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Development in Phytoplankton Assemblages, Ecological Status and Purification Effect in Teglverksdammen

A Study of a Pond the First Two Growth Seasons Following Deculverting of an Urban Stream System in Oslo, Norway.

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Preface

This thesis concludes my Master's Degree in Environment and Natural Resources at the Norwegian University of Life Sciences. The study was done in collaboration with the Norwegian Institute for Water Research (NIVA).

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Oh, and thanks to our dog, Hera. She has been utterly useless as always, but as a wise person once said, "it is always nice to come home to a happy dog" (Me, 2018).

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Abstract

There is growing consensus that deculverting and restoration of buried urban streams may come with a range of ecological and socioeconomic benefits, including; reduced flood risks, improved water quality, facilitation of biodiversity and decreased habitat fragmentation. Ponds in such systems may constitute appreciated landscape-elements and further function as sedimentation basins, thus facilitating removal of environmental pollutants and nutrients associated with suspended solids. However, as such systems are susceptible to nutrient pollution, ponds may also potentially facilitate large amounts of phytoplankton, which may degrade the ecological status and aesthetic appeal of the system.

This thesis is a case study that covers the first two growth seasons of the pond Teglverksdammen that is a part of a newly deculverted reach in Hovinbekken, Oslo, Norway. Relatively few such urban deculverting projects have been conducted in Norway, and studying the development in Teglverksdammen can therefore offer valuable insights to problems and opportunities for future stream deculverting and restoration projects. The early development of the pond's phytoplankton assemblages is described, and it was tested how it related to physio-chemical environmental variables. It was also determined what ecological status was indicated by phytoplankton using the water framework directive classification system. Last, upstream and downstream water samples were used to test whether the pond facilitated net retention of nutrients and organic matter. In situ-measurements, phytoplankton samples and water samples were collected with monthly interval May-October the first two growth seasons following opening of the reach, 2016 and 2017. In addition, water discharge data (available for the first year only) and weather data were obtained.

Apart from during the longer stagnation period recorded, phytoplankton concentrations were moderate in the pond. Nutrient concentration and light availability were generally high, and neither could be identified as important controlling factors for the phytoplankton biovolume. As the residence time in the pond was generally short, variations in the rate of phytoplankton loss through flushing was likely of larger importance for the phytoplankton concentration than the growth controlling variables. The short residence time and location of the pond makes it susceptible to disturbances and the physio-chemical variables recorded also showed large alternations. The phytoplankton assemblage succession showed little order in form of seasonality or consistent response to the environmental variables examined here. In additional samples collected in the littoral zone in April 2017,

several marine species of phytoplankton were also found, indicating an ecological disturbance likely caused by salt pollution from road runoff. The ecological status as indicated by the phytoplankton quality element alone was moderate in 2016 and good in 2017, but as phosphorus concentrations remained high the overall ecological status was moderate for both years. There was large variation in whether the pond acted like a sink or source for nutrients and organic matter, although there was a general trend for retention. Data for stream discharge was only available for the six data points from the first year, but the results indicated longer residence did not increase net nutrient retention in the pond. The results further indicated longer residence time may result in increase in phytoplankton biovolume and overall trophic state of the pond.

Contents

1 Introduction

The practice of burying and culverting streams and rivers during urbanisation has been common in many places around the world (Elmore Andrew & Kaushal Sujay, 2008; European Environment Agency, 2016; Weitzell, Kaushal, Lynch, Guinn, & Elmore, 2016). Burying of streams has freed up space in growing urban areas. Further, urban streams are susceptible to pollution and have often been ecologically degraded, had low aesthetic appeal and have in addition been seen as possible sources of disease as wastewater effluent commonly reaches such streams (European Environment Agency, 2016). As a consequence, culverting streams have been common practice during urban development. In Oslo - Norway alone, almost 250 km of streams were culverted during development (Oslo kommune [Oslo Municipality], 2015). Culverting will of course have a major impact on the ecology of the culverted reach, but may also introduce problems in the remaining open parts of the stream. Most prominently, the culverts can constitute impassable barriers to riverine fauna, hereunder fish (Bates, Barnard, Heiner, Klavas, & Powers, 2003; Poplar-Jeffers Ira et al., 2009). Further, as streams are out of sight, incentives to reduce pollution, such as wastewater effluent reaching the streams may decline. Culverting also alters stream velocities and may therefore disrupt hydrological processes of erosion and deposition both up- and downstream of the culverts (Wild, Bernet, Westling, & Lerner, 2011). Blockages may also occur, which means there may be increased maintenance requirements and risk of flooding (Bates et al., 2003).

