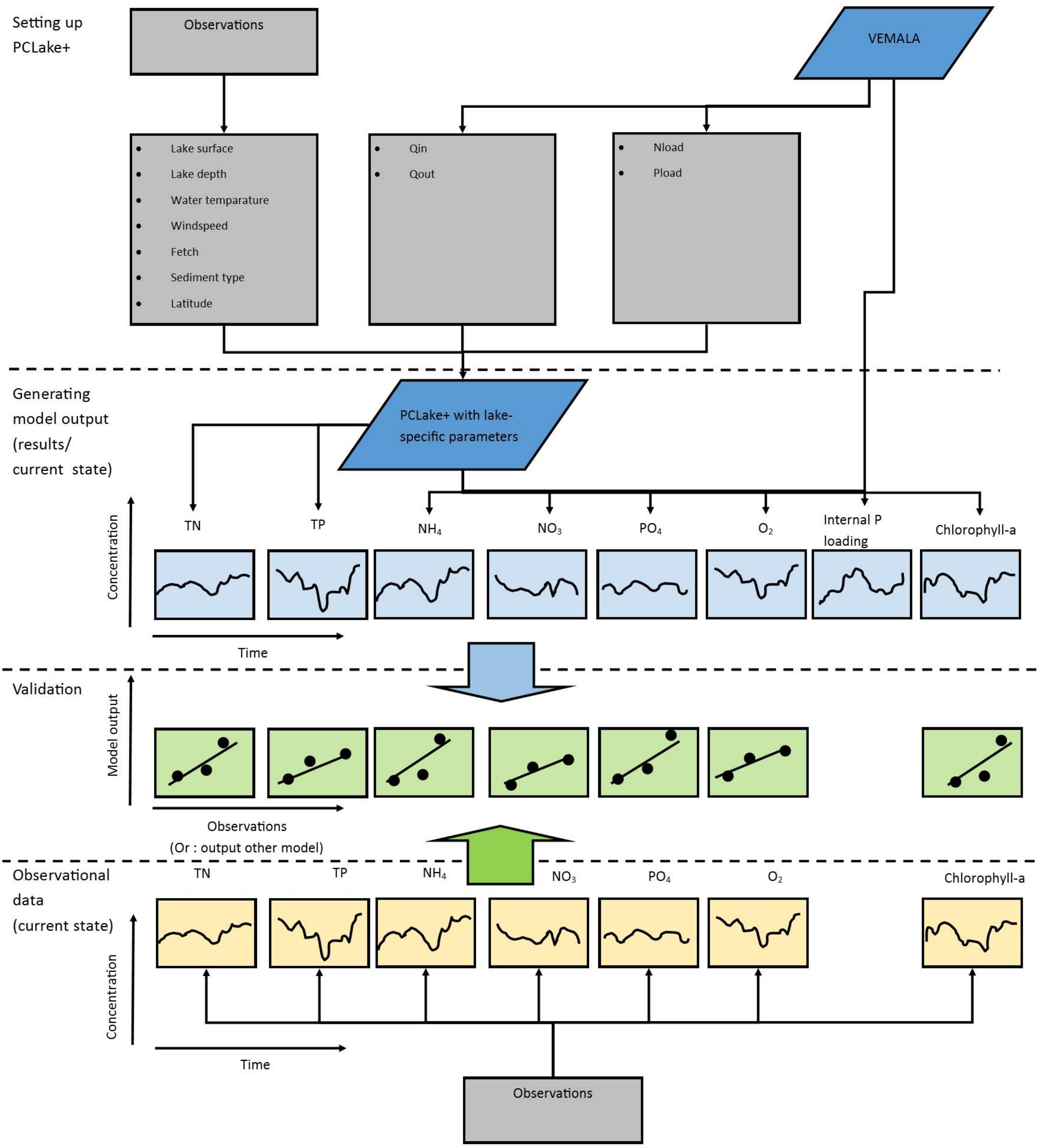


Figure 9: Methodology schematisation of the validation of PCLake+ and VEMALA v3. and the assessment of the historical water quality of lake Enäjärvi.

Figure 9 shows a general visualisation of the steps taken to validate VEMALA V.3 and PCLake+ for lake Enäjärvi and assess the current state of the lake. The first step is to set up the PCLake+ model with lake specific characteristics (e.g. depth and fetch), a water balance, and the external load which is the output of the VEMALA terrestrial model (described in more detail in section 2.2.1.). Once the model is set up it can be run so it generates output for the variables shown in figure 9 and table 4. This, together with the VEMALA V.3 output for these variables and the available observations represent the assessment of the current state of the water quality. Since there were assumptions made for the sediment variables and water depth and additional sensitivity analysis for PCLake+ is done in this step. The third step is to validate both models. In this step, the output of both VEMALA V.3 and PCLake+ is validated against the available observational data and compared to each other, based on which the performance of both models is analysed.

2.2.1. SETTING UP PCLAKE+ FOR THE VALIDATION AND SENSITIVITY ANALYSIS

In this section the preparation of PCLake+ will be discussed, the used data sets and input values used are evaluated. Furthermore, I will zoom in on the sediment, water temperature and water balance and the sensitivity of the model to these input parameters. The here described model preparation is used to validate the model for lake Enäjärvi to assess the current state of the lake. VEMALA v.3 has already produced output for lake Enäjärvi which can be directly used, therefore I will not describe the model setup methodology for VEMALA v.3. For further information on VEMALA v.3, see Korppoo et al. (2017).

INPUT VALUES

Section 2.1.1. describes the general input for PCLake, this section contains the values used in this research for the needed parameters and how they are obtained (see table 2). For the validation analysis time series are used if available. Since PCLake+ does not take into account leap days all leap days are removed from the data series. These input values are also largely used in the bifurcation analysis described in chapter 3.

Table 2: PCLake+ input values for Enäjärvi.

Parameter	Average	Description	Source	Unit conversion
Lake surface	4920000 m^2	Lake surface in m^2	Obtained from SYKE	
sDepthW	3.22 m	Mean depth of the lake	Obtained from SYKE	
mTemp	6.94 C year average 15.95 C summer average	Measured/modelled time series of water temperature	Obtained from SYKE	
mQIn	8.91 $mm*d^{-1}$	Modelled time series of water inflow (river inflow + precipitation)	Obtained from SYKE (precipitation data and river inflow data)	From m^3/d to $mm*d^{-1}$ by dividing by lake surface and transformation from seconds to day and meter to mm)

mQOut	6.74 $mm*d^{-1}$	Modelled time series of water outflow	Obtained from SYKE	Similar unit transformation as for mQIn
mPload	0.00082 $g*m^{-2}*d^{-1}$	Time series of P loading	Obtained from SYKE	From kg/day to $g*m^{-2}*d^{-1}$ by dividing by 1000 and lake surface
mNLoad	0.019 $g*m^{-2}*d^{-1}$	Time series of N loading	Obtained from SYKE	
cLat	60.3472ºN	Latitude lake	Google maps	
cVWind	4.06 $m*s^{-1}$	Average wind speed	Obtained from SYKE	
cFetch	2230 m	Lake fetch	Calculated based on mean wind direction and longest open water distance. Wind direction data is obtained on windfinder.com (suare root of the lake size is a good estimation, this would be 2218m)	
fLutum	0.33 gLu * gDW^{-1}	Lutum content of inorganic matter	Obtained from SYKE	
fFeDIM	0.033 gFE * gDW^{-1}	Fe content of inorganic matter	Obtained from SYKE	
fAlDIM	0.033 gAL * gDW^{-1}	Al content of inorganic matter	Obtained from SYKE	
fDTotS0	0.34 gDW * gDW^{-1}	Initial dry weight fraction in sediment	Obtained from SYKE	
fDOrgS0	0.083 gDW * gDW^{-1}	Initial organic fraction of sediment	Obtained from SYKE	

The sediment parameters (fLutum, fFeDIM, fAlDIM, fDTotS0, fDOrgS0) are dependent on the lake sediment and are based on the table in Janse (2005), which is copied in table 3. The sediment type is, to my knowledge, unavailable for lake Enäjärvi. Therefore the catchment soil type composition is assumed to be representative for the lake. The sediment parameters are ascribed to the model according to soil composition of the catchment and downscaled to the lake. The catchment area consists of 76% of clay, 4% of silt, 18% of “coarse material” here given the characteristics of sand, and 2% of peat. This division is used to determine the sediment soil parameters in lake Enäjärvi (e.g. for 75% the clay characteristics are used, for 4% the silt characteristics, etc.). (For the units see previous table 2)

Table 3: Soil properties adapted from Janse, (2005)

Type	Dry matter	Organic matter	Lutum	Fe	Al
Clay	0.3	0.08	0.4	0.04	0.04
Clay/sand	0.4	0.08	0.215	0.0215	0.0215
Sand	0.5	0.08	0.03	0.003	0.003
Peat	0.1	0.25	0.4	0.04	0.04

PCLake+ has initial values for a turbid and clear state, these variables are determined based on calibration on several Dutch and non-Dutch lakes (Janse et al., 2010). For the validation of the model for lake Enäjärvi, these variables need to be adjusted to better fit the specific lake's initial conditions (Appendix A). This is done by entering the input data as described in table 2. The time series used are entered for one year and consequently repeated for this specific year for the duration of the run. For this validation the model ran for 70 years, meaning that the one-year time series are repeated for 70 years, causing the model to reach an equilibrium. If the output for the same nutrient load but a different initial state significantly differs in equilibrium it is an indication that the lake has two alternative states under the given conditions. To determine in this case whether to use to clear- or turbid output values as input for the validation run depends on the best fit with the data. The generated output (see Appendix A for the list of output variables) is entered the initial state variables used for the actual validation run. In the validation run the time series used are transient, henceforth, not 1 year repeated but the actual multiyear time series are used in the analysis, in this case from 01/01/2000 – 31/12/2017 (step 1 in figure 9).

2.2.2. VALIDATION METHOD

This section contains the validation of both VEMALA v.3 and PCLake+ in lake Enäjärvi to assess their functioning for this specific case. Observation data on water quality and quantity variables are used. This data is obtained by regular sampling of the regional water authority. SYKE provided the observation data and the output of the VEMALA V.3 model. The PCLake+ results are validated with either observation data, VEMALA V.3 output, or both. The VEMALA V.3 results are validated with observation data, PCLake+ results, or both. The observational data and model output is averaged per month to obtain better comparable data sets.

Table 4: Validation variables. The units are in which they are delivered or in which the output is generated

	Observational	VEMALA V.3	PCLake+
NH_4	mg/l	mg/l	$gN*m^{-3}$
NO_3	mg/l	mg/l	$gN*m^{-3}$
PO_4	mg/l	mg/l	$gP*m^{-3}$
O_2	mg/l	mg/l	gO_2*m^{-3}
TN	mg/l	N.A.	$gN*m^{-3}$
TP	mg/l	N.A.	$gP*m^{-3}$
Internal P loading	N.A.	mg/l/d	$gP*m^{-3}*d^{-1}$
Chlorophyll- a	mg/l	mg/l	g chla m^{-3}

To quantify the errors of the models in relation to the observation data and assess the model performance, a set of prediction accuracy measures were executed. For this, I used the R^2 to analyse the similarity in trend between the model output and observational data and how wel the model explains the observational values, and the Mean Relative Absolute Error (RE) comparing a mean error to errors produced by a trivial model (<1 indicates good performance, the higher the value the worse the performance)(Glen, accessed august 2020). The deviation of the regression line from the 1:1 line is also calculated by determining the slope of the regression line which is fitted through the origen. To assess the collinearity between the PCLake+ and VEMALA V.3 output, and the observational data the R^2 is calculated. The R^2 ranges from 0, no relation, to 1, strong positive linearity. Although widely used for the evaluation of models, R^2 is relatively sensitive to outliers and does not perform well for proportional differences between the predictions of the model and the observational data, therefore I will also look at the RE. Here the R^2 is calculated by excel based on the regression line fitted through the origin. The RE indicates the average absolute value of the relative differences between the measured and modelled values. Values close to 0 are considered good. The following formula is used to calculate the RE.

$$\mathrm{RE} = \frac{1}{n}\sum\left[\frac{|d-m|}{d}\right] \qquad \textbf{(Eq.2,)}$$

In equation 2 n is the sample size, d is the observed data, and m is the model output (Janssen et al., 2017).

The regression line is plotted for every validation variable, here the model output is plotted against the observational data (Appendix B). The observational data is projected on the X-axis and the modelled data on the Y-axis. Deviation of the regression line from the 1:1 (which would indicate a perfect model performance) line shows that the model does not give 100% similar results as the observational data. The bigger the difference between the regression line slope and the 1:1 line is, the more the model predicts values deviating to the observational data, either too high or too low. A slope of less than 1 means that the observational values are in general higher, a slope of more than 1 shows the opposite (See Appendix B for the graphs).

2.2.3. SENSITIVITY METHOD

The transient output of PCLake+ may be susceptible to certain input parameters for which I used approximations in the validation run. To quantify the effect of these approximate values I performed a sensitivity analysis for the water depth and sediment soil type. The water depth is set to a constant whereas it in reality slightly fluctuates, the sensitivity analysis is performed to understand the effect a fluctuation in the water depth may have. The sediment data is downscaled from catchment scale to the lake, however, this is no guarantee that the lake consists of the sediment soil type it was given in the validation run (table 2), therefore a sensitivity analysis with respectively clay, peat, and sand sediment input values is done. The sensitivity of the model to both input parameters is tested by calculating the deviation (slope) of the regression line of the basiline (x-axis) vs. the output of the sensitivity/adjusted parameter run (y-axis) to the 1:1 line (which would indicate no difference, thus zero sensitivity, between baseline output and adjusted parameter output). This is calculated by plotting the baseline calculations (x-axis) against the model output with adjusted parameters (y-axis). See Appendix C for the graphs.

INPUT

The sensitivity analysis makes use of the same input values as the validation run, except for the water depth and sediment parameters respectively. The water depth sensitivity analysis runs with a minimum water depth of 2.96 m and a maximum water depth of 3.48 m. These values represent the average of 3.22 m minus and plus two times the standard deviation of the observed value range respectively. The sediment soil type sensitivity analysis makes use of the sediment parameter values as shown in table 3 Janse (2005) for clay, peat, and sand.

The sensitivity analysis is also statistically tested using the R^2 for the baseline output versus the output of the sensitivity run with adjusted parameters, but instead of the RE here the slope of the regression line is used to assess the difference between the model outputs. The regression is fitted through 0. Deviation from the 1:1 line shows that the sensitivity run output differs from the baseline. A slope value smaller than one indicates that the sensitivity run output has lower values then the validation or normal run, values larger than one show the opposite, the value of the slope indicates the percentage the data differs (e.g. a slope of 0.95 indicates that the sensitivity run output is on average 5% lower than the validation run output). The "normal" model output, that is to say, the output from the validation run, is plotted on the X-axis. The output from the sensitivity run is plotted on the Y-axis. If either the R^2 is low or the slope of the regression is substantially different then 1, the model is said to be sensitive to the changed parameter.

2.3. RESULTS

2.3.1. RESULTS VALIDATION

This section contains an overview of PCLake+ validated against observational data, VEMALA V.3 validated against observation data, and both models compared to each other. The transient analysis (analysis over time, non-equilibrium) show the behaviour of the lake in the past (2000-2018) with monthly average values and how both models relate to the observations, the statistical analysis shows how good the models perform statistically (see Appendix B for all linear regression graphs).

The nitrogen compounds used to validate the model in this study are Ammonium (NH_4), Nitrate (NO_3), and the Total Nitrogen (TN) (figure 10). Both PCLake+ and VEMALA V.3 simulate NO_3 and NH_4, however, for TN, although simulated by VEMALA v.3, only results simulated by PCLake+ were available. For NH_4 both models seem to overestimate the concentration. Looking at the spread of the data PCLake+ and VEMALA V.3 have similar mean concentrations of approximately 0.11 – 0.12 mg/l, where the mean of the observations is substantially lower with 0.01 mg/l. The range of the output of PCLake+ is higher than VEMALA V.3, the observational data shows some outliers outside the upper and lower bound. PCLake+ shows a strong variance which is seemingly seasonal, these fluctuations partly agree with the fluctuations as seen in VEMALA V.3 but not with the fluctuations in the observations. Also for NO_3 PCLake+ simulates a bigger range and dominant fluctuations as compared to VEMALA V.3. however, compares fairly well to the observations. Although the low concentrations are overestimated by PCLake+. VEMALA V.3 performs better in the lower concentration range but does not capture the higher concentrations. Here PCLake+ seems to have a better fit than the VEMALA V.3 simulations. The seasonal fluctuations show a general similarity between both VEMALA V.3 and PCLake+ and the observational data. Some of the high peaks in the observational data are not captured by both models. The TN range of PCLake+ and the observations is largely comparable between PCLake+ and the data. On average PCLake+ underestimates the TN concentration, but has a similar range of values. However, on first look, the fluctuations are different, the R^2 however, fitted through the origin, is high with a value of 0.82.

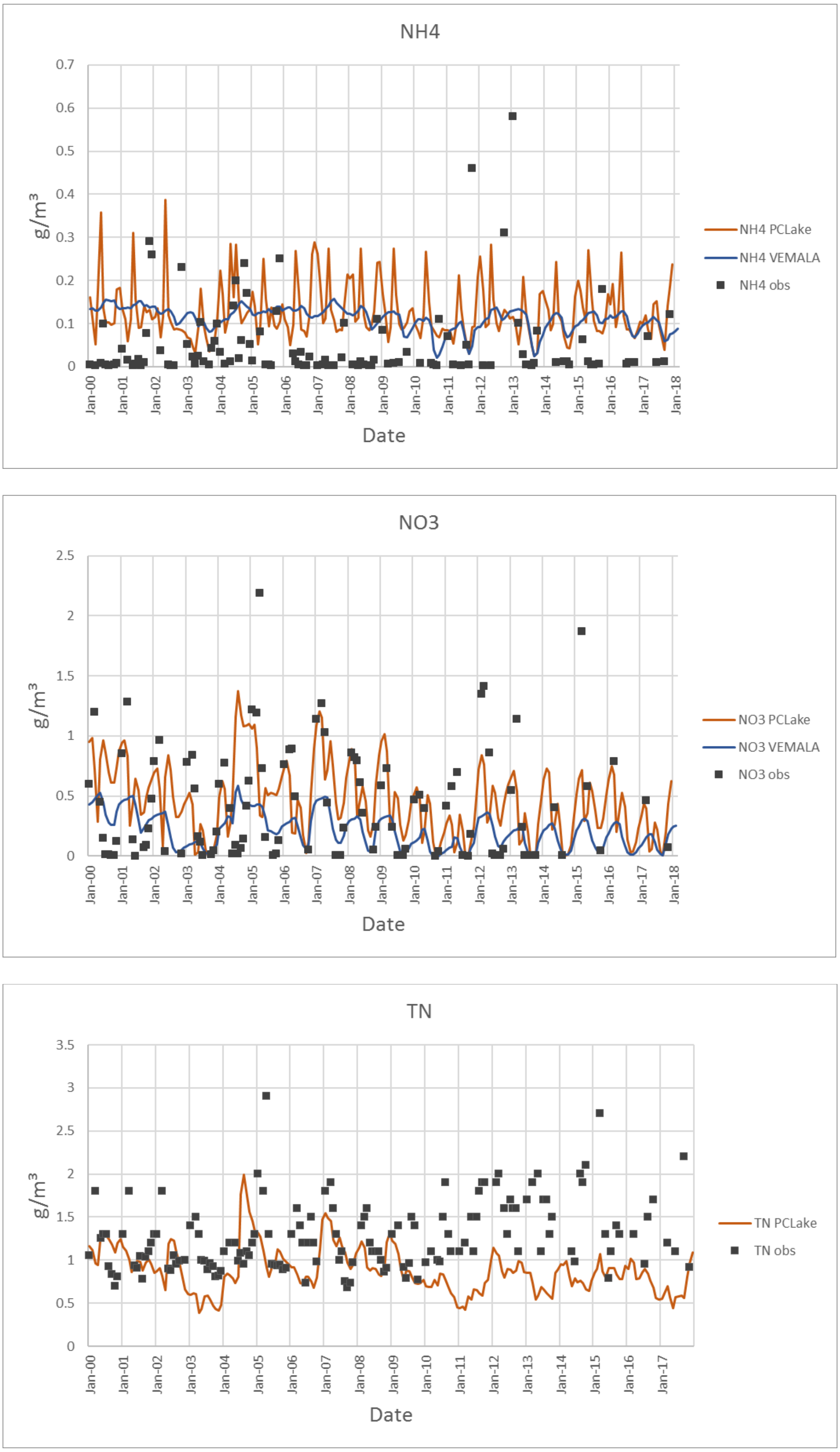
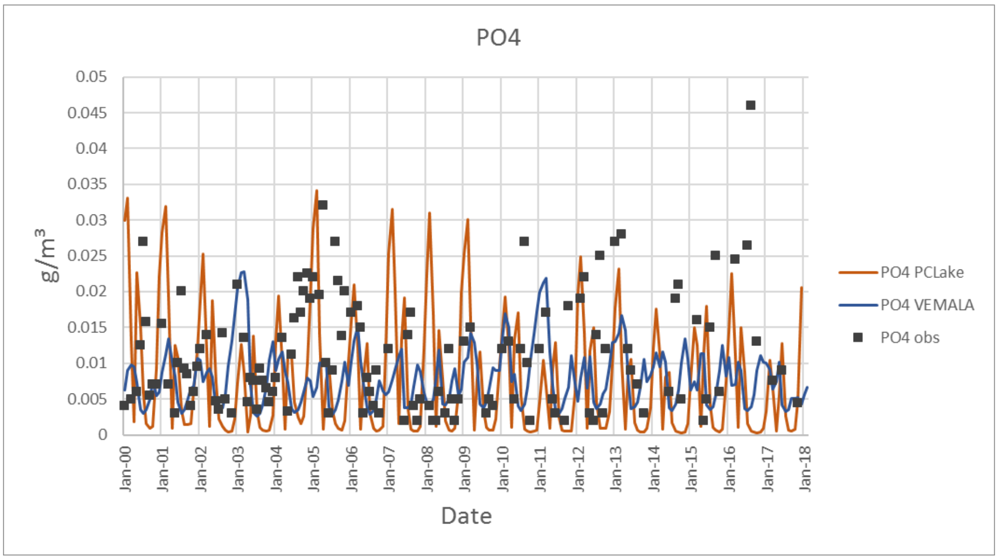
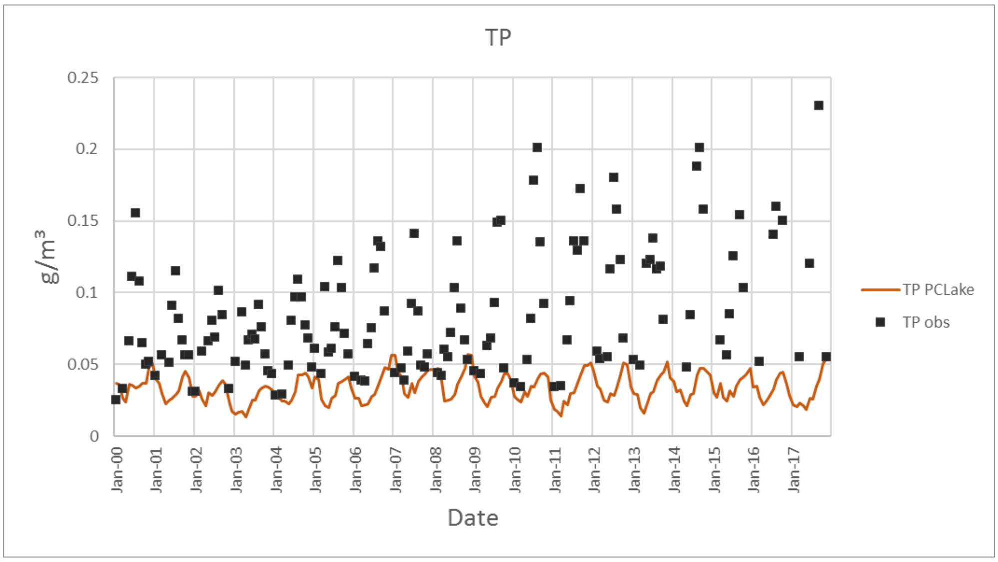


Figure 10: From upper to lower: NH4, NO3 and TN concentrations over the period 2000-2017. Modelled by VEMALA v3. and PCLake+ and observation data

Phosphorus (P)

The P data available consists of Total Phosphorus (TP) and Phosphate (PO_4). Similar to the N components VEMALA V.3 simulates a smaller range as compared to both the observational data and the PCLake+ output for PO_4 as seen in Figure 11. For PO_4 both the models show a similar seasonal variability in concentration, however, some of the high concentration peaks are better captured by PCLake+, where VEMALA V.3 again seems to better simulate the lower ranges. Where it regards the TP (VEMALA v.3 output is missing), PCLake+ underestimates the concentration and does not accurately simulate the seeming trend of increasing concentrations which are shown in the observations (Figure 11). The internal P loading is calculated by both VEMALA V.3 and PCLake+. Both models show a very similar seasonal trend and the ranges are largely similar. The internal P load is not measured since it is to complex. Therefore it is not possible to assess the performance of the models for the internal P load by comparing them to observational data. However, since both simulations are so similar it is plausible to state that the internal P load is accurately predicted by both VEMALA V.3 and PCLake+.





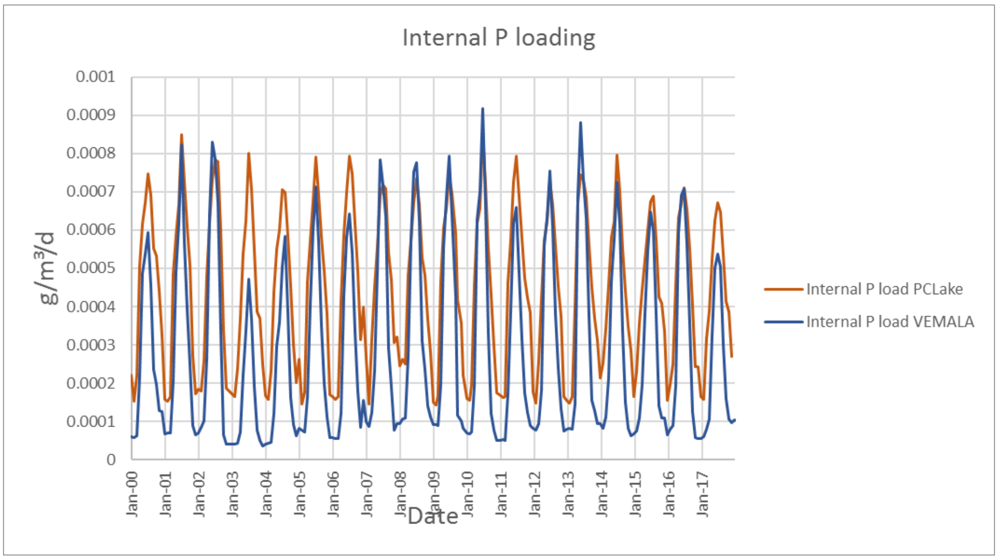


Figure 11: From top to bottom: PO4, TP and internal P loading

Oxygen

As seen in Figure 12 PCLake+ overestimates the oxygen concentration in the water by simulating relatively high peaks. VEMALA V.3, on the other hand, better captures the range found in the observation, both models simulate the seasonal variation adequately, but PCLake+ generates an additional high peak in the spring months, predominantly in April.

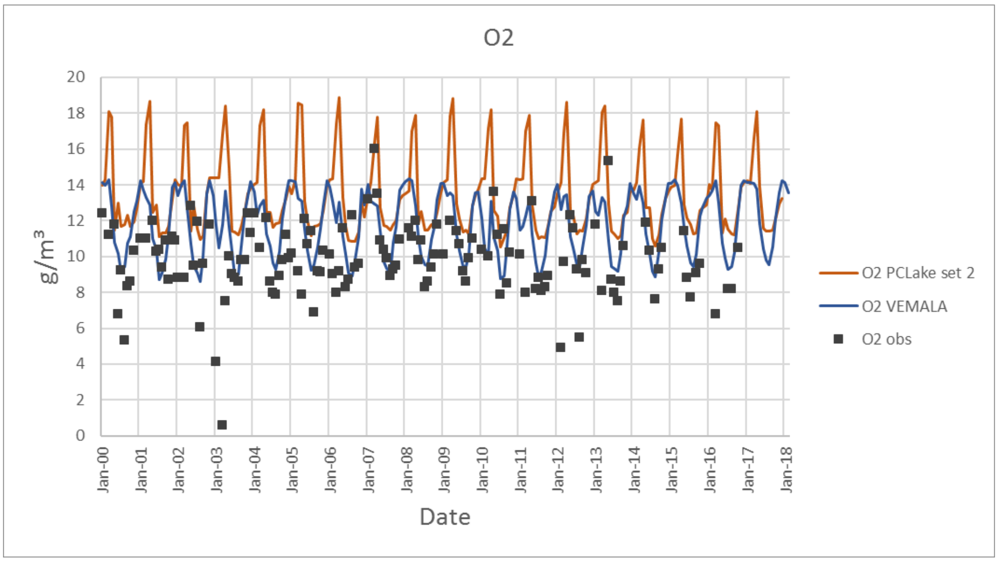


Figure 12: O2 levels in lake Enäjärvi, modelled and observed

Chlorophyll-a

The observational data regarding the chlorophyll-a concentrations show a gradual increase from approximately 2010 onwards (Figure 13). This increase is not captured by either of the models. PCLake+ and VEMALA V.3 are both in a decent range in the first 10 years of the simulation. PCLake+ also simulates a spring bloom consisting mainly of diatoms (see Appendix D.1). This spring bloom can be the result of uncertainty in the silica that is included in PCLake+ as an input factor. There is not silica data available for lake Enäjärvi and therefore the default silica concentration of PCLake+ cannot be validated and might be overestimated. With lower silica concentrations in PCLake+ the spring peak decreases and the increasing trend of the chlorophyll-a concentration is partly followed, however, still not to the extent as shown in the observation (see the discussion of this chapter and Appendix D.4). The PCLake+ values plotted against the observational data gives a slope of 0.47, for VEMALA v.3 this is 0.58, indicating substantially lower values for both models compared to the observational data (Appendix B.1 and 2)

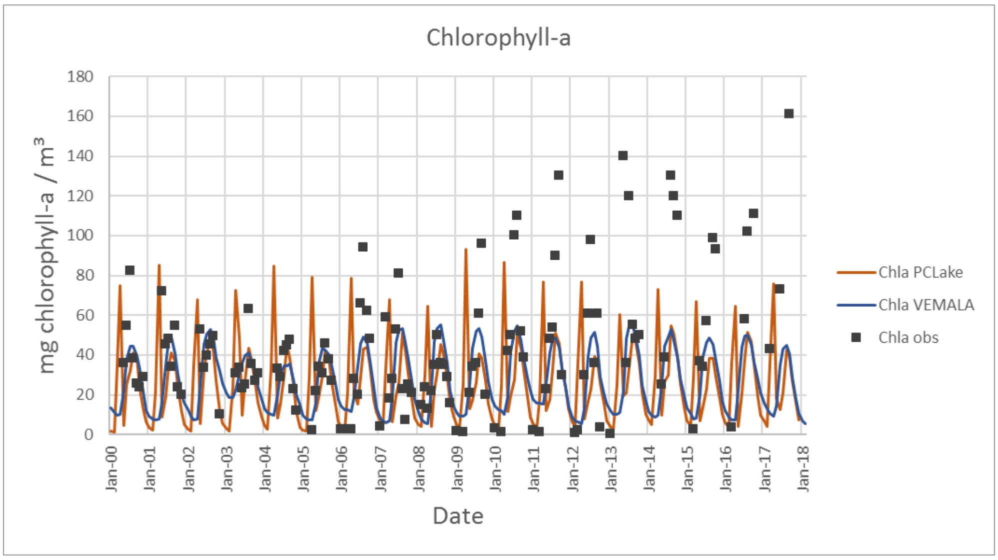


Figure 13: Chlorophyll-a concentration over time

Model performance

The model performs reasonably well when looking at the trends, the RE and the R^2. The R^2 of PCLake+ to the observation based on a trendline fitted through the origin is in the range of 0.17 – 0.94 with the lowest value for NH_4 and the highest for O_2. The RE is in a range indicating a good model performance (0.35 -1.85) except for the outlier of 17.7 and 19.6 for NH_4 and NO_3 respectively. The deviation of the slope of the regression line fitted through the origin compared to the 1:1 line is in the range between 28% and 68% (either positive or negative deviations) with the lowest deviation for O_2 and the highest for TP. VEMALA V.3 produces R^2 values in the range of 0.22 – 0.95 with also the lowest fit for NH_4 and the highest for O_2. The RE is in the range between 0.035 and 1.14 similar as PCLake+ with again outliers for NH_4 (16.4) and NO_3 (7.41). The deviation of the regression line slope to the 1:1 line ranges from 13% to 65% (either a positive or negative deviation) with the lowest again for O_2 and the highest now for NO_3.

VEMALA v.3 seems to perform over all slightly better then PCLake+. However, in general both models show reasonable performance for the RE and by purely looking at the variable output concentrations in the graphs. The deviations of the regression line slopes to the 1:1 lines indicates that the models do not deviate extremely from the observational data, however, do show that they do not perform optimally. R^2 indicate a good fit exept for NH_4.

Table 5: deviation in % of the regression line slope to the 1:1 line (perfect model performance).

Variable	PCLake+ (%)	VEMALA v.3 (%)
NH_4	-44	-47
NO_3	-31	-65
PO_4	-57	-55
O_2	+28	+13
TP	-68	n.a.
TN	-36	n.a.
Chlorophyll-a	-53	-42

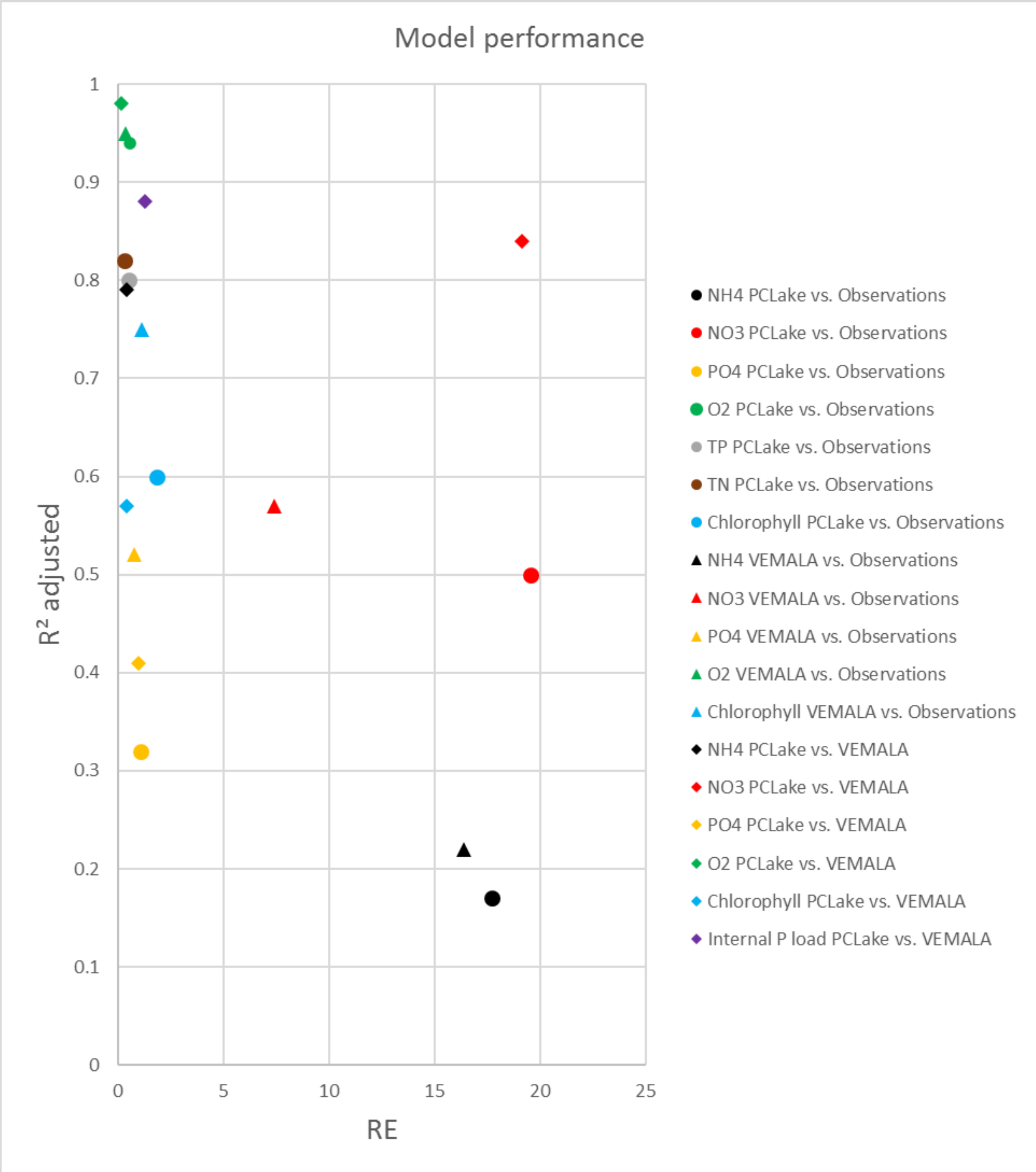


Figure 14: Performance indication for the validation parameters of VEMALA v3. and PCLake+ for lake Enäjärvi. An optimal performing model would give values of 0 RE and R^2 of 1.

2.3.2. RESULTS SENSITIVITY ANALYSIS

2.3.2.1. WATER DEPTH

The PCLake+ model is run with static water depth, however, the water depth fluctuates. To assess whether fluctuating water depth substantially influences the concentration of the output variables I compared the baseline output to the output calculated with a smaller and larger water depth.

LOW WATER TABLE

A lowering of the water table from 3.22 m to 2.96 m has a limited effect on the output variables. However the internal load and chlorophyll-a concentrations are slightly higher (deviation = 9% and 8% respectively), the PO_4 concentration results in a 6% lower slope. The differences are substantially small, henceforth, the model is not too sensitive to the assumed static water table.

Table 6: Deviation in % of the regression line slope to the 1:1 line (no difference between baseline output and adjusted parameter output) for a lower water table (3.22 m VS. 2.96 m).

Variable	Deviation of the regression line slope (%)
Internal load	+9
NH_4	+3
NO_3	-2
PO_4	-6
O_2	0
TP	+5
TN	+1
Chlorophyll-a	+8

High water table

A higher water table, contrary to the lower water table, causes lower concentrations of the internal load and chlorophyll-a with a slope of 8% and 7% lower than the 1:1 line respectively. Increasing the water table from 3.22 m to 3.48 m, causes the PO_4 concentrations to slightly increase. Also here the differences are small, henceforth the model is not sensitive to the implemented change in water depth.

Table 7: Deviation in % of the regression line slope to the 1:1 line (no difference between baseline output and adjusted parameter output) for a higher water table (3.22 m VS. 3.48 m)

Variable	Deviation of the regression line slope (%)
Internal load	-8
NH_4	-3
NO_3	+4
PO_4	+8
O_2	+0
TP	-5
TN	0
Chlorophyll-a	-7

2.3.2.2. SEDIMENT SOIL TYPE

The sediment soil type of lake Enäjärvi is to my knowledge not fully known, therefor the sediment related parameters lutum, iron, aluminium, organic matter, and dry matter are estimated based on the soil types in the whole catchment. However, there is a possibility that this estimation gives an inaccurate representation of the actual sediment type. The sedimentary parameters influence for example the amount of nutrients rereleased from or buried in the sediment. To analyse how sensitive PCLake+ is to the sedimentary input a sensitivity analysis is performed. To this end an additional transient analysis is done in which all input variables and parameters remain similar as in the validation step except the sediment parameters, these will be adjusted according to the table of Janse (2005) (see table 3) for clay, sand, and peat since the first two are the most widely present soil types and the latter differentiates the most in case of organic matter, possible strongly influencing P concentrations in the water column.

To analyse the sensitivity of the model to the sediment parameters, the output based on the sediment data used in the baseline validation (section 2.3.1.) is plotted against to the output of the other sediment soil types. Based on this the R^2 and the percentual difference between the data can be determined (based on the slope).

Clay

PCLake+ does for most output variables not seem sensitive to a change from the sediment values used in the validation to the clay related values. The clay soil has the most influence on the PO_4 values. The slope indicates that on average the PO_4 values of the clay sediment type are on average higher than the baseline.

Table 8: Deviation in % of the regression line slope to the 1:1 line (no difference between baseline output and adjusted parameter output)for a clay soil type

Variable	Deviation of the regression line slope (%)
Internal load	0
NH_4	+1
NO_3	+7
PO_4	+14
O_2	-1
TP	-2
TN	+2
Chlorophyll-a	-6

Sand

Sand soil input has a larger influence then clay on first sight with a deviating slope of the regression line of 29%, 19% and 15% for the internal load, TP, and chlorophyll-a and PO_4. However, sand is not common in southern Finland (Personal communication) and does not produce better fits with the observational data).

Table 9: Deviation in % of the regression line slope to the 1:1 line (no difference between baseline output and adjusted parameter output) for a sandy soil type

Variable	Deviation of the regression line slope (%)
Internal load	+29
NH_4	0
NO_3	-5
PO_4	-15
O_2	0
TP	+19
TN	+4
Chlorophyll-a	+15

Peat

Peat mostly influences the PO_4 concentrations with a slope of 28% higher than the 1:1 line. However, also here, the values of the other variables are substantially low to state that the model is not sensitive to sediment characteristics given in the validation run.

Table 10: Deviation in % of the regression line slope to the 1:1 line (no difference between baseline output and adjusted parameter output) for a peat soil type

Variable	Deviation of the regression line slope (%)
Internal load	+2
NH_4	+1
NO_3	+11
PO_4	+28
O_2	0
TP	+1
TN	+3
Chlorophyll-a	-9

2.4. DISCUSSION

In this chapter I addressed the sub-question ”What is the current state of the water quality of lake Enäjärvi and how do PCLake+ and VEMALA v.3 perform in relation to observational data?

The results yield some additional interesting questions. The oxygen output of PCLake+ showed some high peaks. These oxygen peaks can be explained by the spring bloom of diatomic algae simulated by PCLake+.These diatom peaks might have resulted in higher oxygen production. Another explanation of the oxygen concentration peaksis the reaeration simulated by the PCLake+. Daily wind data is used as input, but the reaeration rate is based on yearly averages, perhaps leading to a day to day overestimation. However, these assumptions are not tested in this study and require further research. One of the main noticeable differences between the observations and the model results is the incapability of both VEMALA V.3 and PCLake+ to capture the increasing trend of the TP and chlorophyll-a concentration (Figures 11 and 13). Over the first part, roughly half of the validation period, both models are well in the range of the observed concentration. But around 2009-2010 observational data shows an increase of especially summer chlorophyll-a concentrations which are not simulated. This mismatch between models and observation can have several causes. Firstly it might relate to the external loading used as input, if there was a similar increase in external loading which was not simulated by VEMALA V.3, both models might have missed the necessary input to simulate the chlorophyll-a increase. A wrong external loading input would not affect the load-response curve since PCLake+ standardly simulates this over a range of nutrient loadings. Another possibility is that uncaptured internal processes lead to this increasing trend, however, if we look at the internal loading this is highly similar among the models, suggesting that they performed well on the internal loading. But if both models are incapable of capturing certain similar processes this could also explain the similarity. A third option, regarding the chlorophyll-a output of PCLake+, might be that the silica concentration (default 3 gSi/m^3) might be too high, leading to an unrealistic bloom of diatoms in spring and limiting the blue algae growth in summer (Appendix D). To that end, a small additional run with lower silica values was done (1.5 gSi/m^3 and 0.5 gSi/m^3 against a default concentration of 3 gSi/m^3). By looking at the division of the chlorophyll-a concentration by algae species as simulated by PCLake+ it becomes apparent that the blue algae do show an increasing trend, which is however not shown in the total chlorophyll-a concentration due to the relative dominance of diatoms. By limiting the silica concentration in the inflow the total chlorophyll-a concentration does show an increasing trend and the spring peak is reduced since blue algae start to dominate (Appendix D.4). The increase, however, is still not as steep as shown in the observations. A different silica input might be a plausible explanation for the inability to show the increasing chlorophyll-a concentration trend, but silica is not measured in the observations and can therefore not be validated. Additionally, the spring observations which might prove the existence or non-existence of a spring bloom are lacking, therefore I will continue with the default silica input with the notion that the results might differ. I also, therefore, recommend the sampling of chlorophyll-a in spring to determine the presence of a spring bloom and thereby adjust, or not adjust, PCLake+ to better resemble the observations.

Both models simulated concentrations in similar ranges as the observational data. R^2 and Mean Relative Absolute Error (RE) values fell within a range indicating a good model performance except for NH_4 and NO_3 (R^2: 0.17-0.22 and 0.5-0.57 and RE: 16.41-17.73 and 7.42-19.57 respectively for NH_4 and NO_3). For most variables, the models agree fairly well with each other. However, NO_3 results in an inter-model RE of 19.45. The deviations from the regression line slopes to the 1:1 lines indicates that the models do not deviate extremely from the observational data, however, do show that they do not perform optimally. In general PCLake+ captures the higher values better then VEMALA V.3, in the lower ranges VEMALA V.3 performs better compared to PCLake+. This gives rise to the thought that an ensemble of both models might give a good indication of the water quality parameters, further research on that account can be beneficial for the modelling results.

A sensitivity analysis was performed to analyse the sensitivity of PCLake+ towards sediment type and water depth. The results did not yield a substantial difference between the baseline and the adjusted runs.

This chapter also brought along some uncertainties regarding the used methodology and the used models. Firstly, stratification was not included in the modelling exercises done in PCLake+. PCLake+ can include stratification if data on the period and depth is available. However, observational data of this was not available. The online tool Flake can be used to estimate possible stratification periods and depths (Kirillin et al., 2011). This was done but including this did not give substantial differences in output. Provided the uncertainties surrounding Flake and the limited differences in output I choose not to include stratification. If however measured data on stratification was available the model performance could have benefited. Another methodological uncertainty is the use of VEMALA output as input to PCLake+. The external nutrient loading data is calculated by VEMALA since there is no measured data available of the external nutrient load due to a diffuse inflow (multiple streams). Henceforth, the external nutrient load is not validated and is therefore a source of uncertainty in the output of PCLake+ because the external load is one of the main input parameters. Flaws in this data may work its way up to the model output of both models and could possibly explain the inability of PCLake+ and VEMALA v.3 to simulate the increasing chlorophyll-a concentration and explain to some degree the low fit of the model for some parameters such as TP.

2.5. CONCLUSION

In this chapter it was the aim to answer the question "What is the current state of the water quality of lake Enäjärvi and how do PCLake+ and VEMALA V.3 perform in relation to observational data?". The state of the lake is defined as poor according to the Finnish standards. Most eutrophication parameters do not show an increase meaning that the lake does not seem to deteriorate further, however, TP and Chlorophyll-a show in the observations an increase in concentration, both of which are not captured by the models.

Both models perform reasonably well with R^2 and Mean Relative Absolute Error (RE) values falling within a range indicating a good model performance except for NH_4 and NO_3 (R^2: 0.17-0.22 and 0.5-0.57 and RE: 16.41-17.73 and 7.42-19.57 respectively for NH_4 and NO_3). The model output, by purely looking at the graphs, gives similar ranges as the observational data and (seasonal) trends are followed well. The deviation of the regression line slope compared to the 1:1 line ranges between 68% and 13%, implying that the model ouput does not extremely over or underestimate the concentrations of the modelled variables, however, the fit is not high. The most prominent uncertainty is the uncaptured increase of TP and Chlorophyll-a concentrations as indicated in the discussion. With this uncertainty in mind, I deem the models suitable to model the lake. The discussion of chapter 3 will go further into detail about the implication of this uncertainty.

PCLake+ is not significantly sensitive to the water depth, henceforth a static depth of 3.22 m can be used. Only a sandy sediment type yields slight differences, however, since sand is not a common soil type in southern Finland (Personal communication) this differences will be discarded and the baseline sediment type will be used in the further analysis. Furthermore, the different sediment types and water depths do not show a better fit to the observations compared to the baseline parameters.

3. DETERMINING THE CRITICAL NUTRIENT LOAD

In this chapter, I will answer the sub-question “what is the critical nutrient load of lake Enäjärvi?”. The critical nutrient load can be seen as a threshold nutrient loading above which the amount of chlorophyll-a is considered undesired. Determining the critical nutrient loading is beneficial for assessing if and which water management strategies are needed to restore or maintain lake water quality. The determination of the critical nutrient load will be done by performing a bifurcation analysis with PCLake+. This chapter consists of a methodological section on the model preparation and how the critical nutrient load will be determined, a methodology for the sensitivity analysis, the results for both the baseline critical nutrient load and the sensitivity analysis, the position of the critical load in regards to the external loading and whether water quality measures are needed, and a discussion and conclusion.

3.1. METHODOLOGY

In this section, the methods used to determine the critical nutrient load of lake Enäjärvi are described. I used PCLake+ to determine the critical nutrient load. I will provide details on the PCLake+ model preparation used to find the critical nutrient load and to calculate thereafter the load-response or bifurcation graph that is needed to determine the critical nutrient load.

3.1.1. MODEL

To determine the load-response curve of lake Enäjärvi a bifurcation analysis is performed using PCLake+. The bifurcation analysis in PCLake+ is an analysis in which the model calculates the chlorophyll-a concentrations in response to the external nutrient loading. PCLake+ determines the load-response curve in four general steps (see Figure 15).

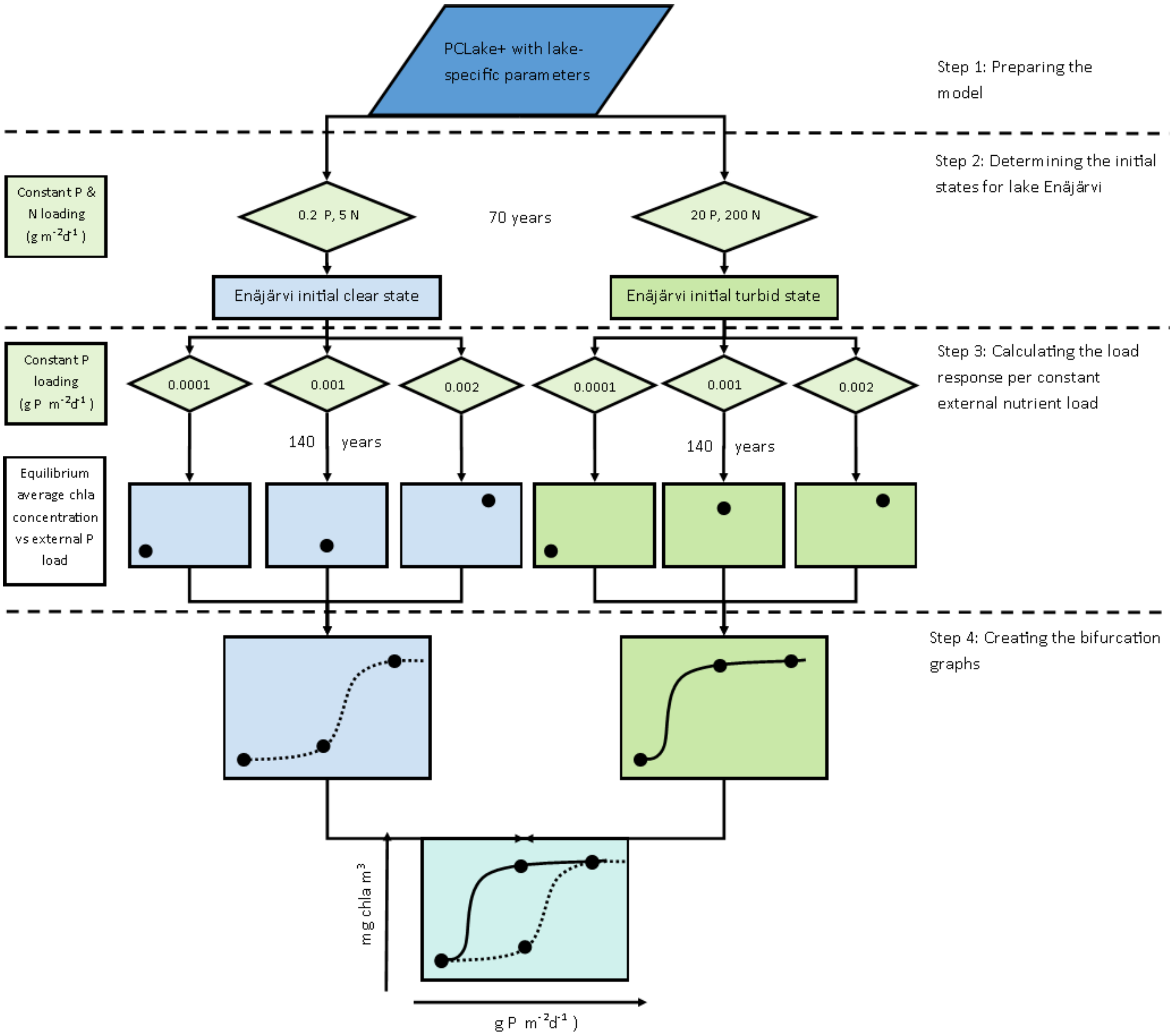


Figure 15: The bifurcation analysis process

The first step (figure 15) is preparing the model: in this step PCLake+ is set up to represent the lake of interest by filling in the known data and lake characteristics. This preparation is similar as described in section 2.2.1. The second step is the determination of the initial states of lake Enäjärvi: PCLake+ has a default turbid and clear state based on a calibration of several Dutch and non-Dutch lakes, however, the initial turbid- and clear state values of lake Enäjärvi are likely to be different (Janse, 2005). To mimic the turbid and clear states of lake Enäjärvi, PCLake+ has to run to equilibrium values with a high nutrient load (20 mg P $m^{-2}d^{-1}$ and 200 mg N $m^{-2}d^{-1}$) and a low nutrient load (0.5 mg P $m^{-2}d^{-1}$ and 5 mg N $m^{-2}d^{-1}$) respectively. These initial states are needed to simulate the load-response curve. The initial states to be used in the bifurcation analysis have to be equilibrium output to diminish the impact the calibrated original initial state values would have on the load-response curve of a specific lake. The third step is to calculate the load response: Once the initial turbid and clear states of lake Enäjärvi are determined, the bifurcation analysis can start. In this step, PCLake+ calculates the chlorophyll-a concentration in response to the external nutrient load. The chlorophyll-a values are expressed as summer average values (day 150-210) in mg/m^3. The load-response can only be calculated for one constant external nutrient load per time, so for one constant nutrient load PCLake+ calculates the chlorophyll-a response. The chlorophyll-a response is calculated in equilibrium to rule out influences of fluctuations in the input data sets. Henceforth, the time series used have to be one year repeated time series a, here I again used the values of the year 2000. This means that the load-response curve will be representative for the climatic conditions of 2000. With the constant external nutrient loads, the model seems to take longer to reach equilibrium as compared to step 2, for the bifurcation analysis the model is run for 115 years. To understand the behaviour of the lake of interest under a broader range of external nutrient loads, PCLake+ calculates the load-response for multiple constant external nutrient loads within a defined range. The range used in this analysis is 0.0001 – 0.001 mg P $m^{-2}d^{-1}$. To give an example: PCLake+ runs 115 years with a constant nutrient load of e.g 0.0001 mg P $m^{-2}d^{-1}$ and calculates the last year summer average chlorophyll-a concentration, this concentration is the chlorophyll-a response to the set nutrient load. Hereafter, the model runs 115 years with a constant nutrient load of 0.000145 mg P $m^{-2}d^{-1}$ and calculates the last year summer average chlorophyll-a concentration. PCLake+ will do this for 21 external nutrient loads within the defined range with turbid initial state values and the same 21 external nutrient loads with clear initial state values. This results in 21 points depicting the chlorophyll-a response to external nutrient loading for a turbid initial state, and 21 points depicting the chlorophyll-a response to external nutrient loading for a clear initial state. The N/P ratio used is 32 following the average N/P ratio found in the observational data. The fourth and last step is creating the bifurcation graphs by merging the obtained information of the previous step into one load-response graph. The separate chlorophyll-a responses to the constant external nutrient loads are plotted in one graph for a turbid initial state and a clear initial state, resulting in a graph with on the X-axis the external P load range in mg P $m^{-2}d^{-1}$, and on the Y-axis the chlorophyll-a concentrations (mg/m^3). The two obtained graphs together form the bifurcation graphs, in the example case (Figure 15) there is hysteresis present, if there is no hysteresis the clear and turbid initial states will lead to the same graph.

Based on the generated load-response curve the critical nutrient loading can be determined. In case of a gradual change in the load-response curve, a human-defined maximum allowable chlorophyll-a concentration is needed to establish the critical nutrient load. For this I use the maximum allowable chlorophyll-a concentration based on Janssen et al., (2017). Janssen et al. (2017) use a range from 20 to 50 mg chlorophyll-a per m^3. I will use the same range in steps of 10 mg chlorophyll-a per m^3.

3.1.2. SENSITIVITY ANALYSIS METHODOLOGY

PCLake+ may be sensitive to certain input variables of which I have insufficient data and made assumptions, the made assumptions may lead to substantial differences in output. To assess PCLake+ sensitivity to the made assumptions I performed a sensitivity analysis for the sediment type and water level. The sediment data available is estimated based on the sediment in the catchment rather than based on in-lake measurements. This might cause uncertainty in the output. Furthermore, the model runs in this study with a constant water depth due to low variations in the water depth. Since both parameters can influence the load-response curve, and thereby the critical nutrient load, a sensitivity analysis in the form of a bifurcation analysis is performed for both the sediment input and the water depth fluctuations. The other input variables stay the same as in the "original" (further called baseline) bifurcation analysis.