There is now growing consensus that opening and restoring culverted streams, also known as deculverting or daylighting, can come with a range of benefits. It may improve urban hydrology and drainage patterns, facilitate biodiversity, reduce habitat fragmentation as well as bring a range of socio-economic benefits associated with including more blue-green spaces in urban areas (Buchholz & Younos, 2007; Oslo kommune [Oslo Municipality], 2015; Palmer et al., 2005; Wild et al., 2011). A well-designed restored system can further have positive impacts on the quality of the water outlet into downstream reaches or the sea: UV-rays kill bacteria, riparian zones may retain particles, riffles aerate the water and wetlands can help with particle- and nutrient retention. Pools and ponds also act as sedimentation basins, which can facilitate removal of both environmental pollutants and nutrients as these often are associated with suspended solids (Horowitz Arthur, Elrick Kent, & Smith James, 2007; Wakida et al., 2014). Incorporation of ponds in urban streams may additionally constitute appreciated landscape elements, and add to the habitat complexity of a restored reach. Habitat

heterogeneity is recognized to be important for the biodiversity of streams, and urban ponds have been shown to contribute significantly to regional invertebrate diversity (Hill Matthew et al., 2016).

The adoption of the European Water Framework Directive (WFD) in Norway (in 2009) and the EU countries (in 2000) introduced stricter environmental objectives for all surface-, ground-and coastal waters. Culverted urban streams will normally be typified as heavily modified water bodies and are therefore not subject to the strictest objectives of good ecological status, but instead good ecological potential (European Environment Agency, 2016). Stricter WFD objectives however adds extra incentive for deculverting.

There however remains challenges and unknowns related to the success of deculverting projects. Urban streams are still susceptible to pollution, both by diffuse and point sources such as wastewater, as well as extraordinary events like spillages of industrial chemicals. They are also often recipients of wastewater (European Environment Agency, 2016). Increased storm water runoff in urban watersheds due to more impermeable surfaces may also affect water quality in streams, in part due to higher loading of suspended solids (Brabec, Schulte, & Richards, 2002). Both the amount of specific pollutants, as well as phosphorous, ammonium and electrical conductivity, have been shown to correlate with the amount of impermeable surface in a catchment (Hatt, Fletcher, Walsh, & Taylor, 2004; Wakida et al., 2014). Total and oxidized nitrogen often correlate with wastewater inputs (Hoare, 1984), which is also a further source for higher loading of organic matter and phosphorous. Consequently, urban streams are frequently nutrient polluted (Hoare, 1984; Hobbie et al., 2017). Ponds in restored reaches are therefore especially susceptible to the environmental pressure eutrophication, and could potentially facilitate large amounts of phytoplankton. The trophic state may affect both flora and fauna of limnic systems and the high loading of organic matter associated with eutrophic conditions can drastically effect oxygen demands. In lakes, both high phosphorus concentrations and high total phytoplankton concentrations often correlate with increased amounts of cyanobacteria (Brettum & Andersen, 2005). Large amounts of cyanobacteria is generally considered undesirable as some taxa can produce compounds that cause foul odours or that are toxic to humans and other organisms (Watson, Ridal, & Boyer2008). In addition, high algal volumes (“algae” here referrers to both eukaryotic algae and cyanobacteria) may drastically reduce the aesthetic appeal of the pond.

In Oslo, the municipality has decided to daylight as much as possible of the almost 250 km of streams and rivers that has been culverted during development in the region.