Fluctuations in water depth are limited, therefore a static water depth of 3.22 m is used. These water depth fluctuations, however limited, may still affect the output values. To quantify this effect and its significance a sensitivity analysis is performed to assess the effect of the minor fluctuations on the load-response curve and critical nutrient load. To analyse the effects of the water table fluctuations a bifurcation analysis with both a larger and smaller water depth is performed. The adjusted water depths are the average depth plus two times the standard deviation (3.22 m + 0.26 m = 3.48 m) and minus two times the standard deviation (3.22 m – 0.26 m =2.96 m) respectively. The other input values stay the same and are shown in chapter 2.

Since the sediment data is not available for the lake, the soil type in the lake is determined based on the catchment characteristics (See chapter 2). Therefore, there is uncertainty in the used values. Sediment affects the P fluxes in the water and may influence the initial variables obtained from the 70-year run and how the lake responds to the external load. To quantify these effects a sensitivity analysis regarding the sediment input parameters is performed.

To evaluate the sensitivity of the bifurcation analysis in PCLake+ to changes in the water depth and sediment type the deviation of the regression line fitted through the origin to the 1:1 line are calculated A deviation of the slope of the regression line fitted through the origin compared to the 1:1 line quantifies whether the adjusted parameters result in different values compared to the baseline load-response curve. The closer the deviation is to 0, the less sensitive the model is.

3.1.3. COMPARING THE CRITICAL NUTRIENT LOAD TO THE EXTERNAL LOAD METHODOLOGY

Assessing whether the critical nutrient load is exceeded and with what frequency is done by comparing the critical nutrient load against the external nutrient load The external nutrient load is compared to the calculated critical nutrient load for the given time period (summer: day 150-210, years 2000-2017). This can be plotted in a graph similar to figure 16. Since the load-response curves did not show hysteresis an exceedance of the critical load is countered by diving under the critical nutrient load again. The external load is averaged over the same period for which the critical nutrient load is determined, meaning that per year the average external nutrient load of the days 150 -210 is compared to the critical nutrient load and looked if this is exceeded, and how often.

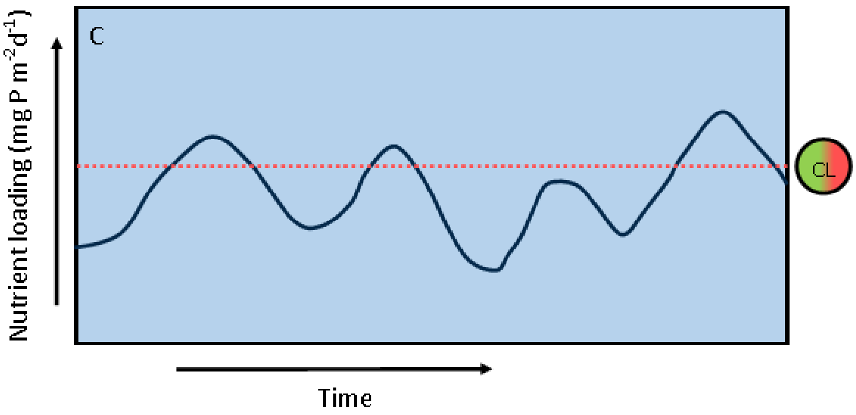


Figure 16: A schematisation of an external nutrient load in relation to a hypothetical critical nutrient load

3.2. RESULTS

The result of the bifurcation analysis in PCLake+ is the load-response curve of the lake. For each of the external P loadings in the used range, the equilibrium summer and spring average chlorophyll-a values are calculated and plotted in Figure 17 and 18 respectively.

The summer algal bloom consists of diatoms and blue algae (see Appendix D.1). The summer load-response curves for the clear and turbid initial states are similar for the full range of external P loading this implies that under this range of external P loading and with the present characteristics of the lake there are no alternative stable states. The summer load-response curve in lake Enäjärvi is nonlinear without substantial hysteresis. Over the lower external nutrient loadings, from 0.1 to 0.37 mgP $m^{-2}d^{-1}$, the chlorophyll-a concentration increases rapidly until a peak of 30 mg Chlorophyll-a per m³ for both the turbid and clear initial state. A further increase in external nutrient loading does not cause a further increase in the chlorophyll-a concentration and after a slight decrease, the chlorophyll-a concentration stabilises. This slight decrease might be explained by a light limitation, however, as Appendix E show the Secchi depth does not increase around this point, therefor further research on to explain this decrease might be needed.

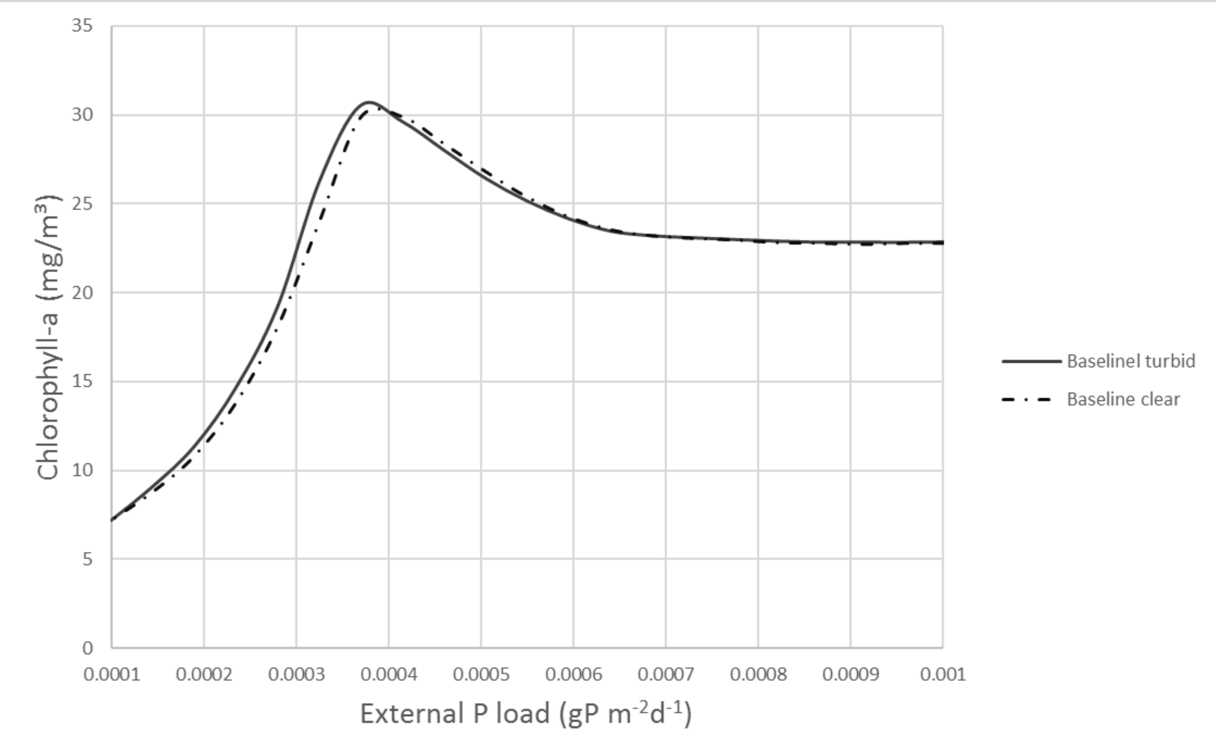


Figure 17: The baseline summer load-response curve of lake Enäjärvi.

I use a range of plausible human-defined maximum allowable (summer) average chlorophyll-a concentrations to calculate the critical nutrient load. Based on Janssen et al. (2017) I use a range varying from 20 mg chlorophyll-a per m³ to 50 mg chlorophyll-a per m³, of which the higher range will only be met during the spring bloom. Under summer conditions the critical nutrient load for a maximum allowable chlorophyll concentration of 20 mg chlorophyll-a per m³ is 0.29 mgP $m^{-2}d^{-1}$ for both initial states. This value implies that an external nutrient loading of this value causes the lake to surpass the human-defined chlorophyll-a concentration threshold of 20 mg/m³. For a threshold of 30 mg chlorophyll-a per m³ the critical nutrient loading is 0.37 mgP $m^{-2}d^{-1}$.

The spring bloom results in a significantly higher chlorophyll-a values in the load-response curve. The spring bloom predominantly consists of diatoms. The curve is nonlinear and shows no sign of hysteresis. The load-response curve of the spring period (day 65 – 125) has a maximum value around 70 mg chlorophyll-a per m³ for both states. Contrary to the summer load-response curve, the spring curve does not seem to decrease over the higher external P loading range but stabilizes for both the turbid and clear state around a chlorophyll-a concentration of 69-70 mg chlorophyll-a per m³ perhaps due to an overestimation of silica input, as mentioned in section 2.4, a lowering of the silica input causes a reduction in the spring bloom. However, due to a lack of silica data and chlorophyll measurements in spring, this cannot be proven.

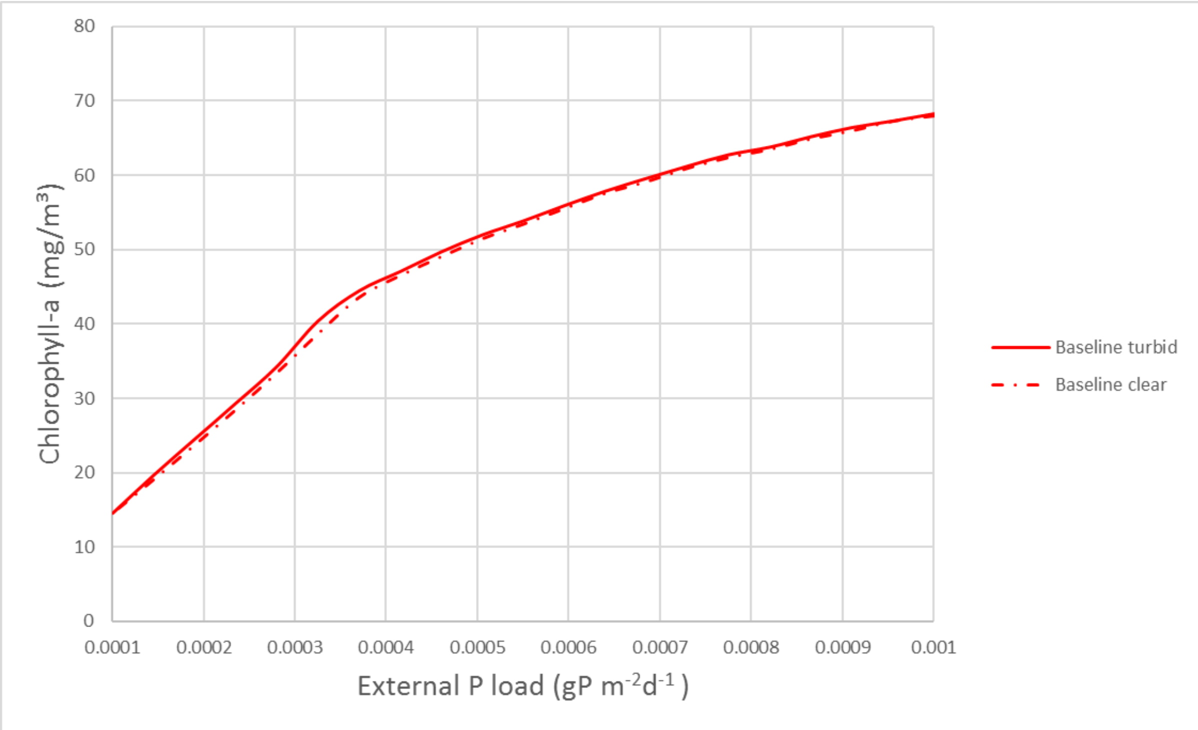


Figure 18: The baseline spring load-response curve for lake Enäjärvi

Table 11: The critical nutrient loadings for a clear and turbid initial state for the thresholds of 20, 30, 40, and 50 mg Chlorophyll-a per m³

Critical loading for a 20 mg chlorophyll-a per m³ threshold		
Time period	Critical nutrient loading clear initial state (mgP $m^{-2}d^{-1}$)	Critical nutrient loading turbid initial state (mgP $m^{-2}d^{-1}$)
Summer	0.29	0.29
Spring	0.15	0.15
Critical loading for a 30 mg chlorophyll-a per m³ threshold		
Time period	Critical nutrient loading clear initial state (mgP $m^{-2}d^{-1}$)	Critical nutrient loading turbid initial state (mgP $m^{-2}d^{-1}$)
Summer	0.37	0.37
Spring	0.25	0.24
Critical loading for a 40 mg chlorophyll-a per m³ threshold		
Time period	Critical nutrient loading clear initial state (mgP $m^{-2}d^{-1}$)	Critical nutrient loading turbid initial state (mgP $m^{-2}d^{-1}$)
Spring	0.35	0.32
Critical loading for a 50 mg chlorophyll-a per m³ threshold		
Time period	Critical nutrient loading clear initial state (mgP $m^{-2}d^{-1}$)	Critical nutrient loading turbid initial state (mgP $m^{-2}d^{-1}$)
Spring	0.48	0.47

3.3. RESULTS SENSITIVITY ANALYSIS

The load-response curve of a lake might be susceptible to changes in water depth and the sediment type, similar as in the validation step a sensitivity for both parameters is performed.

3.3.1. WATER DEPTH

As shown in Figure 19 the adjustment of the water table influences the load-response curve. On average, a water depth of 2.96 m results in 6% lower chlorophyll levels, a depth of 3.22 m results in 3% higher chlorophyll levels, as compared to the baseline. There is still no sign of hysteresis and the response curve is still non-linear.

With a water depth of 2.96 m the peak chlorophyll-a concentration is roughly 28.5 mg chlorophyll-a per m³, reached at an external load of 0.33-0.37 mgP $m^{-2}d^{-1}$. The peak of the baseline water depth of 3.22 m is reached with a slightly higher external load later and is higher with a concentration of 30 mg chlorophyll-a per m³ corresponding to a P load of 0.37 mgP $m^{-2}d^{-1}$. An even bigger average water depth, 3.48m, causes again a slightly higher peak at a higher external nutrient load (31.4 mg chlorophyll-a per m³ at an external P load of 0. 42 mgP $m^{-2}d^{-1}$). After the peak the chlorophyll-a concentrations go down with an increasing external P load. Around a load of 1 mgP $m^{-2}d^{-1}$ they all stabilize around a value of 23 mg chlorophyll-a per m³.

Table 12: The critical nutrient loadings for a clear and turbid initial state for the thresholds of 20 and 30 mg Chlorophyll-a per m³ for different water depths.

Critical loading for a 20 mg chlorophyll-a per m³ threshold		
depth	Critical nutrient loading clear initial state (mgP $m^{-2}d^{-1}$)	Critical nutrient loading turbid initial state (mgP $m^{-2}d^{-1}$)
2.96m	0.27	0.25
3.22m	0.29	0.29
3.48m	0.29	0.23
Critical loading for a 30 mg chlorophyll-a per m³ threshold		
depth	Critical nutrient loading clear initial state (mgP $m^{-2}d^{-1}$)	Critical nutrient loading turbid initial state (mgP $m^{-2}d^{-1}$)
2.96m	n.a	n.a.
3.22m	0. 37	0. 37
3.48m	0. 38	0. 35

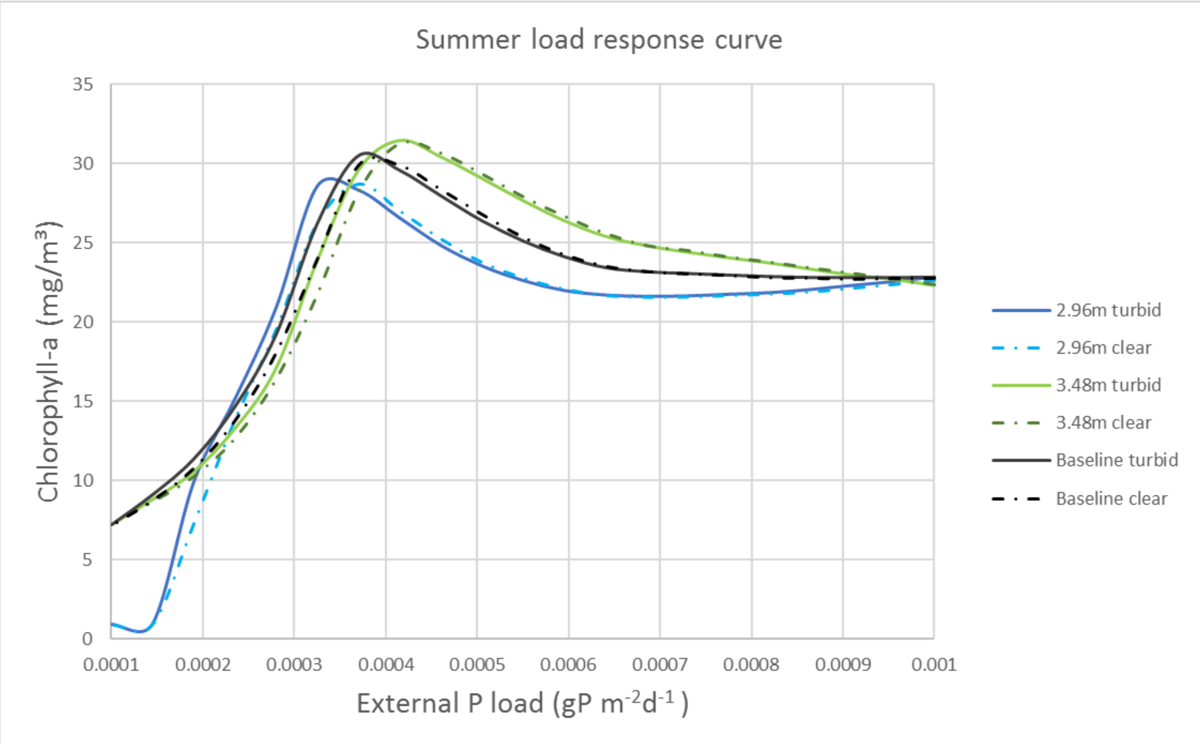


Figure 19: The summer load-response curve with differentiating water depths

If the lake is simulated with a water depth of 2.96 m both the load-response from a clear and a turbid initial state show an on average 6% decrease of the chlorophyll-a concentration, this is mainly due to the lower concentration over the low external load range and the concentrations at which the load-response curve stabilizes. As is also visible in Figure 19 the trend of the low water table slightly differs from the baseline load-response curve, the R^2, however, is still 0.98, which is high. A larger water depth (3.48m) shows a deviation of the slope of 2 – 3% higher values for the chlorophyll-a concentration (see Figure 20).

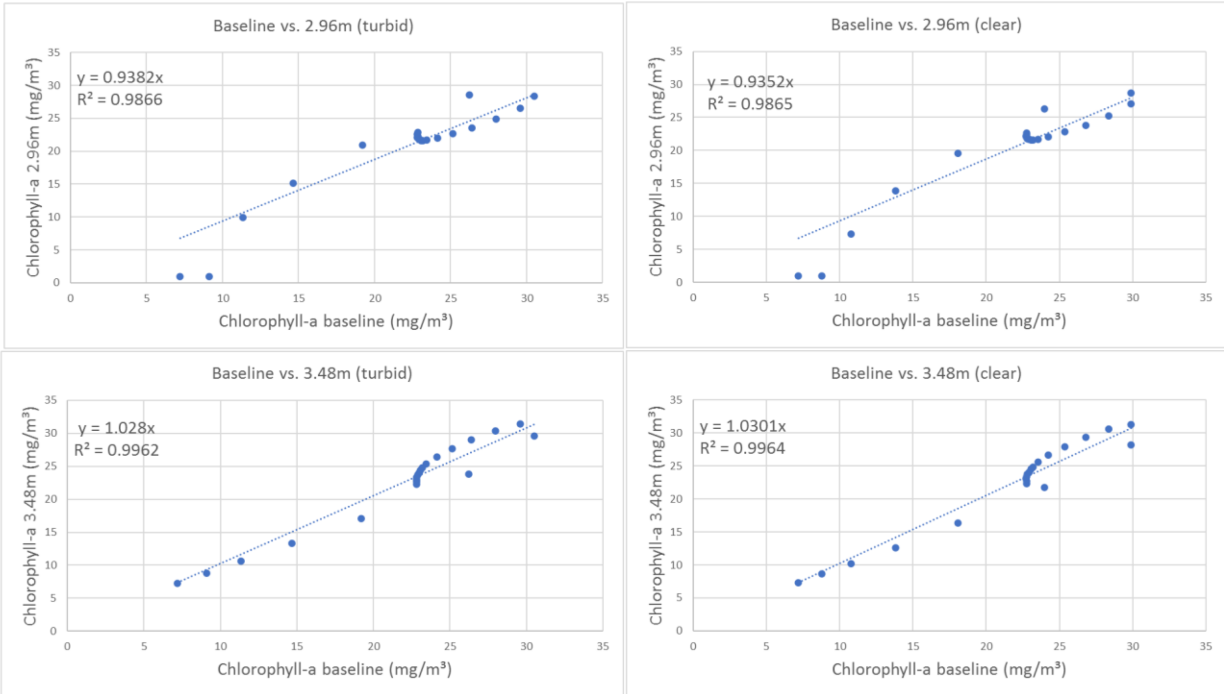


Figure 20: Sensitivity of the load-response curve towards differing water depths

3.3.2. SEDIMENT

To assess the effect of the sediment parameters on the load-response curve and critical nutrient load a sensitivity analysis is executed with sand, clay, and peat soil values Table 3 contains the adjusted parameters and the values used in this sensitivity analysis.

The different sediment types produce a slightly different load-response curve (Figure 21). All load-response curves show a similar pattern and are non-linear without hysteresis. The most predominant difference is the height of the peak with a value of 25.3 mg chlorophyll-a per m^3 when using a sandy sediment type. The peaks of the other sediment types fall in the range between 30– 31 mg chlorophyll-a per m^3. The lake with a sandy sediment type reaches its peak at an external P-load of 0.33 mgP $m^{-2}d^{-1}$, where the other types and the baseline reach their peak around 0.37-0.42 mgP $m^{-2}d^{-1}$. The load-response curves stabilize at different concentrations of chlorophyll-a. The baseline lake and the lakes with clay and peat sediment types settle after the initial peak around a chlorophyll-a concentration of 23 mg chlorophyll-a per m^3. Sand stabilizes after the peak around a value of 25 mg chlorophyll-a per m^3.

Table 13: The critical nutrient loadings for a clear and turbid initial state for the thresholds of 20 and 30 mg Chlorophyll-a per m³ for different sediment types.

20 mg chlorophyll-a per m³		
Sediment type	Critical nutrient loading clear initial state (mgP $m^{-2}d^{-1}$)	Critical nutrient loading turbid initial state (mgP $m^{-2}d^{-1}$)
Clay	0. 29	0.29
Peat	0. 26	0.26
Sand	0. 26	0. 26
Baseline	0. 29	0.29

30 mg chlorophyll-a per m³		
Sediment type	Critical nutrient loading clear initial state (mgP $m^{-2}d^{-1}$)	Critical nutrient loading turbid initial state (mgP $m^{-2}d^{-1}$)
Clay	0.38	0.37
Peat	0.41	0. 37
Sand	n.a	n.a.
Baseline	0. 37	0. 37

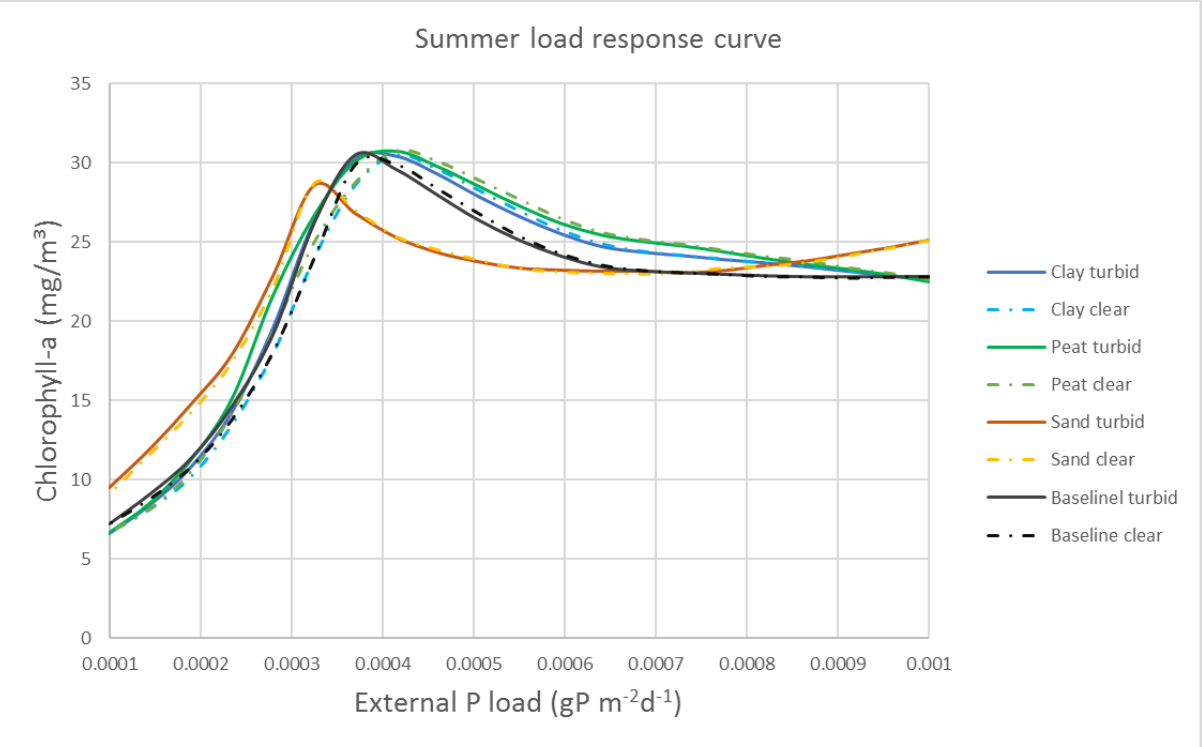


Figure 21: The summer load-response curve with differing sediment types

If we look at the regression lines of the baseline chlorophyll-a concentrations plotted the chlorophyll concentrations with different sediment parameters, it is evident that clay and peat have the least influence on the load-response in comparison with the baseline. Both sediment types result in a slope of the regression line

3% higher compared the 1:1 line (Figure 22). Sand gives slightly lower values of chlorophyll when looking at the load-response curve, however, with a regression only 1% lower than the 1:1 line sand, along with peat and clay had a negligible influence on the load-response curve compared to the baseline. The baseline sediment values can, therefore, be used.

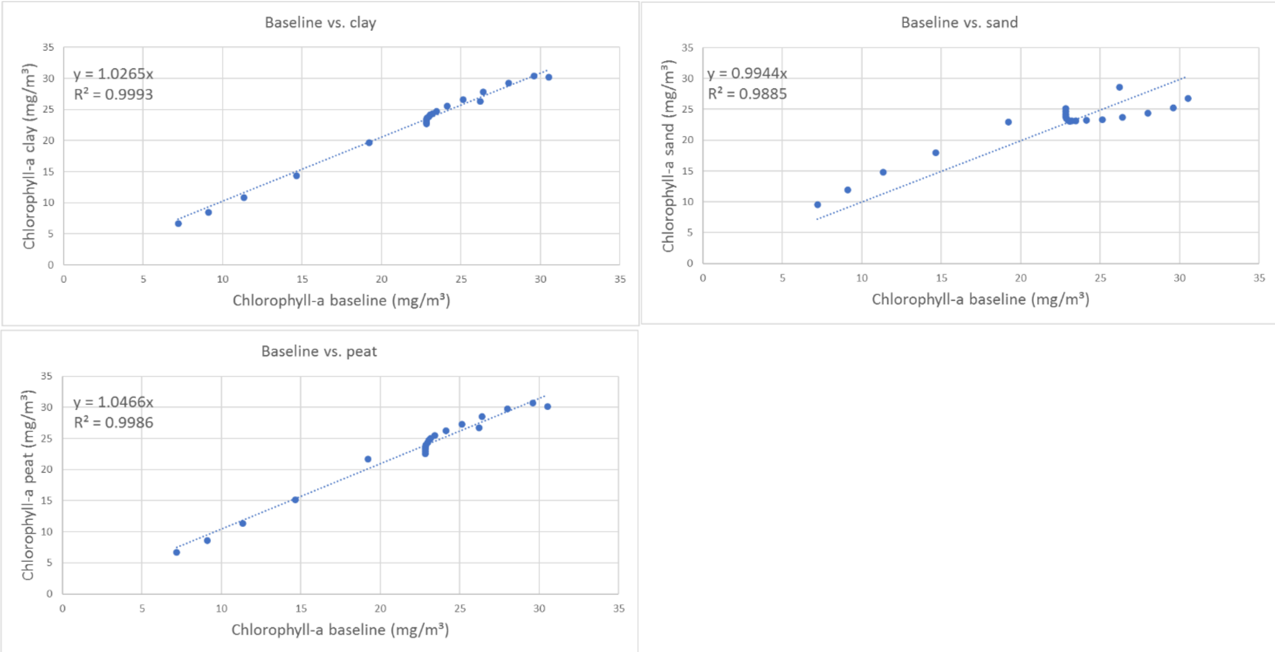


Figure 22: Sensitivity of the load-response curve towards differing sediment types

3.4. COMPARING THE CRITICAL NUTRIENT LOAD TO THE EXTERNAL LOAD

To assess whether lake restoration measures are needed I compare the found critical nutrient loads with the external nutrient load of lake Enäjärvi over the period 2000-2017. I herefor took the average external load for the same days for which the critical nutrient load was determined (day 150-210)

The external nutrient load is compared to the critical nutrient loadings corresponding the 20 and the 30 mg chlorophyll-a per m³ threshold in Figure 23. The 20 mg chlorophyll-a perm³ threshold is exceeded six times in the period 2000-2017 and the 30 mg chlorophyll-a per m³ threshold is surpassed four times. The graph shows that the average summer external nutrient load, at least in the past, is always fairly close to the critical nutrient load. The average summer external nutrient load is used to make a comparison to the critical nutrient load which is also calculated for the summer. The total summer average chlorophyll-a concentration is higher than the critical nutrient load corresponding to the 20 mg chlorophyll-a per m³ threshold (total summer average: 0.33 mgP $m^{-2}d^{-1}$, critical nutrient load: 0. 29 mgP $m^{-2}d^{-1}$). This both implies that the lake fluctuates between states With chlorophyll concentration under the threshold and above the threshold.

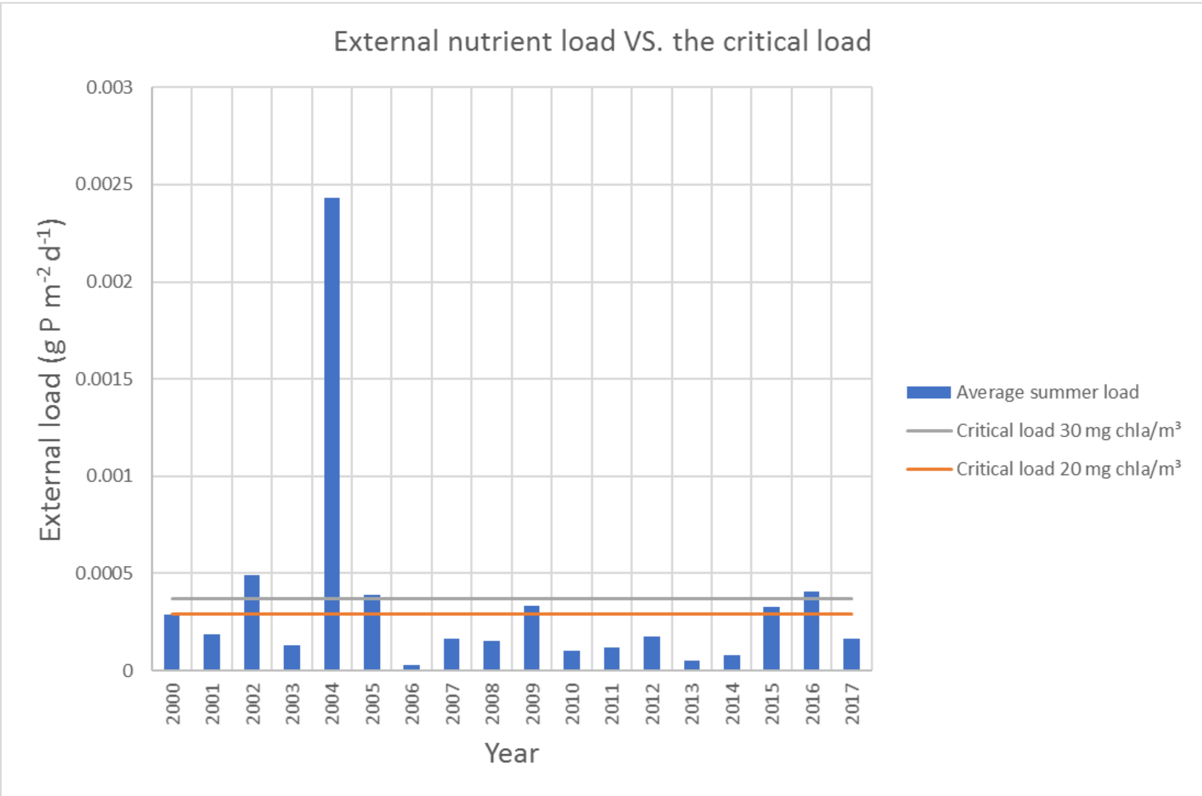


Figure 23: The summer (day 150-210) average external nutrient load per year with the critical nutrient loading for the 20 and 30 mg chlorophyll-a per m³ thresholds.

3.5. DISCUSSION

Producing the load-response curve brought up some discussion points regarding the sensitivity of the model to water depth and sediment type, the spring diatom bloom, and general discussion points regarding the bifurcation analysis.

The sensitivity analysis is performed to answer the question "How sensitive is my output to uncertainty in the input as used in the baseline analysis?". The different water depths do not give significant differences in the load-response curves. Also, clay and peat soils do not give significant differences. A sandy sediment type, however, results in a slightly different load-response curve. However, given that the peak value does not differ much and the load-response over the lower nutrient range is very similar this difference does not appear in the critical nutrient load. the 20 mg chlorophyll-a per m³ threshold gives a critical nutrient load of 0. 29 mgP $m^{-2}d^{-1}$ for a clear initial state and 0.21 mgP $m^{-2}d^{-1}$ for a turbid initial state for the baseline characteristics, with a sandy sediment type the critical nutrient loadings are 0. 27 mgP $m^{-2}d^{-1}$ and 0. 25 mgP $m^{-2}d^{-1}$ respectively. Furthermore, sand is in general not a dominant sediment type in the southern Finnish lakes (Personal communication), and is therefore not likely to be the dominant sediment type. Further sampling of sediment data is not needed since the found differences in the sensitivity analysis were negligible.

The spring bloom produces substantially higher chlorophyll-a concentrations and lower critical nutrient loading (Figures 17 and 18). The load-response curve is nonlinear without a significant form of hysteresis. The spring peak mainly consists of diatom algae which are less harmful compared to blue algae. It is good to consider the critical nutrient load of the spring period to design management measures since it might influence the food web and species distribution, nutrient availability and light regime in the months after the bloom. The spring bloom and corresponding load-response curve and critical nutrient loading should be considered because the spring bloom can be occurring. However, since PCLake+ simulates the spring bloom but observational data lacks for the spring months, the abundance and timing of the bloom cannot be validated. Silica mainly influences the growth of diatoms. However, there is no silica data available and the default silica ratio is assumed in my PCLake+ simulations. Henceforth, this is an extra source of uncertainty and might lead to an incorrect estimation of the diatom concentration increase. Also, VEMALA V.3 does not simulate a spring bloom, possible becauseVEMALA V.3 only simulates one algal species and does not take silica into account, which is one of the main drivers of diatom growth. It is therefore recommended to sample the chlorophyll-a concentrations also in the spring months to provide a validation reference to the PCLake+ results to be able to justify the use of the spring load-response curve.

Related to an overestimation of the spring blooms (see also chapter 2) is that the summer load response curve might be inaccurate if the default silica concentration (3 mgSi/m³) is inaccurate. Appendix F shows the load response curve with a 0.5 gSi/m³ inflow concentration for comparison, resulting in a lower peak and higher critical nutrient load (which would be positive for the lake). This could imply that the underestimation of the summer chlorophyll-a concentrations shown in figure 13 is not caused by an overestimation of the silica default concentration (in that case the peak of the load response curve would be higher) but rather by either an error in the input external loading or some uncaptured internal processes. This would than imply that the load response curve here simulated (figure 17) is accurate, or at least, not inaccurate due to a wrongly assumed silica input. However, as mentioned in section 2.4 the spring bloom and silica concentration can not be validated due to lack of spring measurements and silica data respectively. Additionally, the load-response curve is determined over the days 150-210 of the year, an average load response over different days may result in a different conclusion and therefor it is not yet possible to disregard a wrongly estimated silica concentration as the cause of the underestimation of the summer chlorophyll-a peaks of the second half of the validation period. Since the load-response curve in Figure 17 is "worse" I will use that one rather than the one shown in Appendix F to not underestimate the critical nutrient load.

It is important to state that the critical nutrient loadings are mere indications of the points at which the chlorophyll-a thresholds are reached. The values, however, may differ due to transient dynamics (the bifurcation is run with repeated time series to equilibrium, and does not account for trends etc.) (Janssen et al., 2017). The load response curve and corresponding critical nutrient loading are determined based on an average load respond over the days 150-210. A different period will yield a different load response curve and critical nutrient loading and perhaps even a different influence of the silica concentration. The found critical nutrient loadings therefore, in every case, should not be seen as absolute points. The critical nutrient loadings are compared to the external loading, this loading, however, is simulated by VEMALA and can, therefore, contain uncertainties as well.

I here now used the past external loading to assess whether lake restoration measures might be needed in the lake or not. This does give a view on how the external load in the lake system behaves with regards to the critical nutrient load, which is also based on this time period. However, this does not give a certain image for the future, changes such as climate change or land-use change might result in a different load-response curve or external nutrient loading resulting in a higher or lower need for additional lake restoration measures. Therefore there is a range of additional, next to using a model in general, uncertainties. The loading data does not give a clear trend of increase and decrease and therefore I suggest that the image given here for the need for additional measures might hold in the future.

3.6. CONCLUSION

This chapter aimed to answer the sub-research question; "What is the critical nutrient load of lake Enäjärvi and how does this relate to the past external nutrient loading to the lake?", by performing a bifurcation analysis in PCLake+. The critical nutrient load is dependent on the human-defined threshold. The critical nutrient load for the 20 mg chlorophyll-a per m³ threshold is 0.29 mgP $m^{-2}d^{-1}$, for the 30 mg chlorophyll-a per m³ threshold this is 0.37 mgP $m^{-2}d^{-1}$. In spring, all critical nutrient loads are in the range of 0.15 – 0.48 mgP $m^{-2}d^{-1}$ in case of a clear initial state, and 0.15 – 0.47 mgP $m^{-2}d^{-1}$ for a turbid initial state (table 11). In both the summer and spring bloom, there is no hysteresis present. The non-linearity is caused by nonlinear interactions, these interactions, however, are too weak in this case to cause hysteresis (Janssen et al., 2017).

A sand soil as sediment type is the only parameter of the sensitivity analysis that leads to statistically significant different results from the baseline. This, however, can be disregarded in the analysis due to its minimal influence and the likelihood that the whole sediment of lake Enäjärvi consists of sand.

By comparing the identified critical nutrient loading to the external P loading over the period 2000-2017 it is possible to analyse how they relate to each other and how "stable" the lake is. The critical nutrient load for the 20 mg chlorophyll-a per m³ threshold is surpassed six times, the 30 mg chlorophyll-a per m³ threshold is surpassed four times. If the threshold chlorophyll-a concentrations used in this report are considered undesirable, additional lake restoration measures would be beneficial.

4. POSSIBLE MEASURES.

In the previous chapter it became clear that the calculated critical nutrient loads in lake Enäjärvi were, in the past, surpassed several times. Therefore, I explored water quality measures and their effect on the critical nutrient loading. In this chapter, I aim to answer the sub-question "What are the effects of lake restoration measures on the critical nutrient load?". Lake restoration measures can be separated roughly in two categories: In-lake measures, aimed to alter the load-response curve and thereby the critical nutrient loading of the lake, or tip the system to an alternative state, and measures aimed to reduce the external nutrient load, thereby shifting the position of the lake on the load-response curve. A combination is also possible.

The methodology clarifies how the possible lake restoration measures are found and modelled, the methodology section is followed by the results, a discussion, and the conclusion.

4.1. METHODOLOGY

To quantify the effects of lake restoration measures on the load-response curve of Enäjärvi and thereby the critical nutrient loading for the chlorophyll-a thresholds, I first identified several potential measures that are possible to model in PCLake+ (Table 14). Based on literature I hypothesize their effects. With the help of the bifurcation analysis, hence following the method as described in chapter 3.1.1., the new load-response curves are simulated. The input values for the modified lake parameters are based on estimations but are not always representable or possible in lake Enäjärvi itself. Yet a feasibility study on the measures is outside of the scope of this research. The load-response curves of the lake with the implemented measures are per measure compared to the baseline load-response curve so that the effect of the measures is visualised. The critical nutrient loadings are also calculated and compared to the baseline critical nutrient loadings for the various chlorophyll-a thresholds. Similar as in Figure 23, the critical nutrient loadings with the implemented measures will be compared to the past external nutrient loading to assess their effects and see whether they would be effective in lake Enäjärvi.

It is outside the scope of this research to model load reduction measures in for example the land use component of the VEMALA V.3 model. Because of this, possible measures are also not described based on a literature review.

4.2. RESULTS: POSSIBLE MEASURES

This section contains a modelling study on the possible in-lake measures. I identified several measures from literature. Their theoretical effects are discussed here. Per measure, I discuss the general principles and their possibility to be modelled in PCLake+.

Table 14: In-lake system restoration measures, with the PCLake+ parameter that is used to implement the measure, a short hypothesis and references.

Measure	Adjusted PCLake+ parameter	Hypothesis	Reference
Water level control in spring	mDepthW	Increase of macrophytes and lower concentration of chlorophyll-a.	Gulati et al., 2008
Fetch reduction	cFetch	Less wind influence and henceforth less wind-induced resuspension.	Gulati et al., 2008; Janse, 2005
Add/increase marsh area	InclMarsh fMarsh	Storing nutrients in marsh vegetation which leads to a lower in-lake concentration.	Janse et al., 2008 Lee et al., (1975)
Biomanipulation (repeated)	kHarvFishwin kHarvFishSum	Less bottom-feeding fish induced resuspension	Janse et al., 2008; Tatrai et al., 2008
Combination of fetch reduction and implementation of a marsh area	cFetch InclMarsh fMarsh	Storage of nutrients and reduced wind influence and resuspention	

An in-lake measure can be divided in "system measures" and "internal measures" (Stowa, 2008). System measures are aimed to increase the critical nutrient load, so that a higher load is possible before the lake passes the chlorophyll-a threshold (Figure 24 A, B, and C). Internal measures intent to tip the system from a turbid to a clear state or vice versa, this, however, is only possible in the case of two alternative stable states (Figure 24 D). Internal measures do not alter the load response curve but drastically decrease the chlorophyll-a concentration by "shocking" the lake system. Internal measures are usually bio-manipulative measures such as fish catching (Stowa, 2008). Figure 24 shows a graphical representation of the theoretical principle of both types of measures. Internal measures can also affect the load-response curve, depending on wether they are executed frequently or once. Since I did not find alternative stable states in lake Enäjärvi, the analysis focusses on system measures or repeated internal measures such as annual fishing events.

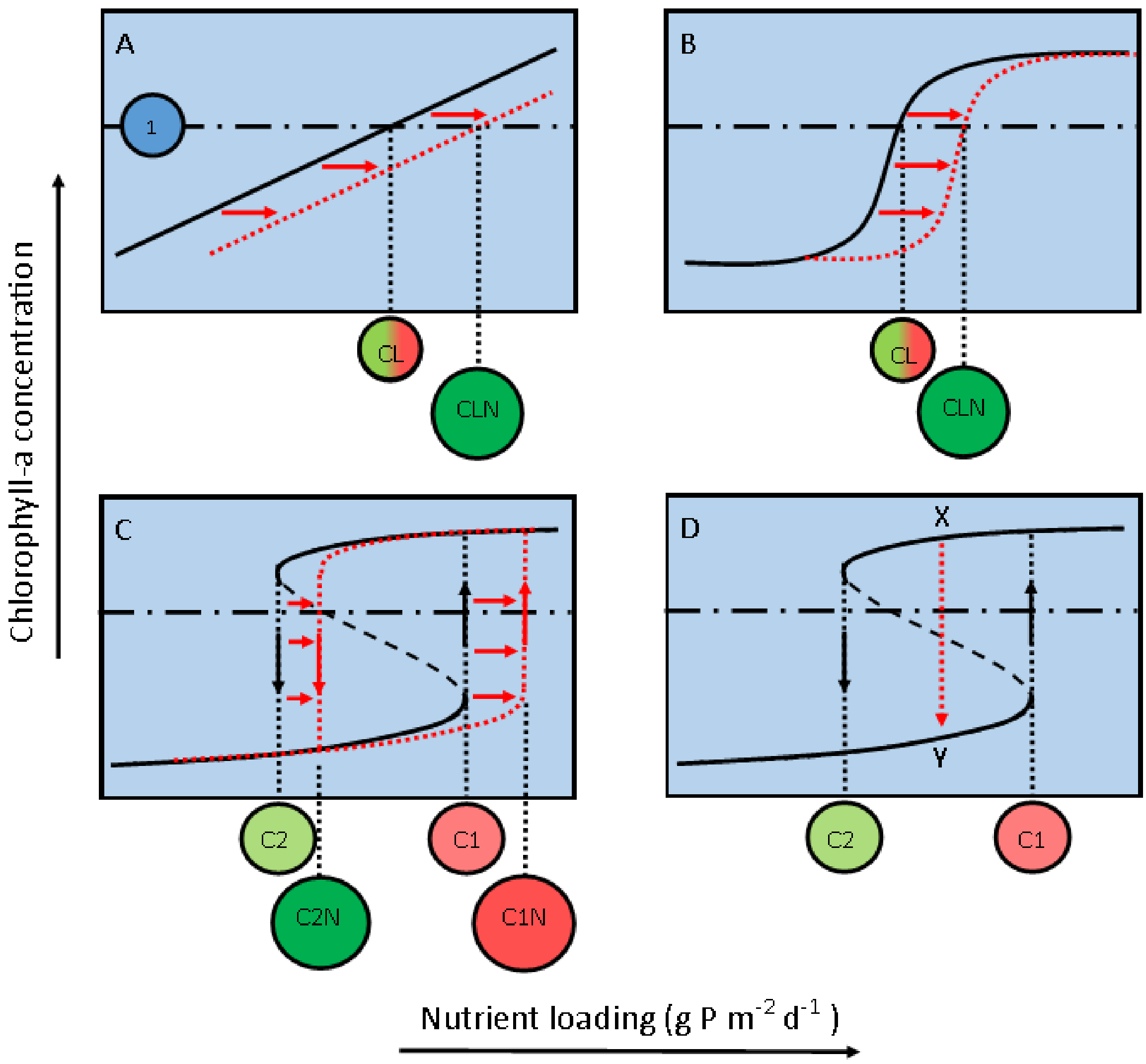


Figure 24: Hypothetical effect of in-lake measures. A, B, and C show the effect of system measures, the red dotted line represents the new load-response curve (linear, non-linear without hysteresis, and non-linear with hysteresis respectively). CLN (critical load new), C1N and C2N are the new critical nutrient loadings when the measure is implemented. Graph D depicts the working of an internal measure. In the case of two alternative states, a push (shock treatment) can tip the system to the opposing state without further nutrient reduction.

REDUCTION OF WATER DEPTH IN SPRING

A reduction of the water depth in the winter/spring months may lead to an expansion of the macrophyte biomass (Gulati et al., 2008). A shallower water level leads to more favourable light conditions for plant growth. Based on the turbidity-vegetation feedback loop I hypothesized that a more rapid increase in submerged vegetation may hamper the increase of turbidity and can, therefore, keep the lake in a clearer state. In a case study, McGowan et al. (2005) found that a decrease in water level in the winter resulted in a 2.5 times increase in macrophyte presence. The role water level fluctuations play in shallow lakes is not yet fully understood, however, Coops et al. (2003) state that managing the water level can be a sufficient tool to for the restoration of lakes. In PCLake+ the water depth in spring can be altered in the time-series mimicking the effect of managing the spring water level.

By adding the water depth to the model as a time series it is possible to adjust the water depth in the spring period. The spring water depth is reduced from 3.22 m to 3.02 m (-0.2 m), 2.72 m (-0.5 m), and 2.22 m (-1 m) for the day 31 till 150.

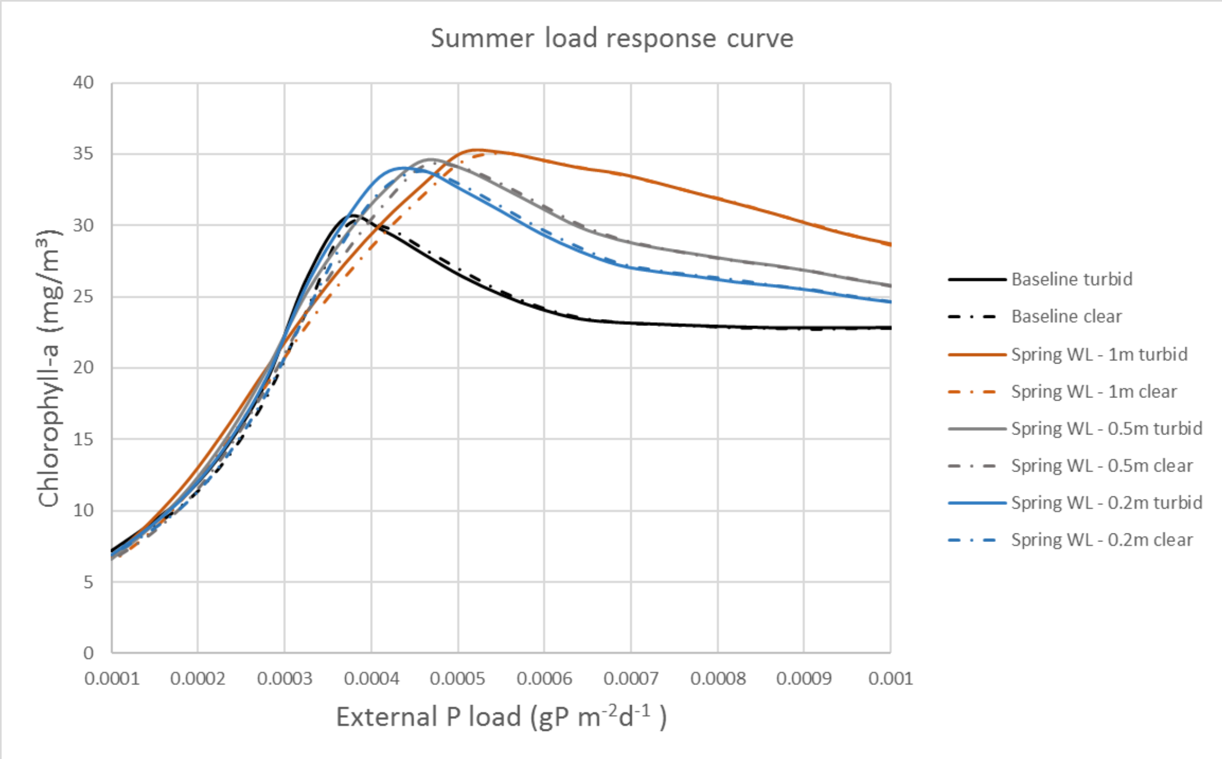


Figure 25: The summer load-response curve for the spring water lever reductions.

The load-response curves of lake Enäjärvi when a reduction of the water level in spring is implemented show an increase in the chlorophyll-a concentration peak (Figure 25). The load response curve is still non-linear without hysteresis. The critical nutrient load of the 20 mg chlorophyll-a per m³ threshold remains similar as for the baseline lake. A water-level reduction of 1 m does show a slight increase of the critical nutrient load for the 30 mg threshold, namely from 0.37 to 0.41 mgP $m^{-2}d^{-1}$. The load-response curve settles on a higher value for all three water level reduction rates. Although theorised to lead to an increase in the water quality by favouring macrophyte growth, the results of PCLake+ here show that in lake Enäjärvi, this measure is not effective. This could be due to the lack of alternative stable states (no hysteresis) hampering a macrophyte dominated state.

Table 15: The critical nutrient loadings for the reduction of the water level in spring.

20 mg chlorophyll-a per m³		
	Critical nutrient loading clear initial state (mgP $m^{-2}d^{-1}$)	Critical nutrient loading turbid initial state (mgP $m^{-2}d^{-1}$)
Spring water lever reduction: 1 m	0.29	0.28
Spring water lever reduction: 0.5 m	0.29	0.28
Spring water lever reduction: 0.2 m	0.29	0.28
30 mg chlorophyll-a per m³		
	Critical nutrient loading clear initial state (mgP $m^{-2}d^{-1}$)	Critical nutrient loading turbid initial state (mgP $m^{-2}d^{-1}$)
Spring water lever reduction: 1 m	0.42	0.41
Spring water lever reduction: 0.5 m	0.39	0.38
Spring water lever reduction: 0.2 m	0.38	0.37

FETCH REDUCTION

The fetch, or fetch length, is the distance of an open water body over which the wind blows, usually expressed from a certain average wind direction. The fetch directly influences the amount of wind-induced resuspension of sediments. The resuspension of sediments leads to enhanced turbidity. Reducing the fetch, therefore, can, in theory, reduce resuspension. The fetch can be reduced by creating islands to break the wind (Gulati et al., 2008). Janse et al. (2008) found in the PCLake+ model that increasing the fetch results in a decrease of the critical nutrient loading, implying there is a direct postive relationship between the critical nutrient load and the fetch length.

The fetch can be directly altered in PCLake by adjusting the cFetch parameter. To analyse the possible effects of a reduced fetch on the load-response curve and critical nutrient load I will half the fetch length from 2230 m to 1115 m with an intermediate fetch of 1672.5m, mimicking a small island somewhere in the lake.

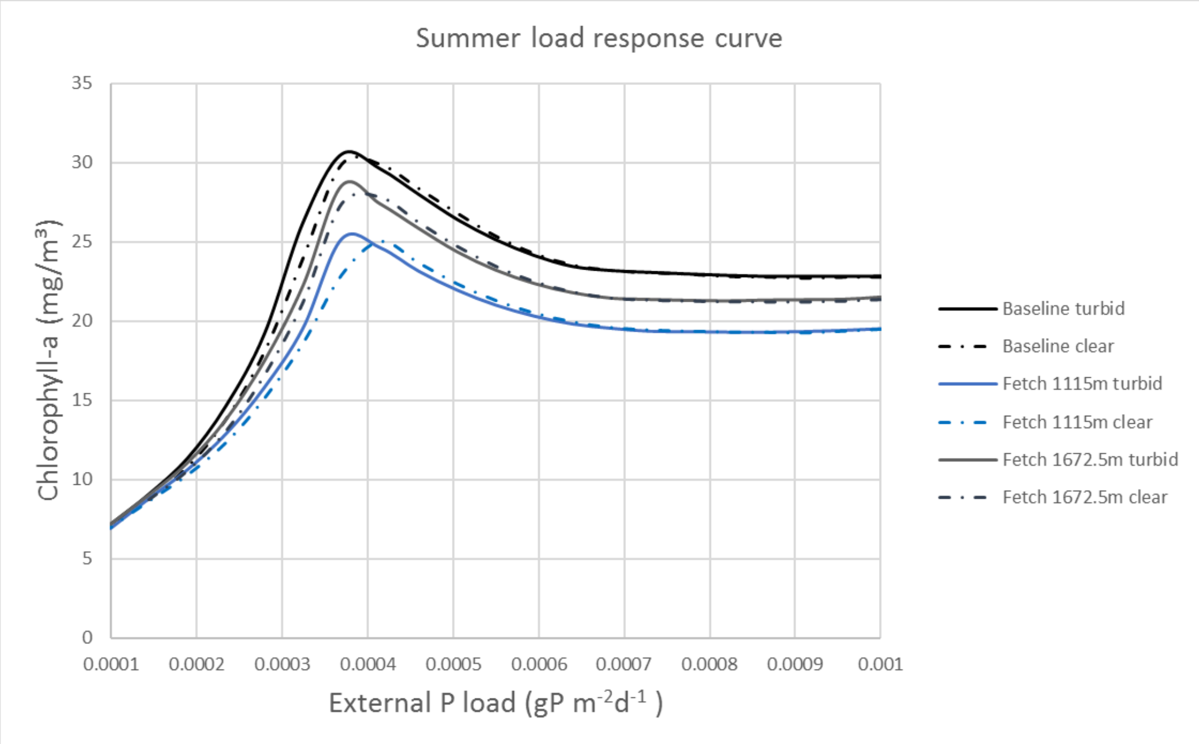


Figure 26: The summer load-response curve per fetch length.