Improvement in storm water drainage and reduced risks of flooding are important incentives for daylighting in places like Norway, where climate change is predicted to increase both the amount and intensity of precipitation (Füssel, 2013). This is one of the most important objectives for daylighting in Oslo (Oslo kommune [Oslo Municipality], 2015). Other important environmental objectives include that stream daylighting and restoration should recreate important biotopes and reduce habitat fragmentation. Although many of the culverted reaches are classified as heavily modified waterbodies, the municipality's policy document for deculverting streams states that striving for good ecological status is an overall objective in these restorations (Oslo kommune [Oslo Municipality], 2015). Further, water quality should be improved through restoring natural processes in parallel with increased measures to protect the streams through improved handling of storm water and wastewater overflows. The streams are also seen as important landscape elements and should provide opportunity for outdoor recreation.

This thesis is a case study that covers the first two growth seasons of the pond Teglverksdammen, that is a part of a newly deculverted reach in Hovinbekken, Hasle, Oslo, the capitol of Norway. The approximately 650 m long deculverted reach, named after the main pond (Teglverksdammen), opened in August 2015 and was one of the first completed daylighting projects in Oslo after the new stream restoration policies were put in place. Like for many urban streams, there are several upstream sources of pollution and the restored reach is recipient of wastewater both through misconnections and leakages. Therefore, further remediation of the upstream culvert network will be an important measure for improving water quality. As for now however, nutrients loading is high in the reach (Norconsult, 2013).

General objectives for deculverting projects in the region apply also for Teglverksdammen. One of the most emphasized objectives for this reach however, is that it should contribute to nature-based purification of the water for the further downstream reaches that have in part been opened since the opening of Teglverksdammen. The reach is therefore designed with several smaller sedimentation basins, riffles, pools, permeable thresholds and dense macrophyte vegetation, developing wetland areas (helophyte vegetation was still sparse during research) and ponds including the larger main pond Teglverksdammen. This design is thought to have an effect on the hygienic quality, pollutants associated with suspended solids as well as for nutrient reduction, in particular phosphorous. Sedimentation is a key purification mechanism, and the feasibility study estimated that the effect on nutrient removal in the pond and wetlands part of the reach might reach 30% when the residence time averages around 24 h (Norconsult, 2013). The reach receives water from the old culvert and the

average inlet discharge may be regulated after what water quality that can be achieved by the outlet of the reach (Norconsult, 2013). The pond is smaller than what normally is typified as a lake and as such has both lentic and lotic characteristics. Removal of nutrients and organic matter by sedimentation and biological assimilation is dependent on the residence time in the pond, with longer residence times potentially allowing for more effective purification. The efficiency of a pond like Teglverksdammen to retain nutrients and organic matter however further depends on other environmental factors. For phytoplankton, longer residence times and high nutrient concentrations may result in large biomass, which degrades the ecological status and may affect aesthetic appeal of the system. There is however little knowledge about the early development in phytoplankton assemblages in such a system in a northern climate.

Since still relatively few such urban deculverting projects have been conducted in Norway, studying the development in Teglverksdammen can offer valuable insights to problems and opportunities for future stream deculverting and restoration projects. In this instance, specifically relating to the success of ecological restoration and the efficiency of nutrient removal in such a pond. The opening of the site further provides an opportunity to document the early development in phytoplankton assemblages in such a system. The individual aims in this study was therefore to:

- document the early development in phytoplankton total biovolume and assemblages and the relationship with physio-chemical environmental variables
- assess the development in ecological status using the eutrophication related quality element phytoplankton, and the supporting quality element phosphorous, from the water framework directive
- assess the ability of the pond and adjoining wetlands to purify water through retention of nutrients and organic matter

2 Methodology

2.1 Area Description – Teglverksdammen and Hovinbekken

Teglverksdammen is a pond in a 650m long deculverted and restored reach in Hovinbekken downstream Økern in Hasle, Oslo, Norway, that was opened in August 2015.