A fetch reduction relative to the baseline (fetch = 2230m) leads to a lowering of the peak of the chlorophyll-a concentration in response to the external nutrient loading (Figure 26). The peak is reached for the different fetch lengths at the same external loading, namely, 0.37 gmP $m^{-2}d^{-1}$. This implies that the slope over the lower nutrient loading differs per fetch length resulting in slightly differentiating critical nutrient loadings (see table 16)

Table 16: The critical nutrient loadings per fetch length

20 mg chlorophyll-a per m³		
	Critical nutrient loading clear initial state (mgP $m^{-2}d^{-1}$)	Critical nutrient loading turbid initial state (mgP $m^{-2}d^{-1}$)
2230 m (baseline)	0.29	0.29
1672.5 m	0.3	0. 3
1115 m	0. 33	0. 33

INCLUDE MARSH AREA

Marshes or wetlands can be beneficial for the water quality of a lake. The main positive effect of marshes is the storage of nutrients in the marsh vegetation, henceforth, decreasing the amount of readily available nutrients in the lake water (Lee et al., 1975). Lee et al (1975) found that marshes denitrify substantial amounts of nitrate. Furthermore, the vegetative nature of marsh zones tends to trap sediments, decreasing the amount of particulate matter entering the lake (Lee et al., 1975). Janse et al. (2008) show an increase in the critical nutrient loading, so a positive restoration effect, related to an increased marsh area. This can be attributed to the uptake of nutrients by the marsh vegetation, denitrification, and extra sediment deposit area since the marsh vegetation is likely to trap extra sediment. Adding or enlarging a marsh area in connection to the lake seems a good restoration measure. However, Lee et al. (1975) also found some possible negative effects. They state that nitrogen fixation is found to also happen in the marsh areas, this nitrogen can then discharge into the lake. In total the beneficial effects seem to outweigh the negative effects (Lee et al., 1975).

PCLake+ has a build-in marsh module. This module can be turned on within the model. The marsh area is defined as the relative marsh area per lake area. Implementing a marsh area is location-specific since it dependents on the land use around the lake, a marsh can also be established in the original lake, decreasing the size of the lake to turn part of it into a marsh area. Because a location study regarding the possible development of a marsh area in or surrounding lake Enäjärvi is outside of the scope of this research, I here sole aim to show the effectivity of a marsh given the current lake characteristics and properties. The marsh areas in this study will be 10% and 30% (0.1 and 0.3 $m^2 * m^{-2}$) of the original lake size and placed outside of the current lake borders, meaning that the lake size will remain the same.

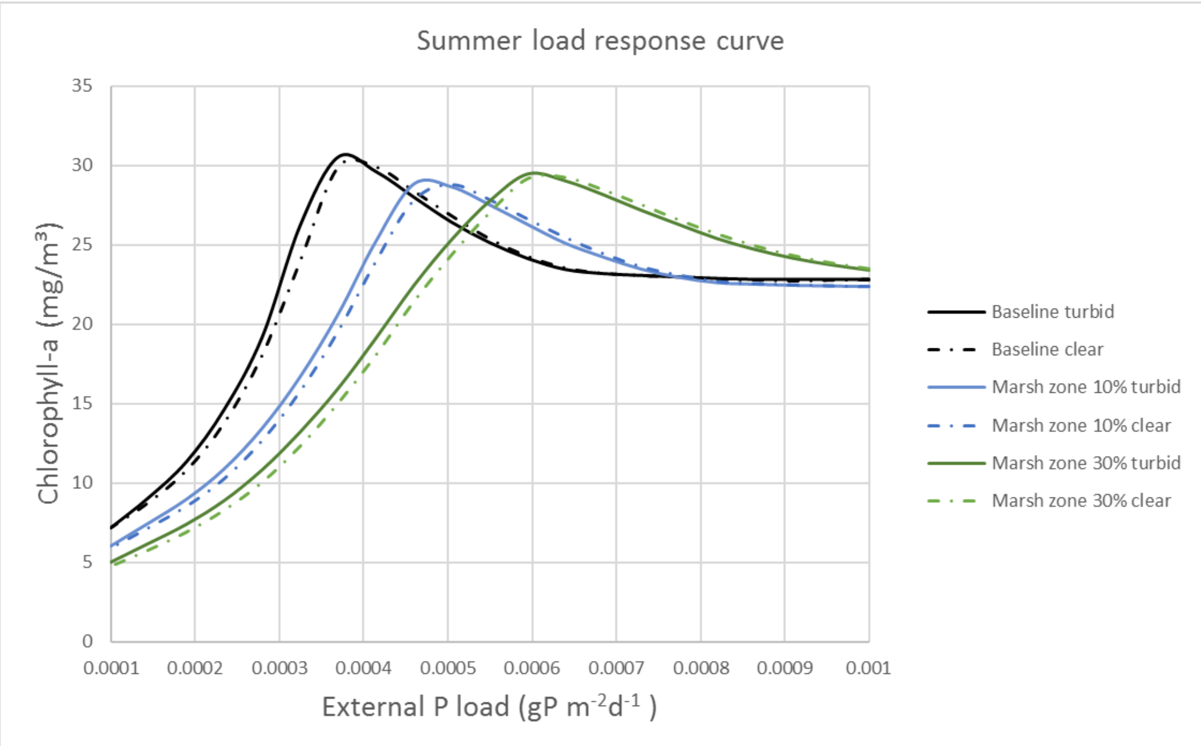


Figure 27: The summer load-response curves per marsh size.

Adding a marsh zone of 30% of the lake size to the system of Enäjärvi results in PCLake+, as shown in Figure 27, in a substantial increase of the critical nutrient loading from 0. 29 mgP $m^{-2}d^{-1}$ to 0. 43 mgP $m^{-2}d^{-1}$. An intermediate marsh size consisting of 10% of the lake size results in a critical nutrient load of 0.00037 g P $m^{-2}d^{-1}$. The peak of the load-response curve, however, is not much lower as compared to the original situation.

Table 17: The critical nutrient loadings per marsh size

20 mg chlorophyll-a per m³		
	Critical nutrient loading clear initial state (mgP $m^{-2}d^{-1}$)	Critical nutrient loading turbid initial state (mgP $m^{-2}d^{-1}$)
No marsh (baseline)	0.29	0.29
Marsh zone 10% of the lake size	0.37	0.37
Marsh zone 30% of the lake size	0.43	0.43

BIOMANIPULATION (ANNUAL)

Biomanipulation is often an example of an internal but if it is repeated with a certain regularity it can be regarded as a system measure. The most obvious form of biomanipulation in lakes is the removal of the benthivores (bottom feeding) fish, hereby reducing their negative effects. A reduction in benthivores results in less resuspension and a reduction of the grazing on zooplankton by young benthivorous fish. Fish removal has proven to be a suitable measure in the restoration of lakes in many cases (e.g. Bernes et al., 2015;Husi et al., 2008; Annadotter et al., 1999; Jeppesen et al., 1990), but is not in every situation a guarantee for success (Bernes et al., 2015). Fish stocks and biomanipulation by fish removal can be managed in several ways. One time fish removal will most likely only have long term effects on the water quality if there are alternative stable states under the present external nutrient loading range. It is also possible to integrate fish removal more permanent in a lakes management by allowing, for example, a fishing club. Usually, benthivores and planktivorous fish are aimed for removal (Tetrai et al., 2008; Bernes et al., 2018; Hamilton, 1992). Piscivorous fish need to remain in the lake to hunt the left planktivorous fish. To strengthen this the hunting pressure they can be introduced to the lake as an additional measure. Adding piscivores alone is less effective (Bernes et al., 2015;), likely due to high mortality among the stocked fish by cannibalism or predation, or bad timing (Søndergaard et al., 2007)

Fish harvesting is often used as a perturbation to flip the lake system to another stable state (Scheffer, 2009). Since I did not identify such alternative stable states in lake Enäjärvi in the summer period a one-time fishing event does probably not have much effect on the load-response curve. Therefore, I will implement an annual fishing event. Since annual biomanipulation may be costly annual fishing events can, as a hypothetical example, also be executed by introducing a fishing club where the fish are not thrown back into the lake. How annual fishing might look like is further outside the scope of this research.

In PCLake+ the fish harvesting fraction (in days) can be adjusted. To analyse the effects of fishing the harvesting fraction was changed to 10 days and 30 days.

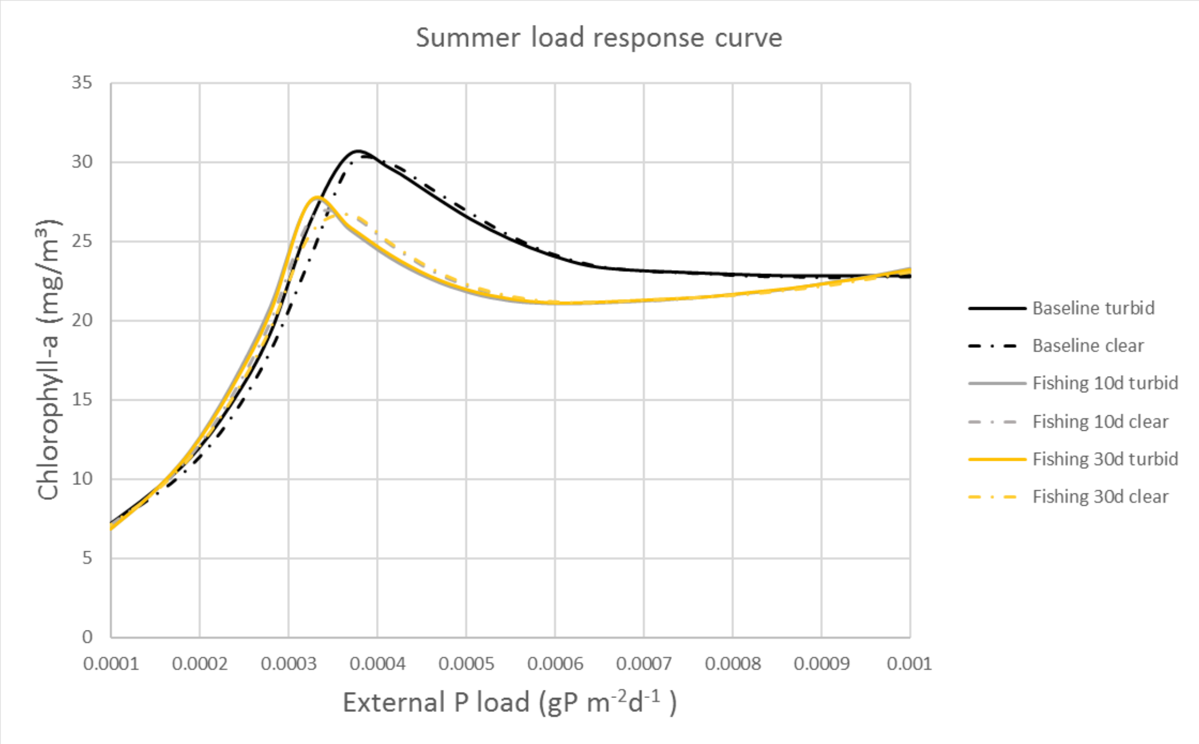


Figure 28: The summer load-response curve per fishing intensity in days.

Introducing annual fish harvesting, for example in the form of a fishing club, affects the height of the peak of the load-response curve (Figure 28). Both fishing intensities give a similar load-response curve. The critical nutrient load does not increase relative to the baseline.

Table 18: The critical nutrient loadings per fishing intensity.

20 mg chlorophyll-a per m^3		
	Critical nutrient loading clear initial state (mgP $m^{-2}d^{-1}$)	Critical nutrient loading turbid initial state (mgP $m^{-2}d^{-1}$)
No fishing (baseline)	0.29	0.29
Fishing harvesting 10 d^{-1}	0.28	0.28
Fishing harvesting 30 d^{-1}	0.28	0.28 g

COMBINATION OF A FETCH REDUCTION AND THE CREATION OF A MARSH ZONE

Adding a marsh zone of 30% is the most effective measure identified in this study, reducing the fetch also has a substantial influence on the load-response curve and the critical nutrient loading. Combining a marsh zone and a reduction of the fetch length is a reasonable combination if the marsh zone replaces part of the lake surface water area, hereby reducing the fetch.

I choose for an intermediate fetch reduction of 25% (Fetch = 1672.5m) and a marsh zone of 30% of the surface of the lake. A marsh of 30 % is the most effective measure simulated. 30% is a fairly sizable marsh zone but also still fairly realistic. Bigger marshes are possible, but the size is dependent on the function of the lake and the surrounding land use. I choose a fetch reduction of 25% arbitrarily, a fetch reduction of 50% or different is also possible. However, I aim here merely to show the effect of both measures together and not every possible combination.

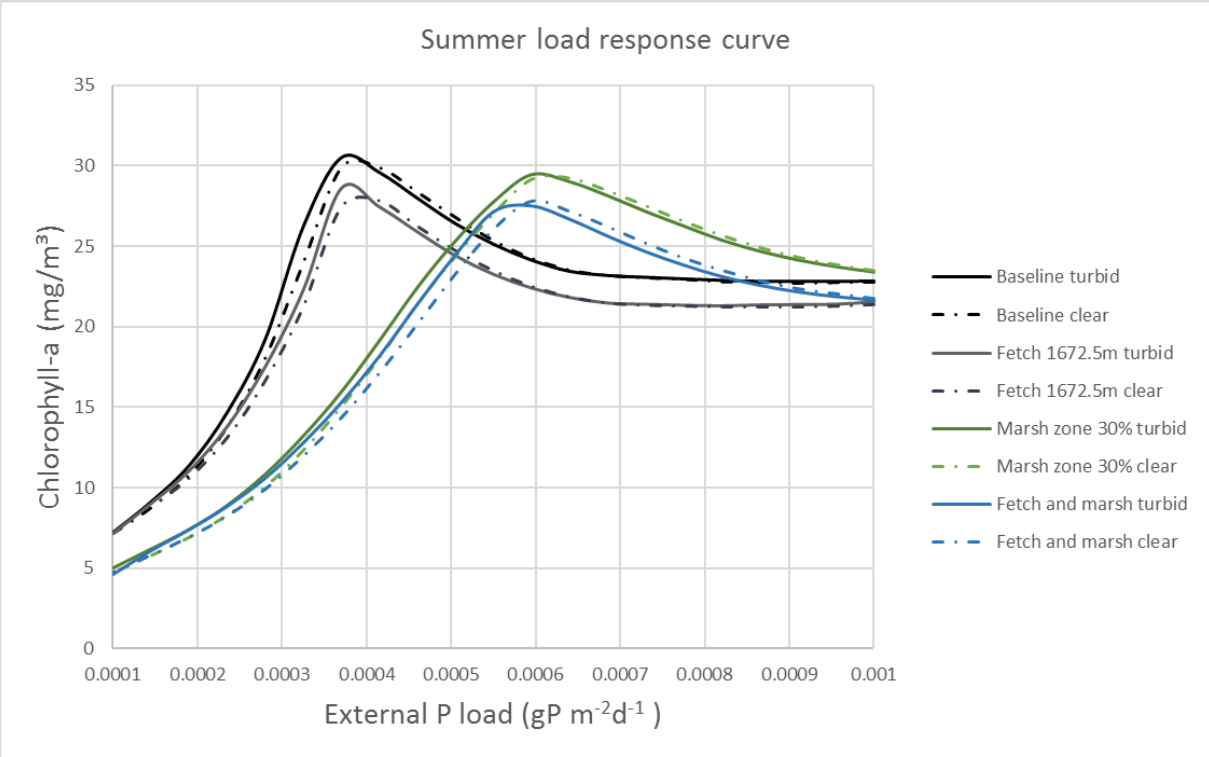


Figure 29: The summer load response curve for a fetch reduction, implementation of a marsh zone, and a combination.

Combining a marsh zone and a fetch reduction yields a load-response curve similar to sole adding a marsh zone of 30% however the peak value is slightly lower which is attributed to the fetch reduction (Figure 29). The critical nutrient load for the 20 mg chlorophyll-a per m³ is slightly higher as when only a marsh zone is implemented (0.43 vs. 0.44 g P $m^{-2}d^{-1}$).

Table 19: The critical nutrient load for a fetch reduction and marsh zone combination.

20 mg chlorophyll-a per m³		
	Critical nutrient loading clear initial state (mgP $m^{-2}d^{-1}$)	Critical nutrient loading turbid initial state (mgP $m^{-2}d^{-1}$)
Marsh zone of 30% and fetch reduction of 25%	0.44 g P $m^{-2}d^{-1}$	0.44 g P $m^{-2}d^{-1}$

4.3. COMPARING THE CRITICAL NUTRIENT OF THE MEASURES LOAD TO THE EXTERNAL LOAD

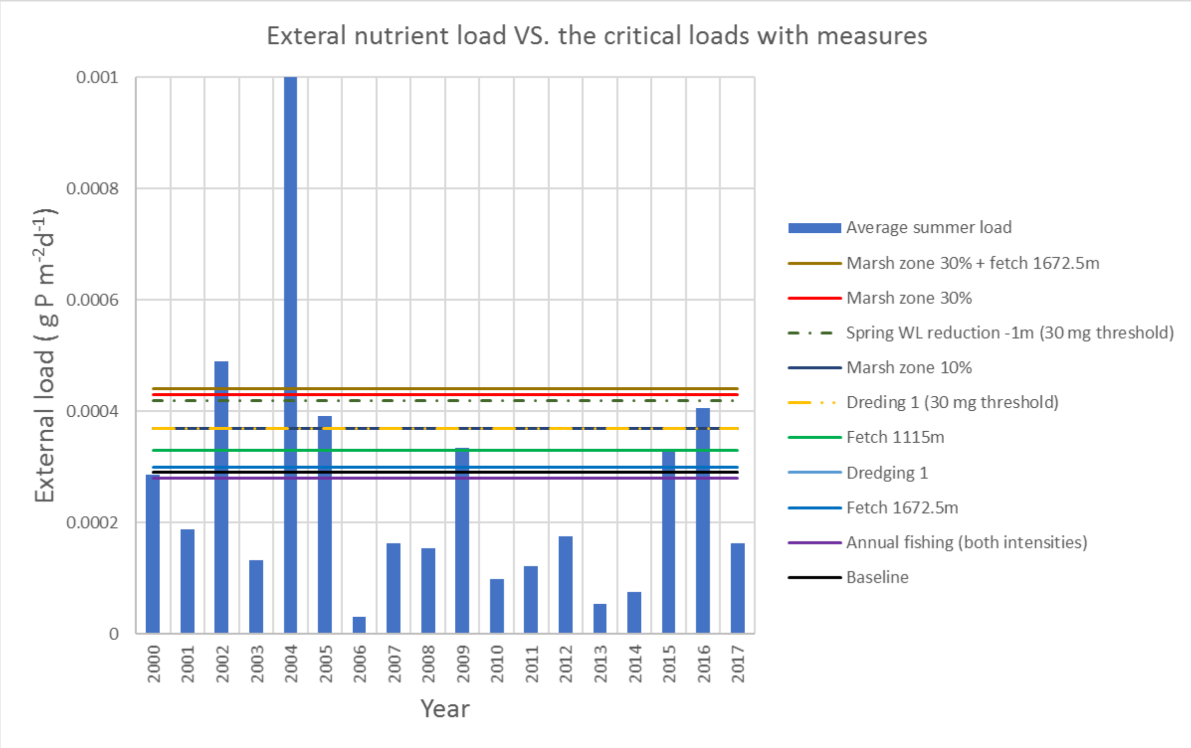


Figure 30: The external nutrient load of the period 2000-2017 in relation to the critical nutrient loading of the various measures for the 20 mg chlorophyll-a per m³ threshold, unless indicated differently. The critical nutrient loads of the water level reduction in spring for the 20 mg chlorophyll-a per m³ threshold are not included in the figure since they are similar to the baseline critical nutrient load, the critical load for the 30 mg threshold is only included for a reduction of one meter since only this one shows an increase compared to the baseline (0.00037 gPm^{-2}d^{-1}).

Figure 30 gives an overview of the effect of the measures on the critical nutrient load and how these respond to the past external load. The critical nutrient loading with the measures can be compared to the external loading in the lake system over the past to see their relative effect. Adding a marsh zone of 30% of the surface of the lake Enäjärvi to the system results in a substantial decrease in the number of times that the critical nutrient load for the 20 mg chlorophyll-a per m³ threshold would have been surpassed (from six times to two). A marsh zone of 10% also increases the stability of the clear state. Fishing as modelled here and reducing the fetch with a quarter does not have much effect. A further reduction of the fetch to 1115m also leads to a reduction in the frequency that the 20 mg chlorophyll-a per m³ critical nutrient loading is surpassed from six to four. A combination of a fetch reduction of 25% and a marsh zone of 30% of the surface of the lake does not yield a different critical nutrient loading as compared to only adding a marsh zone of 30%, but does result in a lower peak value. Lowering the water level in spring with 1m does increase the critical nutrient load for the 30 mg chlorophyll threshold. Since the critical nutrient load of the other reductions and for the 20 mg chlorophyll thresholds remain similar to the baseline they are not included in the figure for clarity sake. However, since the peak of the load response curve is higher, and all other measures result in a load response curve with peaks lower than 30 mg chlorophyll per m³ lowering the water table in spring does not seem effective.

4.4. DISCUSSION

The output of the model runs for different measures described in this chapter show different effect in lake Enäjärvi. However, The values chosen for these measures are arbitrary and thus should be taken as an example rather than absolute results. The values used for the adjusted parameters to mimic lake restoration measures are based on literature and logical estimations in relation to the lake characteristics but are not extensively assessed on implementation possibilities. For example; adding a marsh size of 10% or 30% of the lake area is an effective measure and the size is theoretically not excessive, however, it is lake dependent whether the implementation is possible. Is there a place for a marsh zone of this size? This applies for most measures. In this study the marsh module is not used for the baseline simulations. Lake Enäjärvi, however, does already have wetlands designed to reduce the external P load of which 3 are well enough designed to reduce the P load (Pellikka and Sammalkorpi, 2020). The marsh module in PCLake+ assumes an open exchange between the marsh zone and the lake, implying in and outflow, since for the majority of the wetlands around lake Enäjärvi there is only a one sided connection (flow from wetland to the lake) and for the others it is unknown to my knowledge, the marsh module is not standardly used (Pellikka and Sammalkorpi, 2020; Sollie et al., 2008). Further research on the functioning of the wetlands around Enärärvi is needed to validate if the decision to not use the marsh module for the baseline simulations is indeed valid. However, the relative size of the present wetlands is substantially small that this might possibly not have a great effects on the results. Follow up studies are needed to assess for example the feasibility, costs, carrying capacity of the involved stakeholders.

The simulated measures largely supported the stated hypotheses in table table 14, but in addition to these, quantify their effect in lake Enäjärvi. However, it was hypothesized that lowering the water level in spring would spark a domination by macrophytes and increase the critical nutrient loading, this however, did not happen. This could be explained by the lack of alternative stable states (no hysteresis) hampering a macrophyte dominated state.

The effect of the measures is assessed, in terms of the critical nutrient loading, to the external nutrient loading of the period 2000-2017. The measures should have the goal to perform well in the future so a comparison to the predicted future external nutrient load would give a more complete indication of their future effectiveness.

As discussed in section 3.5 the summer load-response curve might be different (more resembling Appendix F). This, however, does not likely affect the relative effect of the simulated lake restoration measrures. The absolute effect may differ.

4.5. CONCLUSION

The aim in this chapter was to answer the sub-research question; "What are the effects of lake restoration on the critical nutrient load?". To answer this I identified several lake restoration measures which could be simulated with the use of PCLake+. By performing a bifurcation analysis with adjusted parameters corresponding to the measures I aimed to identify the relative effect on the load-response curve and the critical nutrient loadings for the human-defined chlorophyll-a thresholds.

All identified measures lead to a load-response curve which does not surpass the 30 mg chlorophyll-a per m^3 threshold anymore. Their differences however on the critical nutrient load for the 20 mg threshold are noticeable. The measure with the most substantial increase of the critical nutrient load is adding a marsh zone to the lake. A marsh zone of 10% of the lakes surface size leads to a 28% increase of the critical nutrient loading to 0. 37 mgP $m^{-3}d^{-1}$, adding a marsh zone of 30% yields a critical nutrient load of 0.43 mgP $m^{-3}d^{-1}$ (48% increase). Reducing the fetch by 50% also yields an increase of the critical nutrient loading of roughly 13%. A combination is of both these measures is also possible since they can complement each other, however, this does not result in a higher critical nutrient load when compared to only adding a 30% marsh zone, but does yield a lower peak.

Annual fishing results in a lower peak but the critical nutrient load of the 20 mg chlorophyll-a per m^3 does not change.

Temporarily lowering the water table in the spring months results in an increase of the peak of the load-response curve. The critical nutrient load for the 30 mg chlorophyll-a per m^3 threshold does increase, contrary to the critical nutrient load for the 20 mg chlorophyll-a per m^3 threshold, which stays roughly the same as for the baseline lake. This measure therefore does not seem effective.

5. DISCUSSION

All the chapters included a discussion on the limitations and uncertainties found in the respective chapters. Additional to this there are overarching discussion points which I will mention here along with some points from the previous discussions which require, in my view, further attention. A major and general part of the discussion of a modelling study is the uncertainties all models bring along, since they are merely a representation of reality. PCLake+ and VEMALA are process-based models meaning that the relations, feedbacks, and default settings are not finetuned specifically for lake Enäjärvi. Furthermore, the "real world" outside of the models is far more complex than what at this moment is possible to capture. The uncertainty surrounding model studies manifests itself in this thesis firstly in certain assumptions made regarding the lake parameters. Process-based models often need extensive datasets as input and lack certain data of this can result in uncertainties. Secondly, there is an ambiguity surrounding the load-response curve and critical nutrient loading. The first point I tried to discard by performing several sensitivity analyses for PCLake+, these analyses showed that PCLake+ was not sensitive to the assumed static water depth and the assumed sediment variables. For the second point, however, it should be noted that the load-response curve, and thereby the critical nutrient loading, are indications of the external load at which certain chlorophyll-a concentrations become apparent. Janssen et al. (2017) state this clearly by noting that "due to transient dynamics, the response of lakes may be different than would be expected based on the identified critical nutrient load" (PG. 285). Nevertheless, the critical nutrient load does provide insight into the behaviour of the lake and gives a well-founded base of the external nutrient load at which an undesirable algae concentration might become apparent. It should be kept in mind that the found critical nutrient load has uncertainty and should not be used as an absolute point or "holy value". The advantage of using a model is that endless simulations and tests can be done to analyse the water quality and quantify the effects of lake restoration measures. Often in situ experiments on whole lake ecosystems are costly or simply not possible. Modelling allows for extensive research and can provide a solid base for future management.

One major discussion point, in my opinion, is the underestimation of the models of summer chlorophyll-a concentrations in the second half of the validation period and the simulation of a spring bloom by PCLake+. As discussed in the previous discussions this might be due to an underestimation or miss of an increasing trend in the external load, this would not greatly affect the load-response curve and critical nutrient load. A second possibility is that certain internal processes are missed, and thirdly the silica concentration might be estimated wrongly. However, since silica data is not available, similarly to chlorophyll-a measurements in spring to either validate or discard the spring peak, it was not yet possible in this thesis to find the origin of the inability of both models to follow the increasing chlorophyll-a trend.

In comparison with other studies this thesis found some comparable results both for model performance and found effectiveness of the tested lake restoration measures. The PCLake+ performance for the chlorophyll-a concentrations is similar as found in Janssen et al., (2017) of an assessment of lake Taihu, China. The R^2 found in this thesis (0.6) is well within the range of R^2s found for different locations in lake Taihu (0.3-0.85), similarly, the RE of 1.85 in lake Enäjärvi compares to the range of roughly 0.5-4 found in Taihu. Janse (2005) found, similar to this thesis, that the existence of a marsh zone does increase the critical nutrient load substantially. The same study also found that decreasing the fetch also increased the critical nutrient load markedly, similar as found here, however, the increase was more than the presence of a marsh zone which is contrary to the results presented in this thesis. This difference is likely due to a larger reduction in fetch in Janse (2005) as compared to this study. Also, lake characteristics such as depth and vulnerability to resuspension can partly explain this difference and therefore the results can not be compared one to one.

The simulated measures largely supported the stated hypotheses in table table 14, but in addition to these, quantify their effect in lake Enäjärvi. However, it was hypothesized that lowering the water level in spring would spark a domination by macrophytes and increase the critical nutrient loading, this however, did not happen. This could be explained by the lack of alternative stable states (no hysteresis) hampering a macrophyte dominated state.

With the results of this study lake restoration management can take a more focussed approach. The results from this study give an overview of the water quality and the effect of several lake restoration measures. This provides a base for future lake management of lake Enäjärvi. To eliminate uncertainties regarding the chlorophyll-a concentration simulated by PCLake+ and the simulated load response curve I recommend sampling of chlorophyll-a in spring to either prove or discard the presence of a spring algae bloom, additional to the sampling of silica. Furthermore, the comparison of the measures with the external loading is done over the past external loading (2000-20017). The measures should have the goal to perform well in the future so a comparison to the predicted future external nutrient load would give a more complete indication of their future effectiveness.

6. CONCLUSION

Here I aim to concisely answer the main research question *"How does the nutrient loading in lake Enäjärvi relate to the critical nutrient loading and what are water quality measures that can be taken to either keep the lake in a clear state or switch the lake from turbid to clear?"* Both models used in this thesis, PCLake+ and VEMALA, were validated for lake Enäjärvi and their performance analysed. Both models were validated for the concentrations of chlorophyll-a, NH_4, NO_3, PO_4, O_2, TN, TP, and internal P loading against observational data and compared with each other.

Both models perform reasonably well with R^2 and Mean Relative Absolute Error (RE) values falling within a range indicating a good model performance except for NH_4 and NO_3 (R^2: 0.17-0.22 and 0.5-0.57 and RE: 16.41-17.73 and 7.42-19.57 respectively for NH_4 and NO_3). The model output, by purely looking at the graphs, gives similar ranges as the observational data and (seasonal) trends are followed well. The deviation of the regression line slope compared to the 1:1 line ranges between 68% and 13%, implying that the model ouput does not extremely over or underestimate the concentrations of the modelled variables. In general PCLake+ captures the higher values better then VEMALA V.3, in the lower ranges VEMALA V.3 performs better compared to PCLake+. This gives rise to the thought that an ensemble of both models might give a good indication of the water quality parameters, further research on that account can be beneficial for the modelling results.

PCLake+ for the analysis in lake Enäjärvi has proven not to be sensitive to the assumed sediment parameters and the use of a static water depth. Adjusted values of these parameters to analyse the sensitivity also did not lead to a better fit of the model to the observational data. Based on an assessment of the historical water quality, the critical nutrient load, and several lake restoration measures. PCLake+ was used to determine the critical load in lake Enäjärvi for chlorophyll-a thresholds of 20, 30, 40, and 50 mg chlorophyll-a/m³. The critical load for the 20 and 30 mg chlorophyll-a/m³ thresholds are 0.29 and 0.37 mgP $m^{-2}d^{-1}$. These levels were exceeded 4 and 6 times in the period 2000-2017. The 40 and 50 mg thresholds were not reached in summer. All measures, except a reduction of the spring water level, resulted in a lower peak of the load response curve to under the 30 mg chlorophyll-a/m³ threshold, therefore the further mentioned results relate to the 20 mg chlorophyll-a per m³ threshold. Interesting, a reduction of the water level results in an increase of the load-response curve peak, likely due to the lack of alternative stable states, hampering domination of macrophytes. Adding a marsh area of 30% of the lake's surface to the lake resulted in a 48% increase of the 20 mg threshold critical nutrient load to 0.43 mgP m-2d-1. A marsh zone of 10% results in a 28% increase of this critical nutrient load, while a reduction of the wind fetch by 50% results in a 13% increase. Regular fish catch did not reduce the critical load. It should be noted that implementation of these measures is dependent on a lot more factors which are not researched in this thesis, results merely show their simulated effect on the load response curve and critical nutrient loading.

REFERENCES

Blöch, H. (1999). The european union water framework directive: Taking european water policy into the next millennium†. *Water Science and Technology, 40*(10), 67-71.

Blue adapt. (n.d.). Retrieved from https://blueadapt.fi/en/project-info/#tyopaketti1

Forbes Bulletin of the Scientific Association (Peoria, 11.), 77 (1887).

Stephanie Glen. "Relative Absolute Error" From **StatisticsHowTo.com**: Elementary Statistics for the rest of us! https://www.statisticshowto.com/relative-absolute-error/ (accessed august 2020)

Gulati, R. D., Dionisio Pires, L. M., & Van Donk, E. (2008). Lake restoration studies: failures, bottlenecks and prospects of new ecotechnological measures. *Limnologica*, *38*(3-4), 233–247.

Huttunen, I., Huttunen, M., Piirainen, V., Korppoo, M., Lepistö, A., Räike, A., . . . Vehviläinen, B. (2016). A national-scale nutrient loading model for finnish watersheds -VEMALA. *Environmental Modeling & Assessment, 21*(1), 83-109.

Janse, J. H. (1997). A model of nutrient dynamics in shallow lakes in relation to multiple stable states. *Hydrobiologia -The Hague-*, *342/343*(Com), 1–8.

Janse, J. (2005). *Model studies on the eutrophication of shallow lakes and ditches* (dissertation). s.n.

Janse, J., Domis, L., Scheffer, M., Lijklema, L., Liere, L., Klinge, M., & Mooij, W. (2008). Critical phosphorus loading of different types of shallow lakes and the consequences for management estimated with the ecosystem model pclake. *Limnologica 38 (2008) 3-Apr*. (2008).

Janse, J., Scheffer, M., Lijklema, L., Liere, L., Sloot, J., & Mooij, W. (2010). Estimating the critical phosphorus loading of shallow lakes with the ecosystem model pclake: Sensitivity, calibration and uncertainty. *Ecological Modelling 221 (2010) 4*. (2010).

Janssen, A., Jager, V., Janse, J., Kong, X., Liu, S., Ye, Q., & Mooij, W. (2017). Spatial identification of critical nutrient loads of large shallow lakes : Implications for lake taihu (china). *Water Research 119 (2017)*.

Janssen, A., Janse, J., Beusen, A., Chang, M., Harrison, J., Huttunen, I., . . . Mooij, W. (2019). How to model algal blooms in any lake on earth. *Current Opinion in Environmental Sustainability 36*

Janssen, A., Teurlincx, S., Beusen, A., Huijbregts, M., Rost, J., Schipper, A., . . . Janse, J. (2019a). Pclake : A process-based ecological model to assess the trophic state of stratified and non-stratified freshwater lakes worldwide. *Ecological Modelling, 396*(March), 23-32.

Kallio, K., Kutser, T., Hannonen, T., Koponen, S., Pulliainen, J., Vepsäläinen, J., & Pyhälahti, T. (2001). Retrieval of water quality from airborne imaging spectrometry of various lake types in different seasons. *Science of The Total Environment*, *268*(1-3), 59–77.

Kirillin, G., Hochschild, J., Mironov, D., Terzhevik, A., Golosov, S., & Nützmann, G. (2011). Flake-global: online lake model with worldwide coverage. *Environmental Modelling and Software*, *26*(5), 683–684.

Korppoo, M., Huttunen, M., Huttunen, I., Piirainen, V., & Vehviläinen, B. (2017). Simulation of bioavailable phosphorus and nitrogen loading in an agricultural river basin in finland using vemala v.3. *Journal of Hydrology, 549*, 363-373.

Korppoo, M. (2020) National scale nutrient/metals loading model - VEMALA

Lake & Sea Wiki, (2014). *Enäjärvi (22.005.1.001)*. Accessed at 20 january 2020, from http://www.jarviwiki.fi/wiki/En%C3%A4j%C3%A4rvi_(22.005.1.001)?setlang=en

Mooij, W., Janse, J., Senerpont Domis, L., H©ơlsmann, S., & Ibelings, B. (2007). Predicting the effect of climate change on temperate shallow lakes with the ecosystem model pclake. *Hydrobiologia -The Hague-, 584*(1), 443-454.

Mellios, N., Kofinas, D., Laspidou, C., & Papadimitriou, T. (2015). Mathematical modeling of trophic state and nutrient flows of lake karla using the pclake model. *Environmental Processes : An International Journal*, *2*(1), 85–100.

Nielsen, A., Trolle, D., Bjerring, R., Søndergaard, M., Olesen, J., Janse, J., . . . Jeppesen, E. (2014). Effects of climate and nutrient load on the water quality of shallow lakes assessed through ensemble runs by pclake. *Ecological Applications, 24*(8), 1926-1944

Pellikka, K. and Sammalkorpi, I. (2020). *Lake Enäjärvi*. (presentation)

Scheffer, M., Hosper, S. H., Meijer, M.-L., Moss, B., & Jeppesen, E. (1993). Alternative equilibria in shallow lakes. *Trends in Ecology & Evolution*, *8*(8), 275–279. https://doi.org/10.1016/0169-5347(93)90254-M

Scheffer, M. (2009). *Critical transitions in nature and society* (Princeton studies in complexity). Princeton, N.J.: Princeton University Press.

Schep, S.A., Ter Heerdt, G., Janse, J., Ouboter, M. R. L. (2007) Possible effects of climate change on ecological functioning of shallow lakes, Lake Loenderveen as a case study.

Sollie, S., Janse, J. H., Mooij, W. M., Coops, H., & Verhoeven, J. T. A. (2008). The contribution of marsh zones to water quality in dutch shallow lakes: a modeling study. *Environmental Management*, *42*(6),

Søndergaard, M., Christian, S., Egbert, H. V. N., Rudi, R., Eddy, L., Jeppesen, E., … Portielje, R. (2007). Lake restoration: successes, failures and long-term effects. *Journal of Applied Ecology*, *44*(6), 1095–1105.

Stowa. (n.d.) Een ecologische systeemanalyse en diagnose van ondiepe meren en plassen voor de kader richtlijn water.

SYKE. (n.d.). Retrieved January 30, 2020, from http://paikkatieto.ymparisto.fi/vesikarttaviewers/Html5Viewer_2_11_2/Index.html?configBase=http://paikkatieto.ymparisto.fi/Geocortex/Essentials/REST/sites/VesikarttaKansa/viewers/VesikarttaHTML525/virtualdirectory/Resources/Config/Default&locale=fi-FI

Finnish Environment Institute (SYKE) (2018) / Geoinformatics systems and Geoinformatics research units have produced these Remote Sensing products. Satellite products are based on several NASA, NOAA and EU Copernicus program satellite instruments. Currently Sentinel-2 (ESA), Sentinel-3 (ESA), Landsat-8 (NASA)and TERRA MODIS (NASA) satellites are utilized for monitoring. Previously also NOAA-AVHRR (NOAA), AQUA/TERRA MODIS (NASA), ENVISAT MERIS (ESA), RADARSAT (CSA) and AMSR-E (NASA) were utilized. ENVISAT MERIS and Sentinel-2 images have been downloaded from ESA. Landsat-8 and a subset of the AQUA MODIS images have been downloaded from NASA. NOAA-AVHRR and TERRA / AQUA MODIS images have been received by Finnish Meteorological Institute.

Tatrai, I., Toth, L.-G., Istvanovics, V., & Kobor, I. (2008). Management measures and long-term water quality changes in lake balaton (hungary). *Fundamental and Applied Limnology*, *172*(1), 1–11.

Tuominen, M., Schultz, E., & Sillanpää, M. (2013). Toxicity and stability of silver nanoparticles to the green alga Pseudokirchneriella subcapitata in boreal freshwater samples and growth media. *Nanomaterials and the Environment*, *1*.

Yang, X., Wu, X., Hao, H., & He, Z. (2008). Mechanisms and assessment of water eutrophication. *Journal of Zhejiang University Science B : An International Biomedicine & Biotechnology Journal, 9*(3), 197-209.

APPENDIX

A. INITIAL STATE VARIABLES

Initial state name	Unit	Description	turbid	clear
cDepthW0	_m_	Depth_of_hypolimnion_lake_water	3.22	3.22
cNH4WHyp0	_gN_*_m_^-3	N_in_NH4_in_hypolimnion_lake_water	0.001137	1.89E-05
cNO3WHyp0	_gN_*_m_^-3	N_in_NO3_in_hypolimnion_lake_water	0.01274	0.000149
cPO4WHyp0	_gP_*_m_^-3	P_in_PO4_in_hypolimnion_lake_water	0.002635	1.28E-05
cPAIMWHyp0	_gP_*_m_^-3	P_adsorbed_onto_IM_in_hypolimnion_lake_water	2.15E-06	1.13E-07
cSiO2WHyp0	_gSi_*_m_^-3	Dissolved_Si_in_hypolimnion_lake_water	0.020703	0.010493
cO2WHyp0	_gO2_*_m_^-3	Oxygen_in_hypolimnion_lake_water	0.021481	0.036202
cDDetWHyp0	_gDW_*_m_^-3	Detritus_DW_in_hypolimnion_lake_water	0.019152	0.000761
cNDetWHyp0	_gN_*_m_^-3	Detritus_N_in_hypolimnion_lake_water	0.001304	3.95E-05
cPDetWHyp0	_gP_*_m_^-3	Detritus_P_in_hypolimnion_lake_water	0.00019	6.41E-06
cSiDetWHyp0	_gSi_*_m_^-3	Detritus_Si_in_hypolimnion_lake_water	0.001046	5.86E-05
cDIMWHyp0	_gDW_*_m_^-3	Inorganic_matter_in_hypolimnion_lake_water	0.006292	0.005841
cDDiatWHyp0	_gDW_*_m_^-3	Diatoms_DW_in_hypolimnion_lake_water	1.92E-05	1.06E-06
cNDiatWHyp0	_gN_*_m_^-3	Diatoms_N_in_hypolimnion_lake_water	9.59E-07	5.23E-08
cPDiatWHyp0	_gP_*_m_^-3	Diatoms_P_in_hypolimnion_lake_water	1.12E-07	5.24E-09
cDGrenWHyp0	_gDW_*_m_^-3	Green_algae_DW_in_hypolimnion_lake_water	5.15E-05	1.25E-06
cNGrenWHyp0	_gN_*_m_^-3	Green_algae_N_in_hypolimnion_lake_water	5.11E-06	1.22E-07
cPGrenWHyp0	_gP_*_m_^-3	Green_algae_P_in_hypolimnion_lake_water	7.65E-07	1.77E-08
cDBlueWHyp0	_gDW_*_m_^-3	Blue-greens_DW_in_hypolimnion_lake_water	0.001303	0.000382
cNBlueWHyp0	_gN_*_m_^-3	Blue-greens_N_in_hypolimnion_lake_water	0.000193	5.42E-05

cPBlueWHyp0	_gP_*_m_^-3	Blue-greens_P_in_hypolimnion_lake_water	3.26E-05	9.44E-06
cDZooHyp0	_gDW_*_m_^-3	Zooplankton_DW_in_hypolimnion_lake_water	0.000136	3.94E-06
cNZooHyp0	_gN_*_m_^-3	Zooplankton_N_in_hypolimnion_lake_water	9.54E-06	2.76E-07
cPZooHyp0	_gP_*_m_^-3	Zooplankton_P_in_hypolimnion_lake_water	1.36E-06	3.94E-08
cDFiAd0	_gDW_*_m_^-2	Adult_fish_DW_in_lake_water	2.778	2.64434
cDFiJv0	_gDW_*_m_^-2	Young_fish_DW_in_lake_water	1.30097	1.1248
cNFiAd0	_gN_*_m_^-2	Adult_fish_N_in_lake_water	0.278155	0.264884
cNFiJv0	_gN_*_m_^-2	Young_fish_N_in_lake_water	0.130584	0.11309
cPFiAd0	_gP_*_m_^-2	Adult_fish_P_in_lake_water	0.061063	0.058097
cPFiJv0	_gP_*_m_^-2	Young_fish_P_in_lake_water	0.028545	0.024632
cDPisc0	_gDW_*_m_^-2	Predatory_fish_DW_in_lake_water	0.000353	0.000353
cNH4S0	_gN_*_m_^-2	N_in_NH4_in_lake_sediment_pore_water	0.069499	0.068657
cNO3S0	_gN_*_m_^-2	N_in_NO3_in_lake_sediment_pore_water	0.051048	0.051752
cPO4S0	_gP_*_m_^-2	P_in_PO4_in_lake_sediment_pore_water	0.019443	0.015519
cPAIMS0	_gP_*_m_^-2	P_adsorbed_onto_IM_in_lake_sediment	6.20493	6.20076
cDDetS0	_gDW_*_m_^-2	Detritus_DW_in_lake_sediment	338.531	328.219
cNDetS0	_gN_*_m_^-2	Detritus_N_in_lake_sediment	12.387	12.5518
cPDetS0	_gP_*_m_^-2	Detritus_P_in_lake_sediment	1.15792	1.0433
cSiDetS0	_gSi_*_m_^-2	Detritus_Si_in_lake_sediment	56.0346	53.6103
cDHumS0	_gDW_*_m_^-2	Humus_DW_in_lake_sediment	7065.72	3771.08
cNHumS0	_gN_*_m_^-2	Humus_N_in_lake_sediment	317.669	111.353
cPHumS0	_gP_*_m_^-2	Humus_P_in_lake_sediment	41.1241	10.8925
cDIMS0	_gDW_*_m_^-2	Inorganic_matter_in_lake_sediment	25974.2	31875.6

cDDiatS0	_gDW_*_m_^-2	Diatoms_DW_on_lake_sediment	0.017677	0.011136
cNDiatS0	_gN_*_m_^-2	Diatoms_N_on_lake_sediment	0.000849	0.000537
cPDiatS0	_gP_*_m_^-2	Diatoms_P_on_lake_sediment	8.28E-05	4.02E-05
cDGrenS0	_gDW_*_m_^-2	Green_algae_DW_on_lake_sediment	0.005064	0.004992
cNGrenS0	_gN_*_m_^-2	Green_algae_N_on_lake_sediment	0.000447	0.000446
cPGrenS0	_gP_*_m_^-2	Green_algae_P_on_lake_sediment	5.66E-05	3.77E-05
cDBlueS0	_gDW_*_m_^-2	Blue-greens_DW_on_lake_sediment	0.19416	0.207206
cNBlueS0	_gN_*_m_^-2	Blue-greens_N_on_lake_sediment	0.025861	0.027588
cPBlueS0	_gP_*_m_^-2	Blue-greens_P_on_lake_sediment	0.004653	0.004784
cDVeg0	_gDW_*_m_^-2	Vegetation_DW_in_lake_water	0.06061	0.060685
cNVeg0	_gN_*_m_^-2	Vegetation_N_in_lake_water	0.002114	0.002117
cPVeg0	_gP_*_m_^-2	Vegetation_P_in_lake_water	0.000211	0.000212
cVegHeight0	_m_	Height_of_vegetation	2	2
cDBent0	_gDW_*_m_^-2	Zoobenthos_DW_in_lake_sediment	2.50668	2.43043
cNBent0	_gN_*_m_^-2	Zoobenthos_N_in_lake_sediment	0.175468	0.170475
cPBent0	_gP_*_m_^-2	Zoobenthos_P_in_lake_sediment	0.025067	0.024019
cDepthWM0	_m_	Depth_of_marsh_water	0.5	0.5
cNH4WM0	_gN_*_m_^-3	N_in_NH4_in_marsh_water	0.015528	0.015528
cNO3WM0	_gN_*_m_^-3	N_in_NO3_in_marsh_water	0.015528	0.015528
cPO4WM0	_gP_*_m_^-3	P_in_PO4_in_marsh_water	0.001553	0.001553
cPAIMWM0	_gP_*_m_^-3	P_adsorbed_onto_IM_in_marsh_water	0	0
cSiO2WM0	_gSi_*_m_^-3	Dissolved_Si_in_marsh_water	0.465839	0.465839
_cO2WM0_	_gO2_*_m_^-3	Oxygen_in_marsh_water	1.552795	1.552795

cDDetWM0	_gDW_*_m_^-3	Detritus_DW_in_marsh_water	0.310559	0.310559
cNDetWM0	_gN_*_m_^-3	Detritus_N_in_marsh_water	0.007764	0.007764
cPDetWM0	_gP_*_m_^-3	Detritus_P_in_marsh_water	0.000776	0.000776
cSiDetWM0	_gSi_*_m_^-3	Detritus_Si_in_marsh_water	0.003106	0.003106
cDIMWM0	_gDW_*_m_^-3	Inorganic_matter_in_marsh_water	0.776398	0.597618
cDDiatWM0	_gDW_*_m_^-3	Diatoms_DW_in_marsh_water	0.07764	0.07764
cNDiatWM0	_gN_*_m_^-3	Diatoms_N_in_marsh_water	0.007764	0.007764
cPDiatWM0	_gP_*_m_^-3	Diatoms_P_in_marsh_water	0.000776	0.000776
cDGrenWM0	_gDW_*_m_^-3	Green_algae_DW_in_marsh_water	0.07764	0.07764
cNGrenWM0	_gN_*_m_^-3	Green_algae_N_in_marsh_water	0.007764	0.007764
cPGrenWM0	_gP_*_m_^-3	Green_algae_P_in_marsh_water	0.000776	0.000776
cDBlueWM0	_gDW_*_m_^-3	Blue-greens_DW_in_marsh_water	0.465839	0.465839
cNBlueWM0	_gN_*_m_^-3	Blue-greens_N_in_marsh_water	0.046584	0.046584
cPBlueWM0	_gP_*_m_^-3	Blue-greens_P_in_marsh_water	0.004658	0.004658
cDZooM0	_gDW_*_m_^-3	Zooplankton_DW_in_marsh_water	0.007764	0.007764
cNZooM0	_gN_*_m_^-3	Zooplankton_N_in_marsh_water	0.000543	0.000543
cPZooM0	_gP_*_m_^-3	Zooplankton_P_in_marsh_water	7.76E-05	7.76E-05
cNH4SM0	_gN_*_m_^-2	N_in_NH4_in_marsh_sediment_pore_water	1	1
cNO3SM0	_gN_*_m_^-2	N_in_NO3_in_marsh_sediment_pore_water	0.01	0.01
cPO4SM0	_gP_*_m_^-2	P_in_PO4_in_marsh_sediment_pore_water	0.181703	0.181703
cPAIMSM0	_gP_*_m_^-2	P_adsorbed_onto_IM_in_marsh_sediment	17.9886	17.9886
cDDetSM0	_gDW_*_m_^-2	Detritus_DW_in_marsh_sediment	181.703	181.703
cNDetSM0	_gN_*_m_^-2	Detritus_N_in_marsh_sediment	4.54258	4.54258

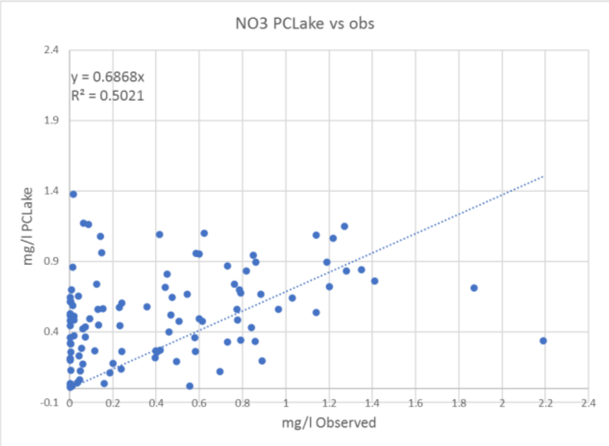
cPDetSM0	_gP_*_m_^-2	Detritus_P_in_marsh_sediment	0.454258	0.454258
cSiDetSM0	_gSi_*_m_^-2	Detritus_Si_in_marsh_sediment	1.81703	1.81703
cDHumSM0	_gDW_*_m_^-2	Humus_DW_in_marsh_sediment	3452.36	3452.36
cNHumSM0	_gN_*_m_^-2	Humus_N_in_marsh_sediment	172.618	172.618
cPHumSM0	_gP_*_m_^-2	Humus_Pin_marsh_sediment	17.2618	17.2618
cDIMSM0	_gDW_*_m_^-2	Inorganic_matter_in_marsh_sediment	32706.5	32706.5
cDRootPhra0	_gDW_*_m_^-2	Root_biomass_DW_in_marsh_sediment	5000	5000
cDShootPhra0	_gDW_*_m_^-2	Shoot_biomass_DW_in_marsh_water	1000	1000
cNRootPhra0	_gN_*_m_^-2	Root_biomass_N_in_marsh_sediment	100	100
cNShootPhra0	_gN_*_m_^-2	Shoot_biomass_N_in_marsh_water	20	20
cPRootPhra0	_gP_*_m_^-2	Root_biomass_P_in_marsh_sediment	10	10
cPShootPhra0	_gP_*_m_^-2	Shoot_biomass_P_in_marsh_water	2	2
cNH4WEpi0	_gN_*_m_^-3	N_in_NH4_in_epilimnion_lake_water	0.097018	0.092627
cNO3WEpi0	_gN_*_m_^-3	N_in_NO3_in_epilimnion_lake_water	0.452317	0.452165
cPO4WEpi0	_gP_*_m_^-3	P_in_PO4_in_epilimnion_lake_water	0.008775	0.00191
cPAIMWEpi0	_gP_*_m_^-3	P_adsorbed_onto_IM_in_epilimnion_lake_water	3.03E-05	2.38E-05
cSiO2WEpi0	_gSi_*_m_^-3	Dissolved_Si_in_epilimnion_lake_water	2.28855	2.355978
cO2WEpi0	_gO2_*_m_^-3	Oxygen_in_epilimnion_lake_water	14.32854	14.34022
cDDetWEpi0	_gDW_*_m_^-3	Detritus_DW_in_epilimnion_lake_water	0.408981	0.389748
cNDetWEpi0	_gN_*_m_^-3	Detritus_N_in_epilimnion_lake_water	0.030929	0.031062
cPDetWEpi0	_gP_*_m_^-3	Detritus_P_in_epilimnion_lake_water	0.004661	0.004352
cSiDetWEpi0	_gSi_*_m_^-3	Detritus_Si_in_epilimnion_lake_water	0.022194	0.017753
cDIMWEpi0	_gDW_*_m_^-3	Inorganic_matter_in_epilimnion_lake_water	0.19149	0.190208

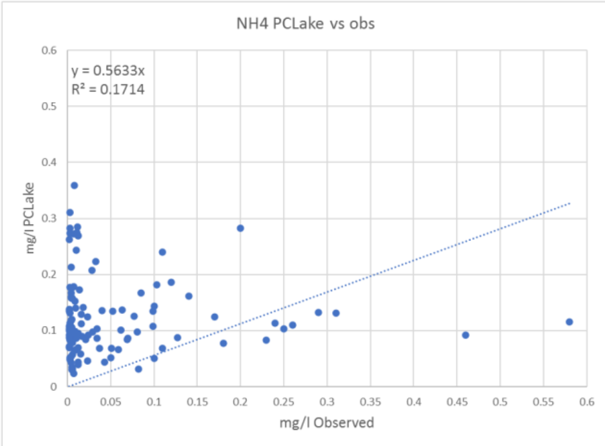
cDDiatWEpi0	_gDW_*_m_^-3	Diatoms_DW_in_epilimnion_lake_water	0.00169	0.001024
cNDiatWEpi0	_gN_*_m_^-3	Diatoms_N_in_epilimnion_lake_water	7.82E-05	4.75E-05
cPDiatWEpi0	_gP_*_m_^-3	Diatoms_P_in_epilimnion_lake_water	7.83E-06	3.6E-06
cDGrenWEpi0	_gDW_*_m_^-3	Green_algae_DW_in_epilimnion_lake_water	0.00148	0.001395
cNGrenWEpi0	_gN_*_m_^-3	Green_algae_N_in_epilimnion_lake_water	0.000126	0.00012
cPGrenWEpi0	_gP_*_m_^-3	Green_algae_P_in_epilimnion_lake_water	1.72E-05	1E-05
cDBlueWEpi0	_gDW_*_m_^-3	Blue-greens_DW_in_epilimnion_lake_water	0.758043	0.809447
cNBlueWEpi0	_gN_*_m_^-3	Blue-greens_N_in_epilimnion_lake_water	0.101001	0.107781
cPBlueWEpi0	_gP_*_m_^-3	Blue-greens_P_in_epilimnion_lake_water	0.018185	0.019034
cDZooEpi0	_gDW_*_m_^-3	Zooplankton_DW_in_epilimnion_lake_water	0.024649	0.02128
cNZooEpi0	_gN_*_m_^-3	Zooplankton_N_in_epilimnion_lake_water	0.001725	0.00149
cPZooEpi0	_gP_*_m_^-3	Zooplankton_P_in_epilimnion_lake_water	0.000246	0.000213
cDExtTotT0	_gDW_*_m_^-2	Total_amount_of_DW_moved_into_or_out_from_the_system	33389.8	35985.9
cNExtTotT0	_gN_*_m_^-2	Total_amount_of_N_moved_into_or_out_from_the_system	332.99	126.811
cPExtTotT0	_gP_*_m_^-2	Total_amount_of_P_moved_into_or_out_from_the_system	48.7289	18.3462
cSiExtTotT0	_gSi_*_m_^-2	Total_amount_of_Si_moved_into_or_out_from_the_system	63.4786	61.2559
cO2ExtTotT0	_gO2_*m_^-2	Total_amount_of_O2_moved_into_or_out_from_the_system	46.1379	46.1755

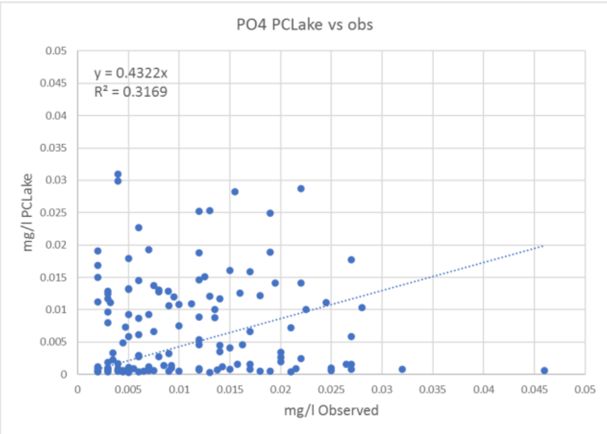
B. MODEL FIT (REGRESSION)

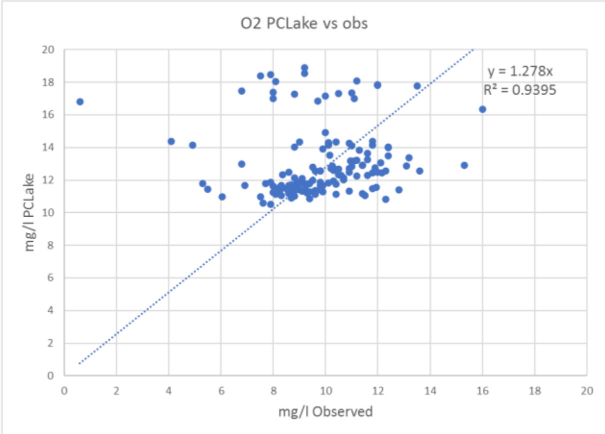
The model output data (y-axis) is compared to the observations (x-axis) and plotted against each other. The trendline giving the R^2 is fitted though the origin, this R^2 is not used for the model performance but is merely and indication how well the points fit on the regression line.