Hovinbekken is one of the 10 major water courses that runs through Oslo. It is a small to medium sized stream, with an average water flow of 0.18 m^3/s in the last 20 years (Bækken, 2011). It drains from Årvollmarka and is a partly open, partly culverted stream through Årvoll, Brobekk and Risløkka. From here however, it has until recently been culverted almost the whole way from Økern through Ensjø and Grønland, where it merges with one of Oslo's other large rivers, Akerselva (Tønnessen, 2010). As such, it is the most culverted stream in Oslo (Fergus, 2016).

Hovinbekken's catchment consists mostly of forest in the upper part of the catchment, and largely urban and industrial areas in the lower regions above Teglverksdammen (Figure 1). A small percentage of the catchment is also farmland (see generated catchment map from NEVINA in Appendix A).

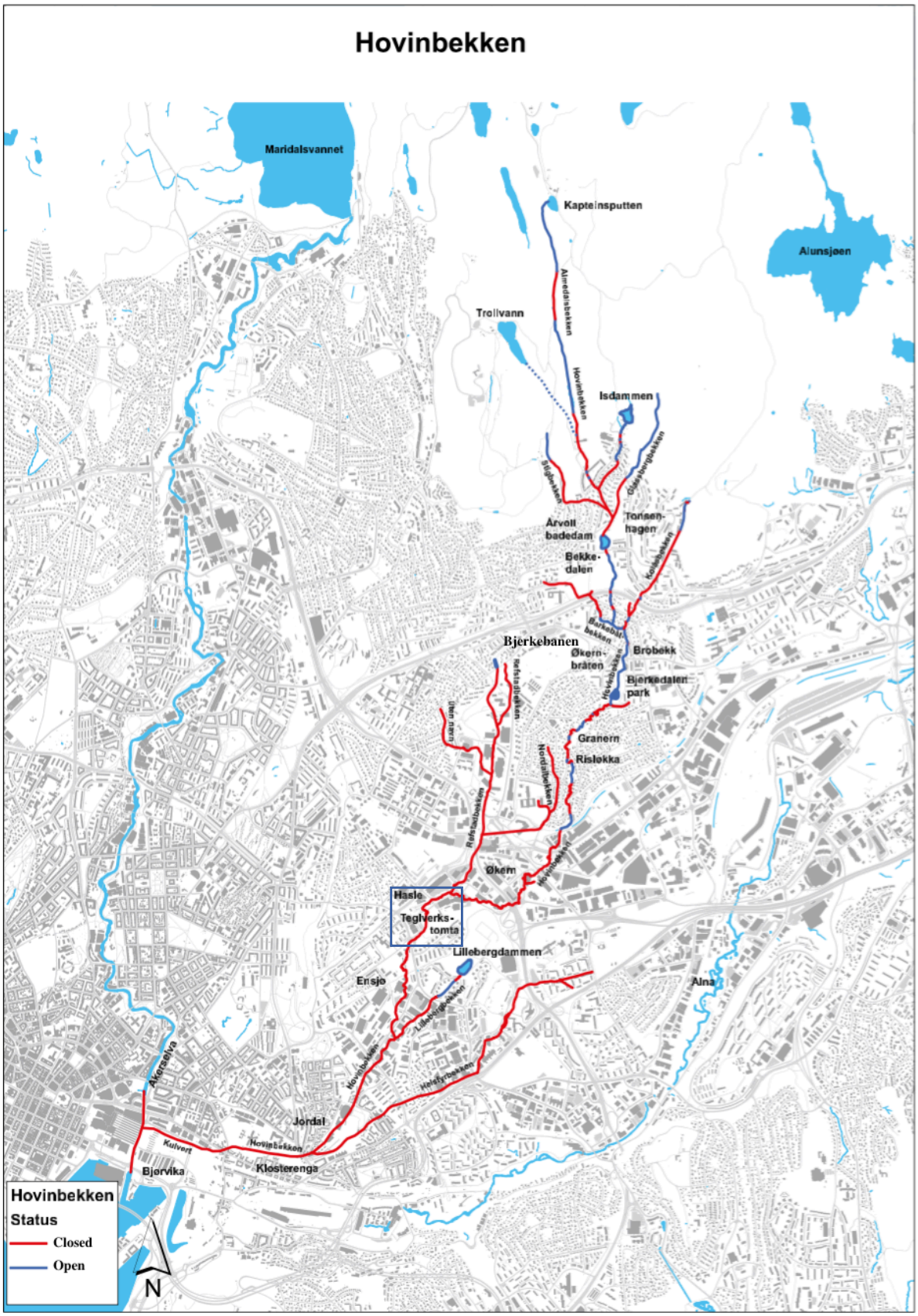

Figure 1. Map over Hovinbekken and the catchment before deculverting. Teglverksdammen is situated downstream of Økern, in the marked area named Teglverkstomta. Map developed by Oslo Elveforum and Oslo VAV, used with permission from Oslo VAV.

Several sources of pollution are present in the catchment. Hovinbekken is the recipient of much untreated surface runoff from the urban catchment, including runoff from major roads such as Østre Aker vei. Runoff from the major road "Ring 3" should mainly reach the Hovinbekken culvert downstream of where water for the restored reach is extracted. Untreated runoff from the horse racecourse Bjerkebanen also reaches Hovinbekken and could contribute to considerable amounts of nutrients and *E. coli*. Last, Hovinbekken also receives wastewater, both trough misconnections and leakages (Norconsult, 2013).

Upstream of Økern, Hovinbekken is registered as having moderate ecological status according to the WFD, and the environmental objective is at least good ecological and chemical status for the waterbody (Sandlund et al., 2015). Downstream of Økern however, the whole stream has been culverted until recently and the stream is considered a heavily modified water body ("Vann-Nett [Water-Net]," 2018). The environmental objective is therefore "good ecological potential" according to the WFD standard. The policy document for stream daylighting in Oslo however further states that daylighting project should facilitate "as good water quality as possible" and that handling of storm and waste water should not hinder an overall environmental objective of reaching good ecological status of deculverted urban streams (Oslo kommune [Oslo Municipality], 2015).