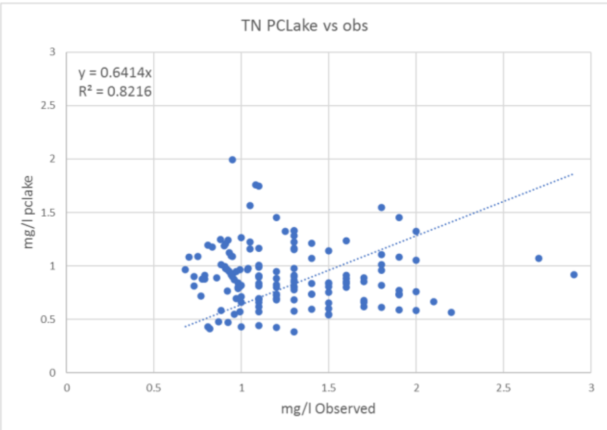
B.1. PCLAKE+ VS. OBSERVATIONS

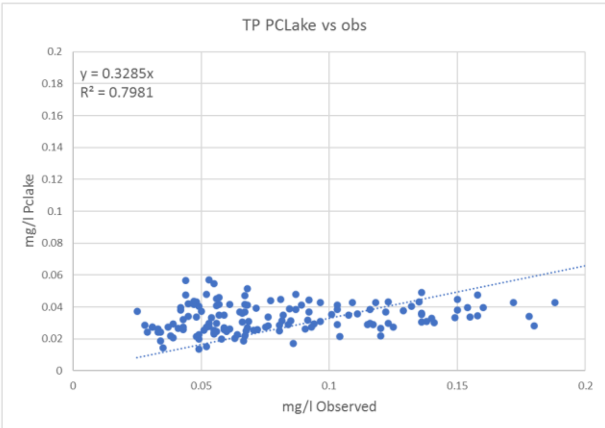

NO3 PCLake vs obs
y = 0.6868x
R^2 = 0.5021
mg/l PCLake
mg/l Observed

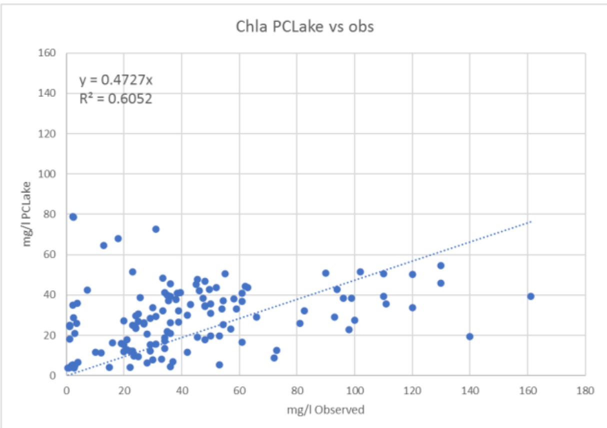

NH4 PCLake vs obs
y = 0.5633x
R^2 = 0.1714
mg/l PCLake
mg/l Observed


PO4 PCLake vs obs
y = 0.4322x
R^2 = 0.3169
mg/l PCLake
mg/l Observed

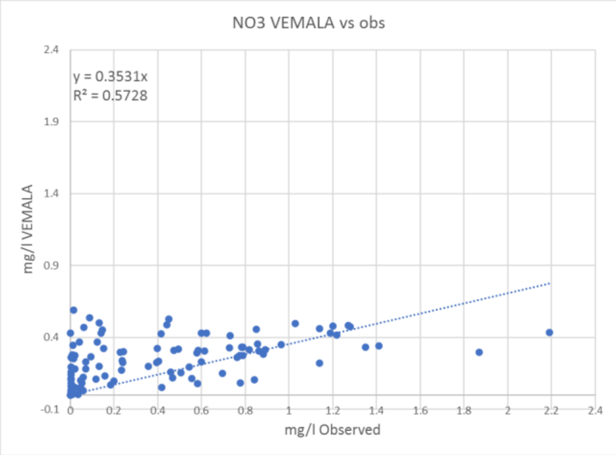

O2 PCLake vs obs
y = 1.278x
R^2 = 0.9395
mg/l PCLake
mg/l Observed

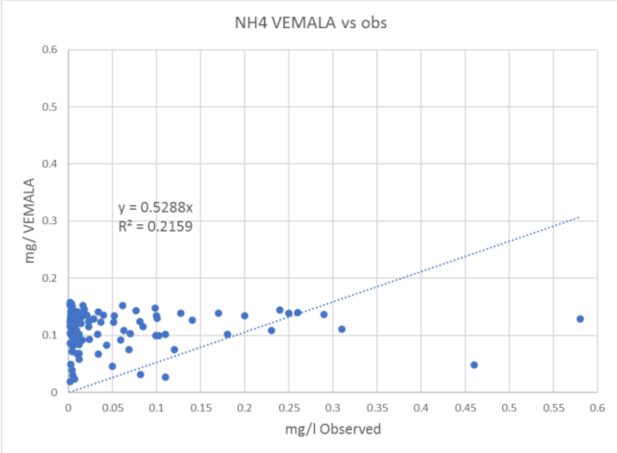

TN PCLake vs obs
y = 0.6414x
R^2 = 0.8216
mg/l pclake
mg/l Observed

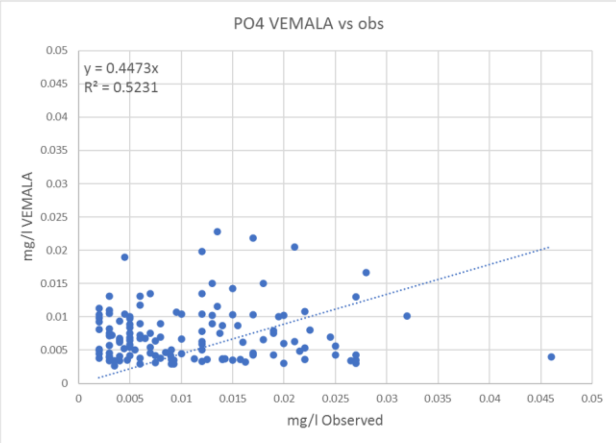

TP PCLake vs obs
y = 0.3285x
R^2 = 0.7981
mg/l Pclake
mg/l Observed

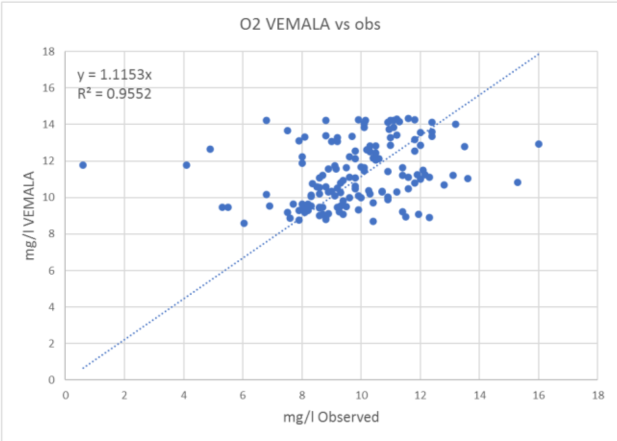

Chla PCLake vs obs
y = 0.4727x
R^2 = 0.6052
mg/l PCLake
mg/l Observed

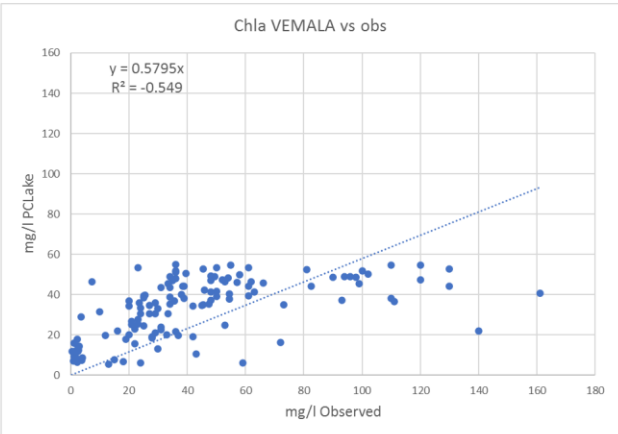
B.2. VEMALA V.3 VS. OBSERVATIONS



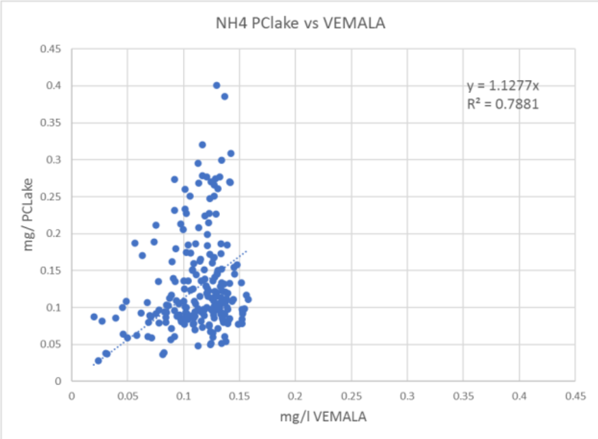


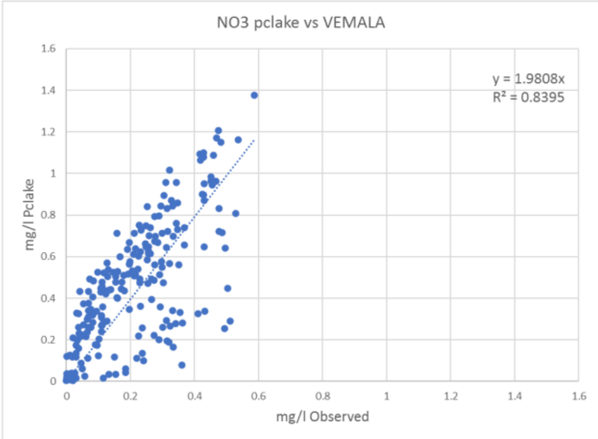


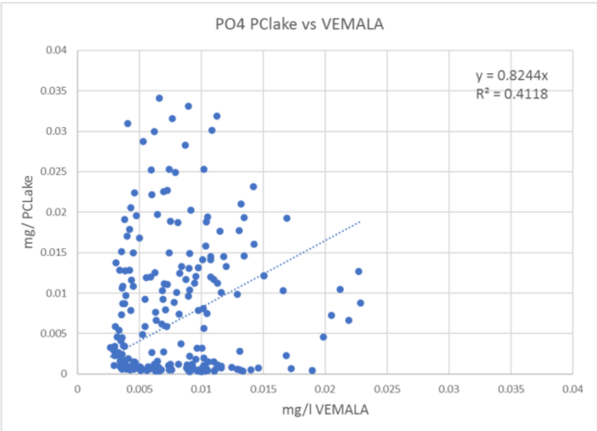


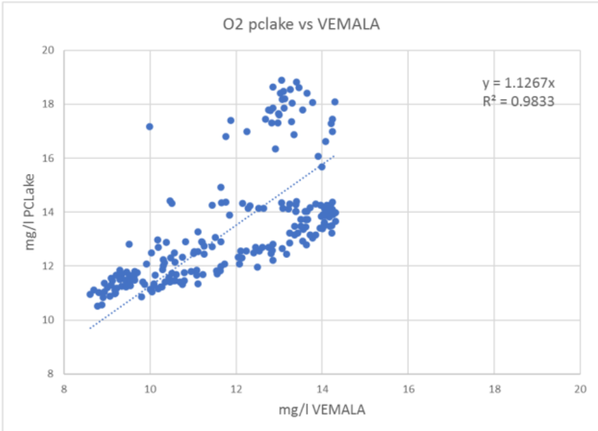


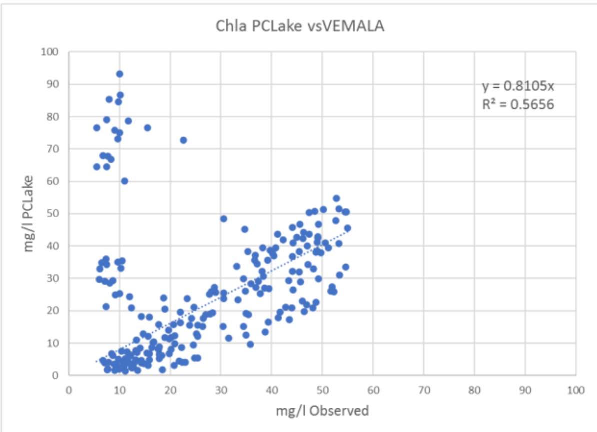
B.3. PCLAKE+ VS VEMALA V.3

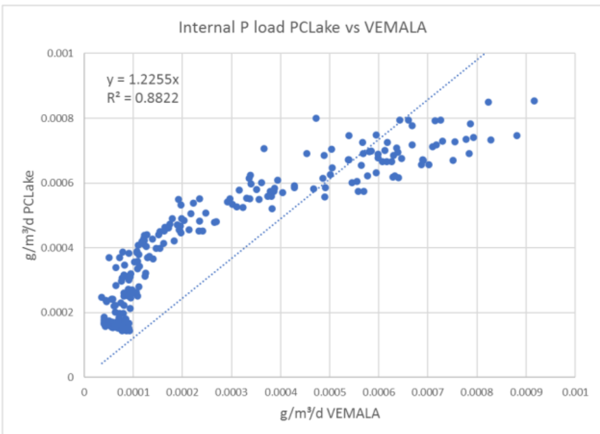








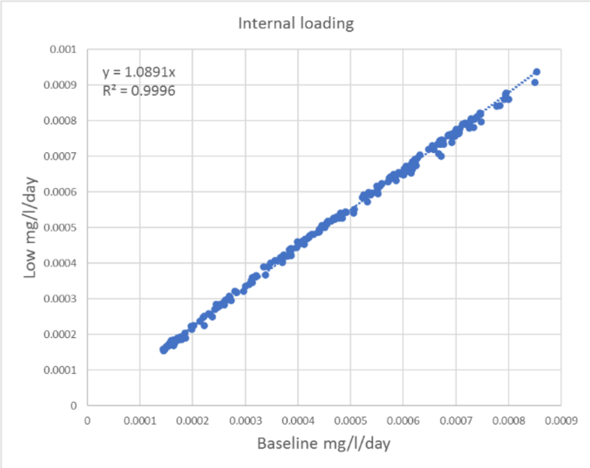


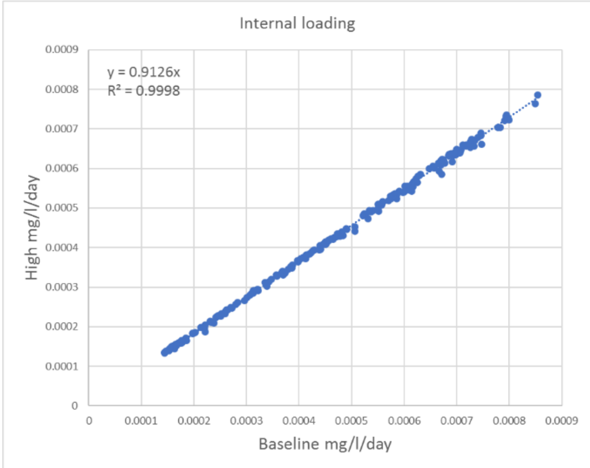


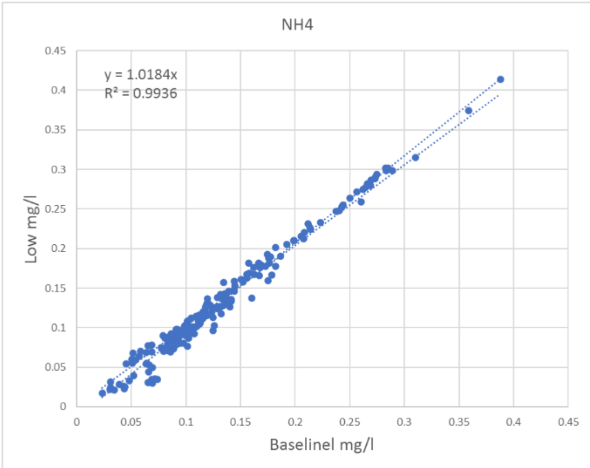
C. SENSITIVITY ANALYSIS (TRANSIENT)

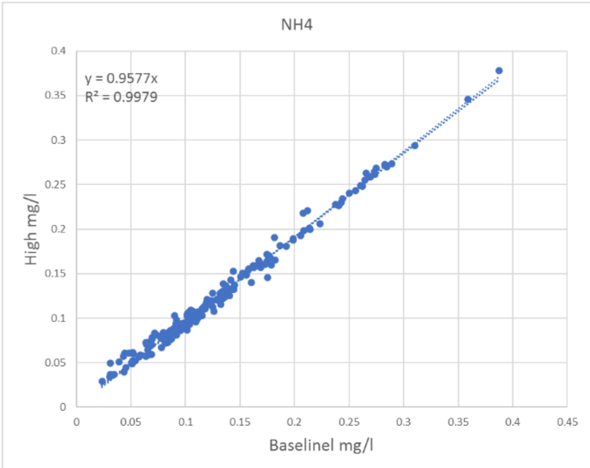
The PCLake+ output with adjusted parameter (y-axis) is plotted against the baseline PCLake+ output adjusted parameter (x-axis)

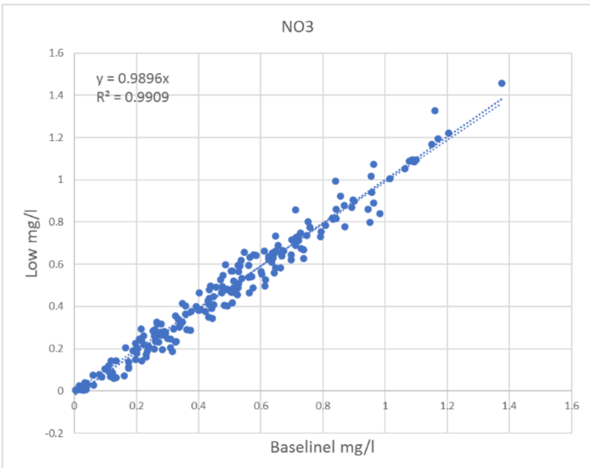
C.1. WATER TABLE SENSITIVITY (HIGHER AND LOWER WATER TABLE)

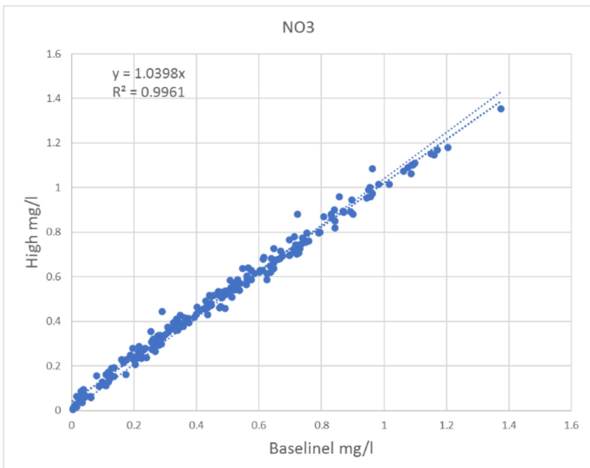

Internal loading
y = 1.0891x
R² = 0.9996
Low mg/l/day
Baseline mg/l/day

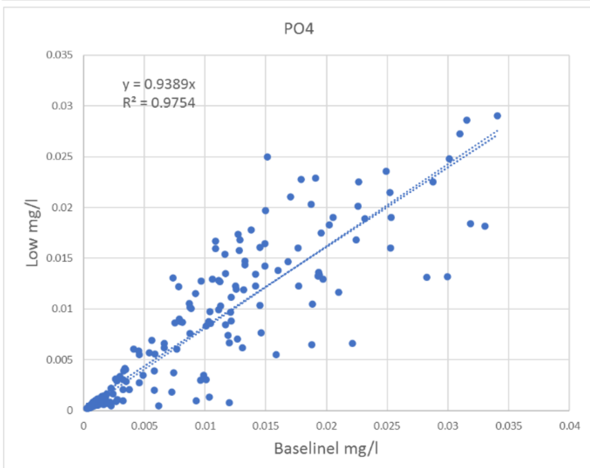

Internal loading
y = 0.9126x
R² = 0.9998
High mg/l/day
Baseline mg/l/day

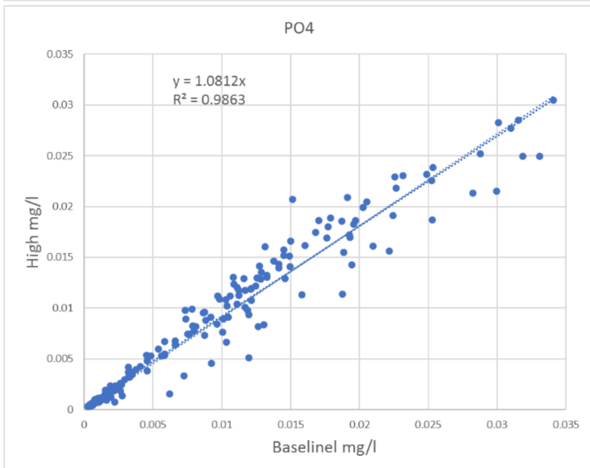

NH4
y = 1.0184x
R² = 0.9936
Low mg/l
Baselinel mg/l

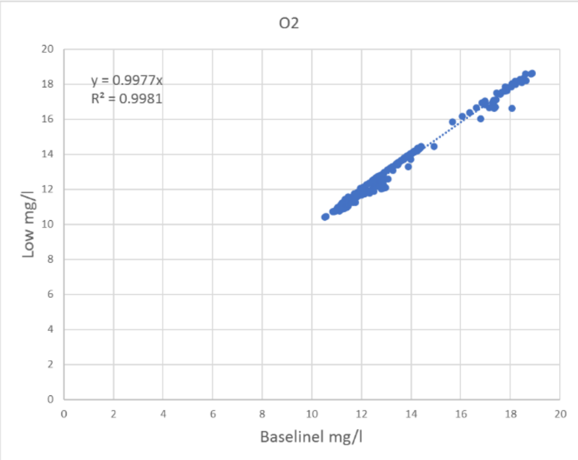

NH4
y = 0.9577x
R² = 0.9979
High mg/l
Baselinel mg/l

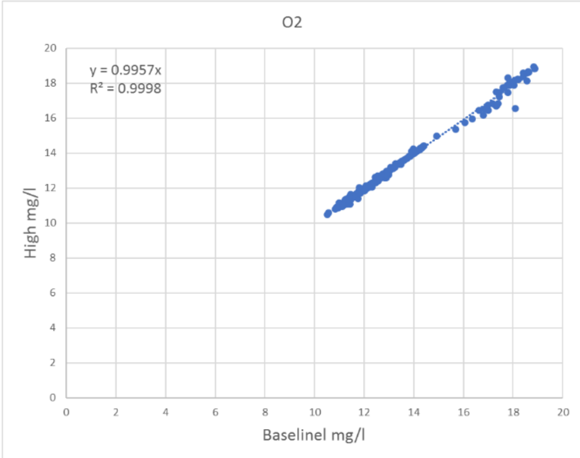

NO3
y = 0.9896x
R² = 0.9909
Low mg/l
Baselinel mg/l

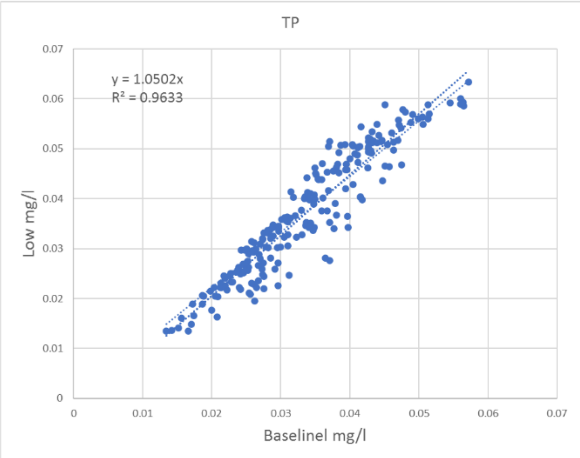

NO3
y = 1.0398x
R² = 0.9961
High mg/l
Baselinel mg/l

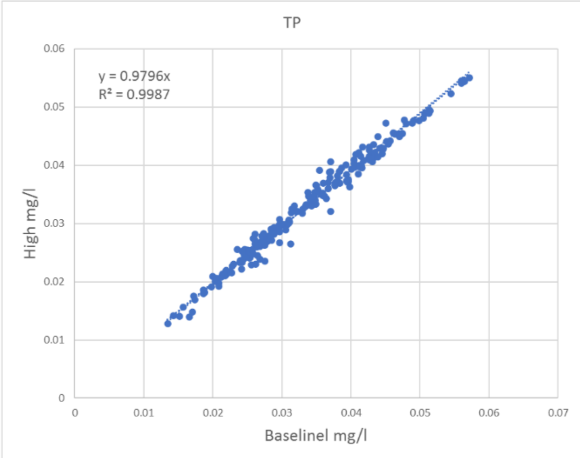

PO4
y = 0.9389x
R² = 0.9754
Low mg/l
Baselinel mg/l

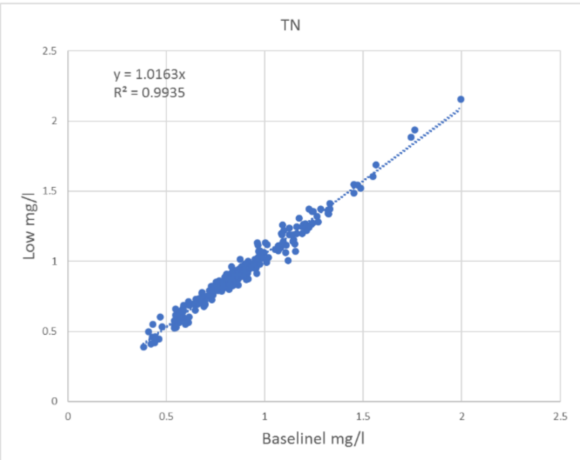

PO4
y = 1.0812x
R² = 0.9863
High mg/l
Baselinel mg/l

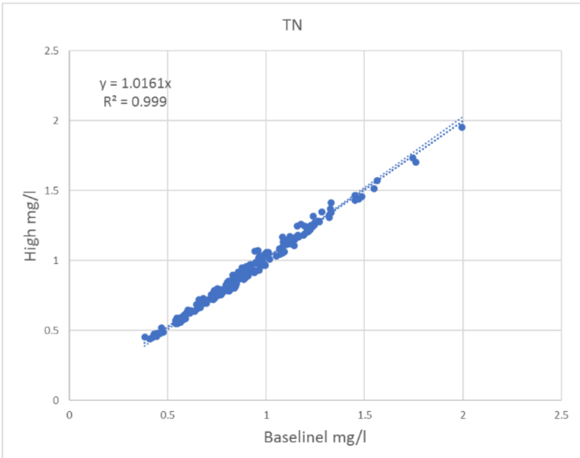

O2
y = 0.9977x
R² = 0.9981
Low mg/l
Baselinel mg/l

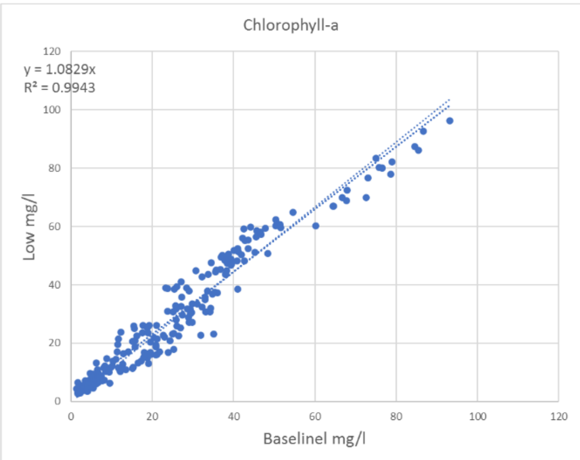

O2
y = 0.9957x
R² = 0.9998
High mg/l
Baselinel mg/l

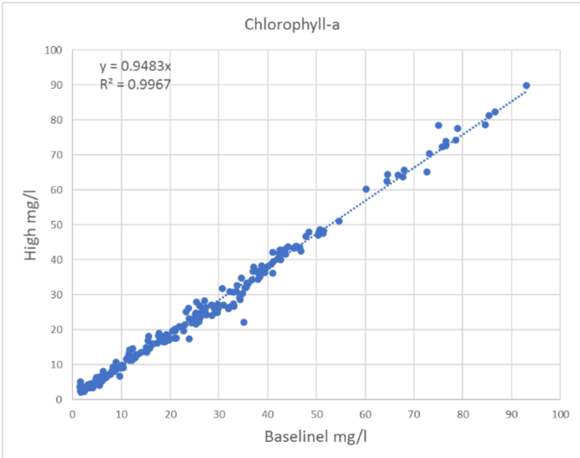

TP
y = 1.0502x
R² = 0.9633
Low mg/l
Baselinel mg/l


TP
y = 0.9796x
R² = 0.9987
High mg/l
Baselinel mg/l


TN
y = 1.0163x
R² = 0.9935
Low mg/l
Baselinel mg/l

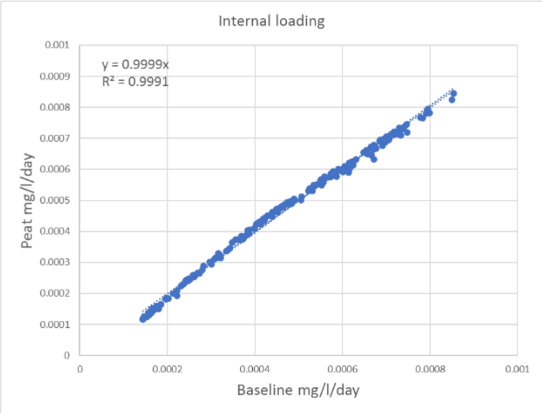

TN
y = 1.0161x
R² = 0.999
High mg/l
Baselinel mg/l

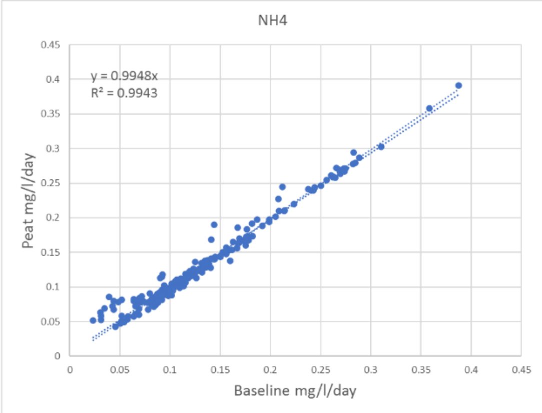

Chlorophyll-a
y = 1.0829x
R² = 0.9943
Low mg/l
Baselinel mg/l

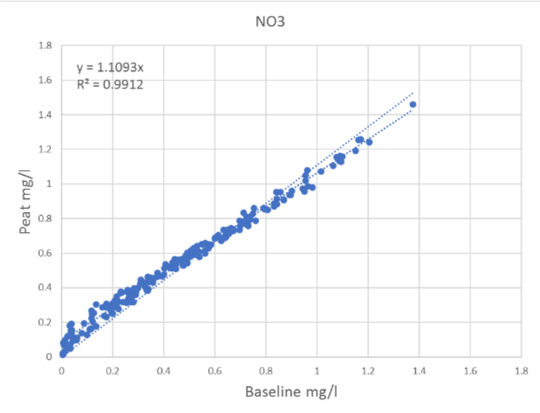

Chlorophyll-a
y = 0.9483x
R² = 0.9967
High mg/l
Baselinel mg/l

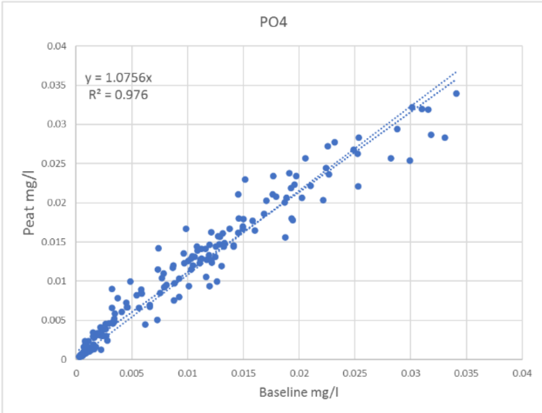
C.2. SEDIMENT SENSITIVITY

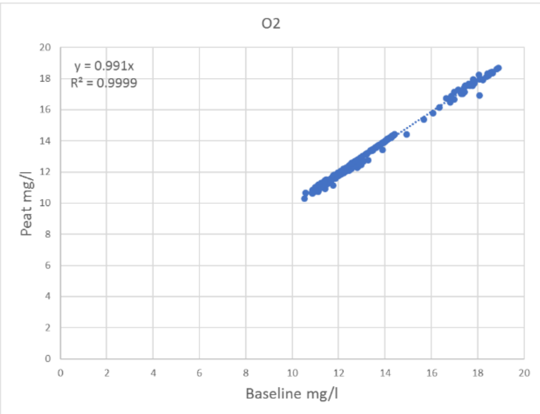
C.2.A. PEAT

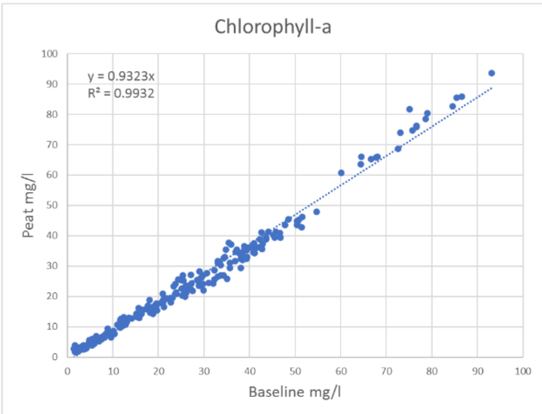


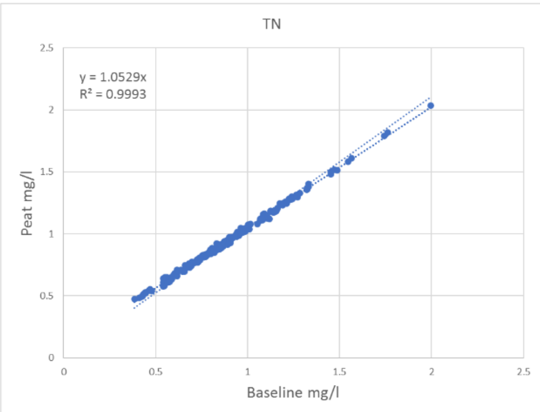


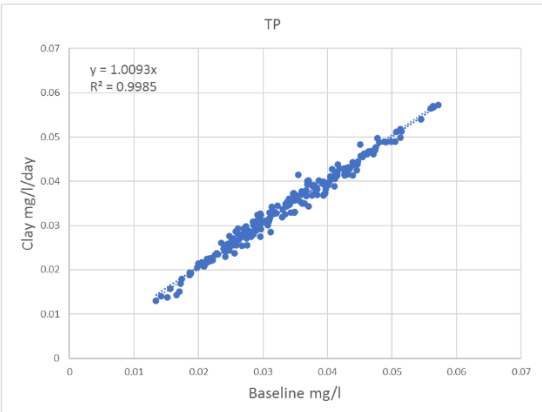




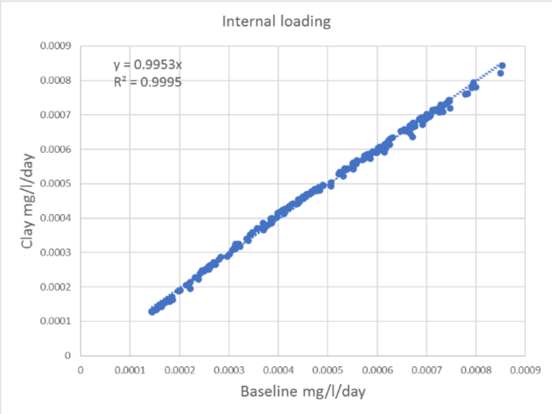


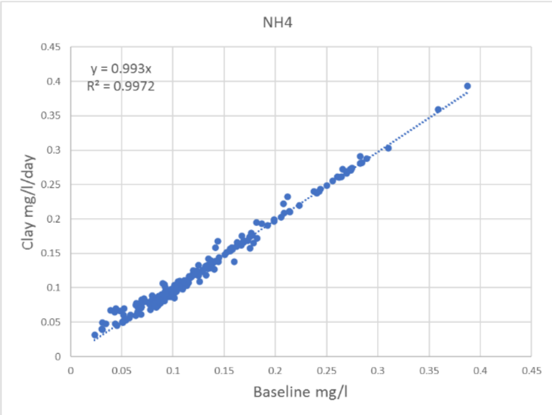


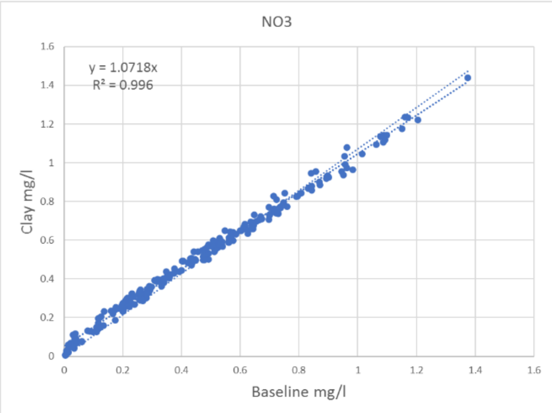


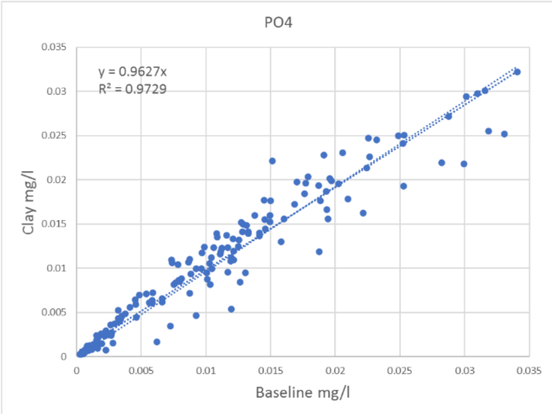


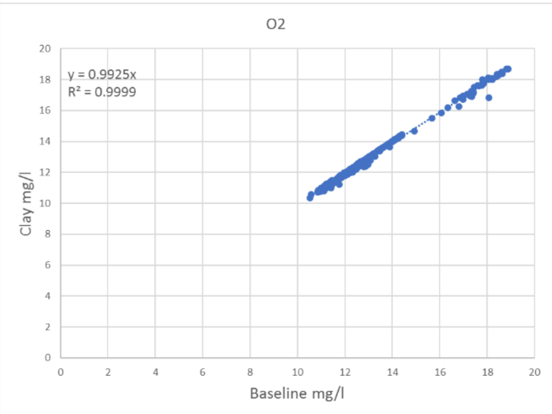
C.2.B. CLAY

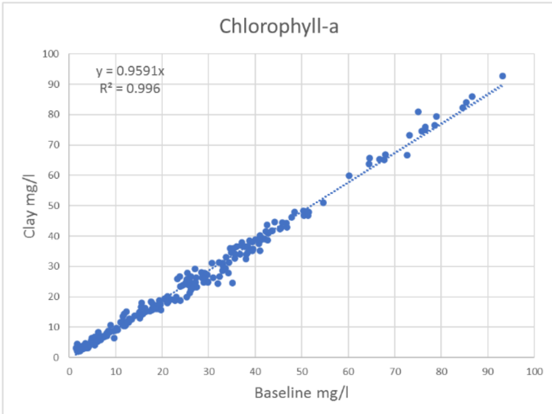

Internal loading
y = 0.9953x
R² = 0.9995
Clay mg/l/day
Baseline mg/l/day

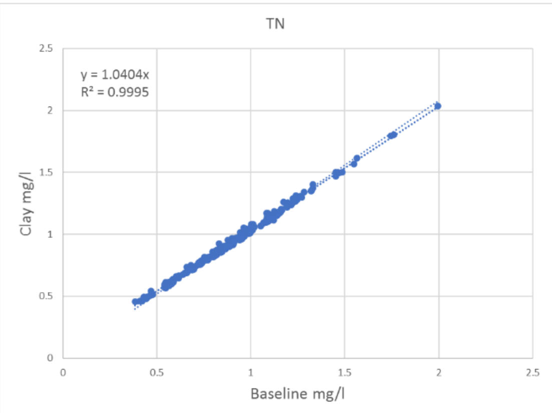

NH4
y = 0.993x
R² = 0.9972
Clay mg/l/day
Baseline mg/l

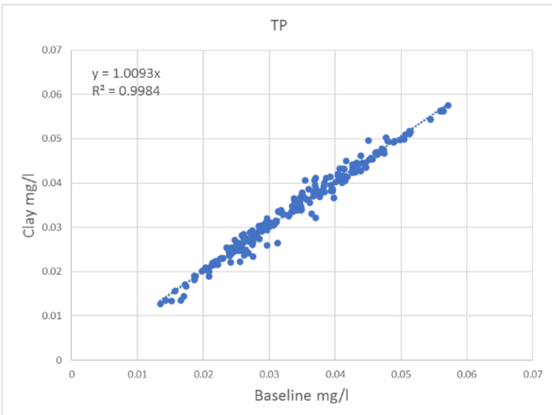

NO3
y = 1.0718x
R² = 0.996
Clay mg/l
Baseline mg/l


PO4
y = 0.9627x
R² = 0.9729
Clay mg/l
Baseline mg/l

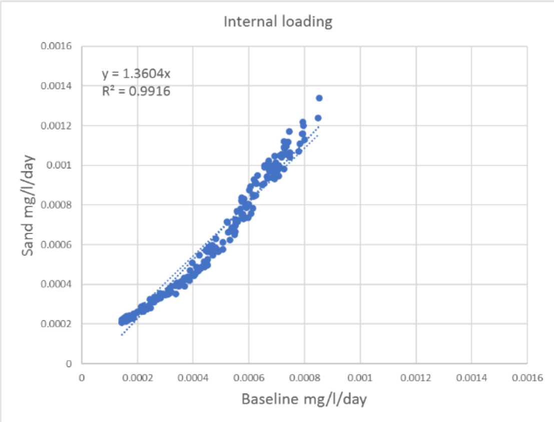

O2
y = 0.9925x
R² = 0.9999
Clay mg/l
Baseline mg/l


Chlorophyll-a
y = 0.9591x
R² = 0.996
Clay mg/l
Baseline mg/l

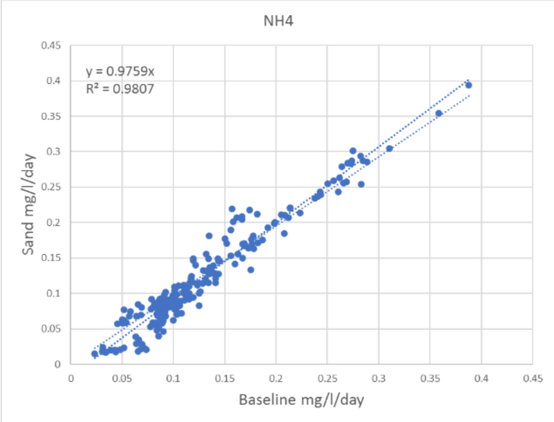

TN
y = 1.0404x
R² = 0.9995
Clay mg/l
Baseline mg/l


TP
y = 1.0093x
R² = 0.9984
Clay mg/l
Baseline mg/l

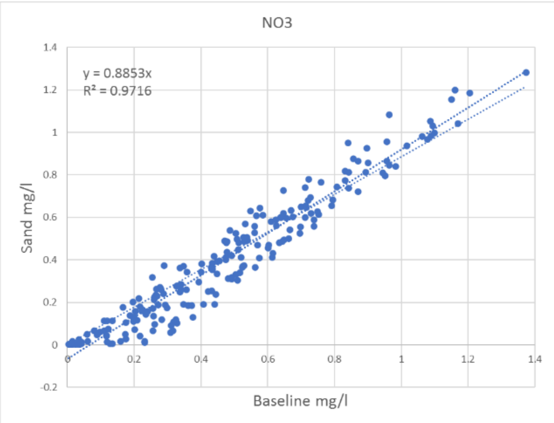
C.2.C. SAND



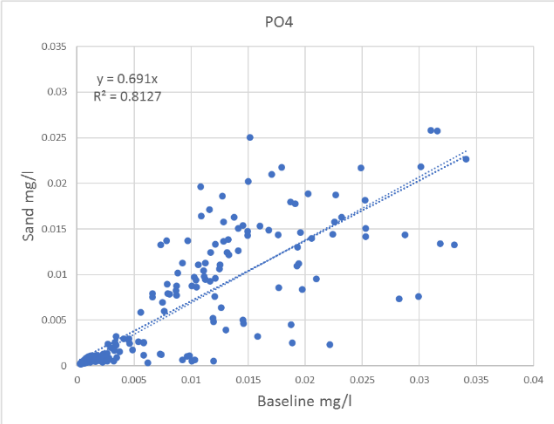
Internal loading
y = 1.3604x
R² = 0.9916
Sand mg/l/day
Baseline mg/l/day



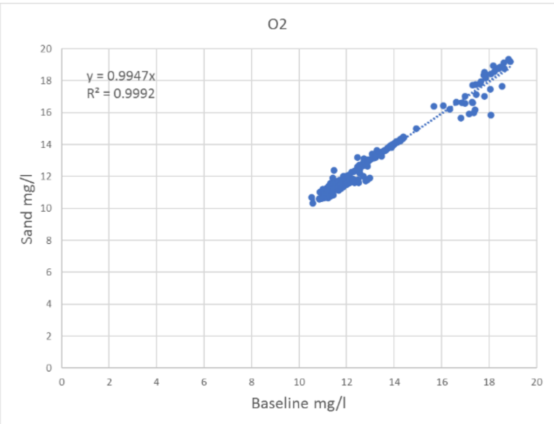
NH4
y = 0.9759x
R² = 0.9807
Sand mg/l/day
Baseline mg/l/day



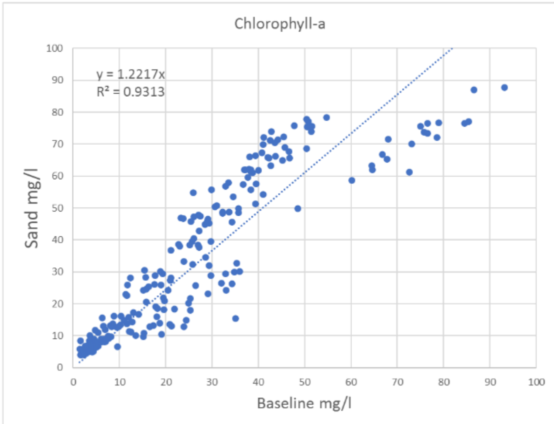
NO3
y = 0.8853x
R² = 0.9716
Sand mg/l
Baseline mg/l



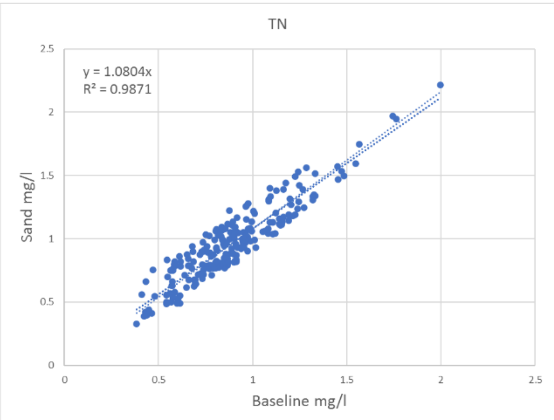
PO4
y = 0.691x
R² = 0.8127
Sand mg/l
Baseline mg/l



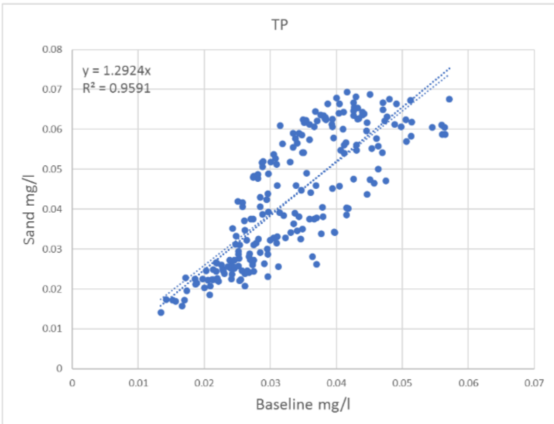
O2
y = 0.9947x
R² = 0.9992
Sand mg/l
Baseline mg/l



Chlorophyll-a
y = 1.2217x
R² = 0.9313
Sand mg/l
Baseline mg/l



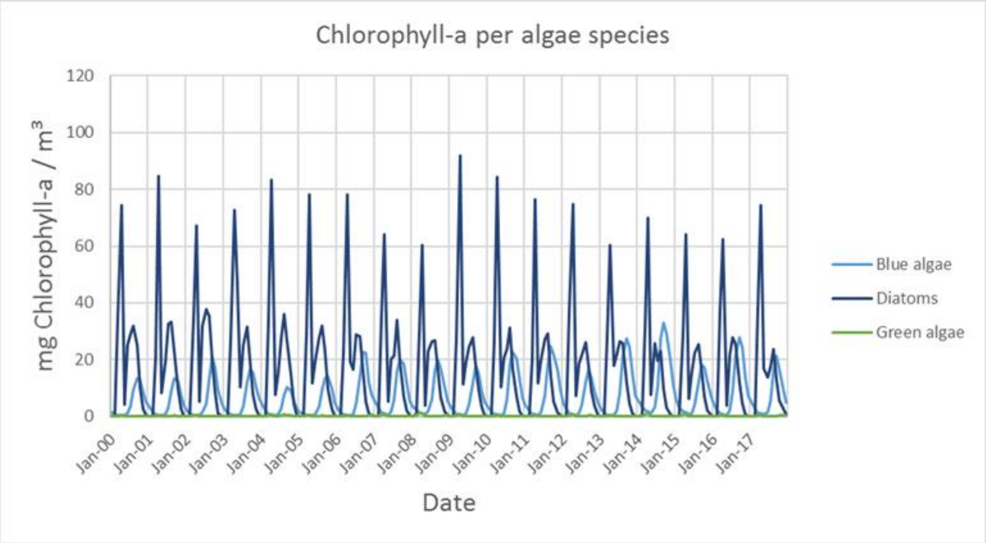
TN
y = 1.0804x
R² = 0.9871
Sand mg/l
Baseline mg/l



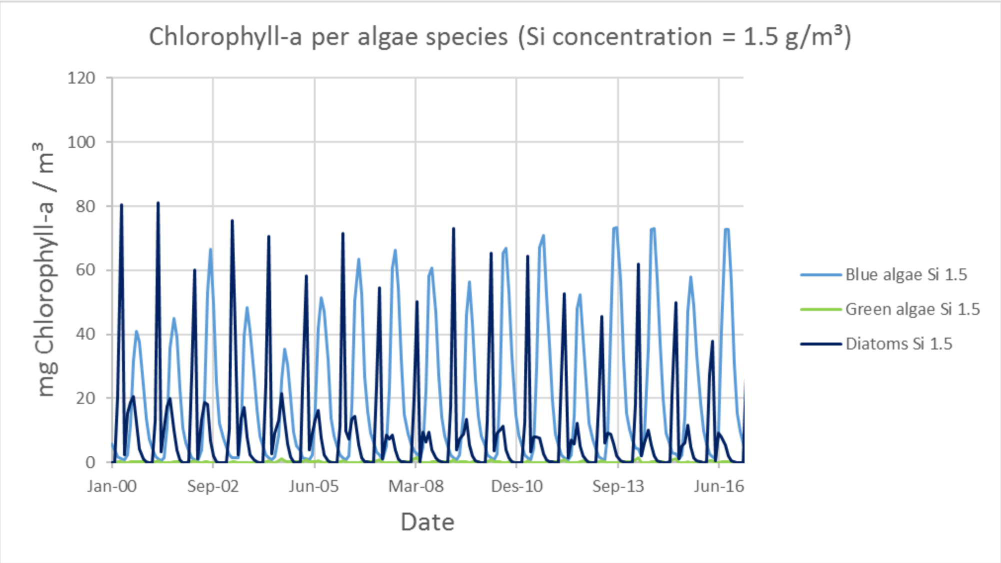
TP
y = 1.2924x
R² = 0.9591
Sand mg/l
Baseline mg/l

D. ALGAE SPECIES COMPOSITION PER SILICA CONCENTRATION IN INFLOW

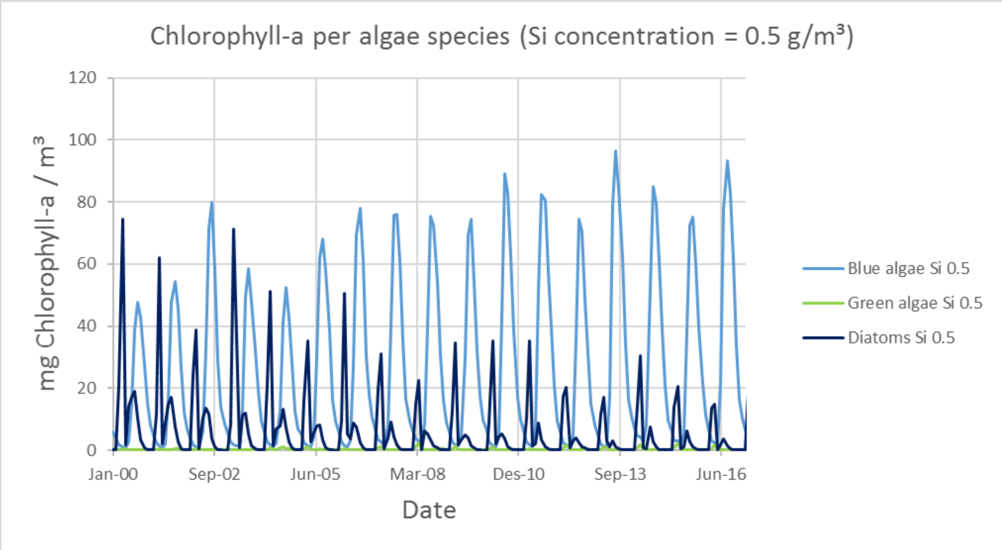
D.1. BASELINE CHLOROPHYLL CONCENTRATION PER ALGAE SPECIES (SI = 3 G/M³)



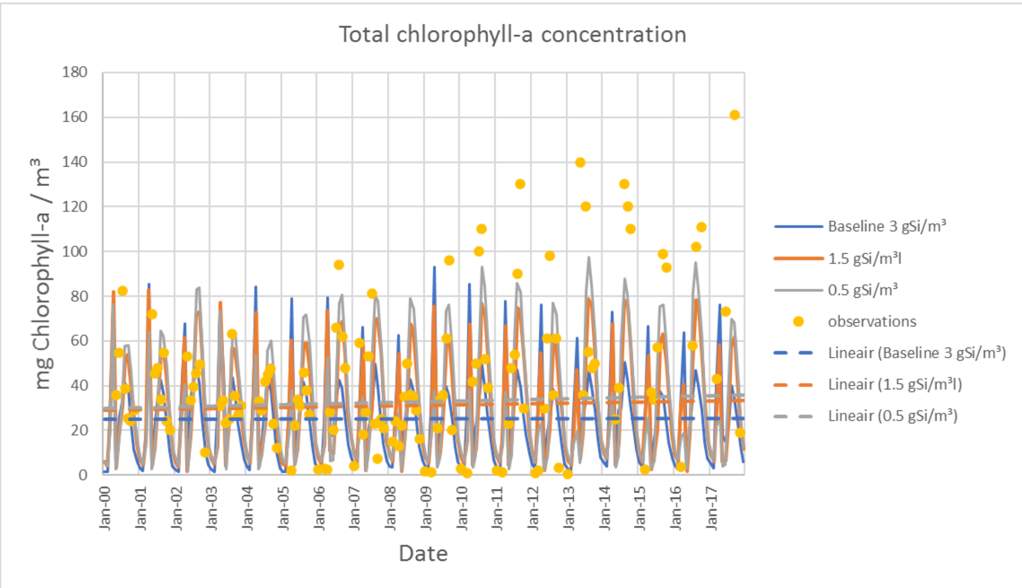
D.2. CHLOROPHYLL CONCENTRATION PER ALGEA SPECIES (SI = 1.5 G/M³)



D.3. CHLOROPHYLL CONCENTRATION PER ALGEA SPECIES (SI = 0.5 G/M³)



D.4. Total chlorophyll concentration with different silica concentrations compared to observations



E. SECCHI DEPTH LOAD-RESPONSE CURVE

F. LOAD-RESPONSE CURVE WITH A SILICA CONCENTRATION OF 0.5 G SI/M³