The approximately 650 m long restored reach below Økern (Figure 2) was officially opened in August 2015. The name Teglverksdammen in this paper refers to the largest pond in the restored reach, but the name is often used to refer to the whole reach. One of the most prioritised objectives when designing the restored reach was that the reach should treat the water for the further downstream reaches through Ensjø, which were partly opened and connected to the outlet of Teglverksdammen in Autumn 2016. The reach is therefore built like a natural open water cleaning facility with settling ponds, a stream with dense vegetation, riffles and pools as well as wetlands above and below Teglverksdammen(Norconsult, 2013). The sections of the reach designed as wetlands are still developing, some parts only holding scattered helophytes. These sections could therefore partly be considered a shallow, wide stream with slow flowing water, but will in this study be referred to as wetlands. In the inlet, untreated water from the culvert is pumped into the restored reach. The first part of the reach, Tennisdammen, therefore consists of two pre-treatment settling pools and permeable thresholds with emerging macrophytes. In the feasibility study (Norconsult, 2013), this part of the reach was considered the most important element for water treatment. The aim was that this part should retain most of the sludge and suspended particles, and with that also associated nutrients, heavy metals and oils. From here, the water runs through a short culvert

under a smaller road and into a stream section with riffles, pools, dense emerging macrophyte vegetation and permeable thresholds. This section ends in the first not-yet-developed wetland, which transitions in to the largest pond, Teglverksdammen. Teglverksdammen then transitions to another wetland below the pond before the water enters the last sedimentation pond in the reach, Grensedammen. Teglverksdammen has a surface area of 6000 m^2, a maximum depth of little more than 3 m, and with the adjoining wetlands a volume of approximately 13000 m^3 (Norconsult, 2013). The east side of the pond faces a hill, and has a restored riparian zone, while the west side has a flat asphalt and concrete interface. The main treatment effect in this part is through sedimentation of particles, but nutrient uptake by vegetation and UV-treatment of *E. coli* is also important mechanisms (Norconsult, 2013).

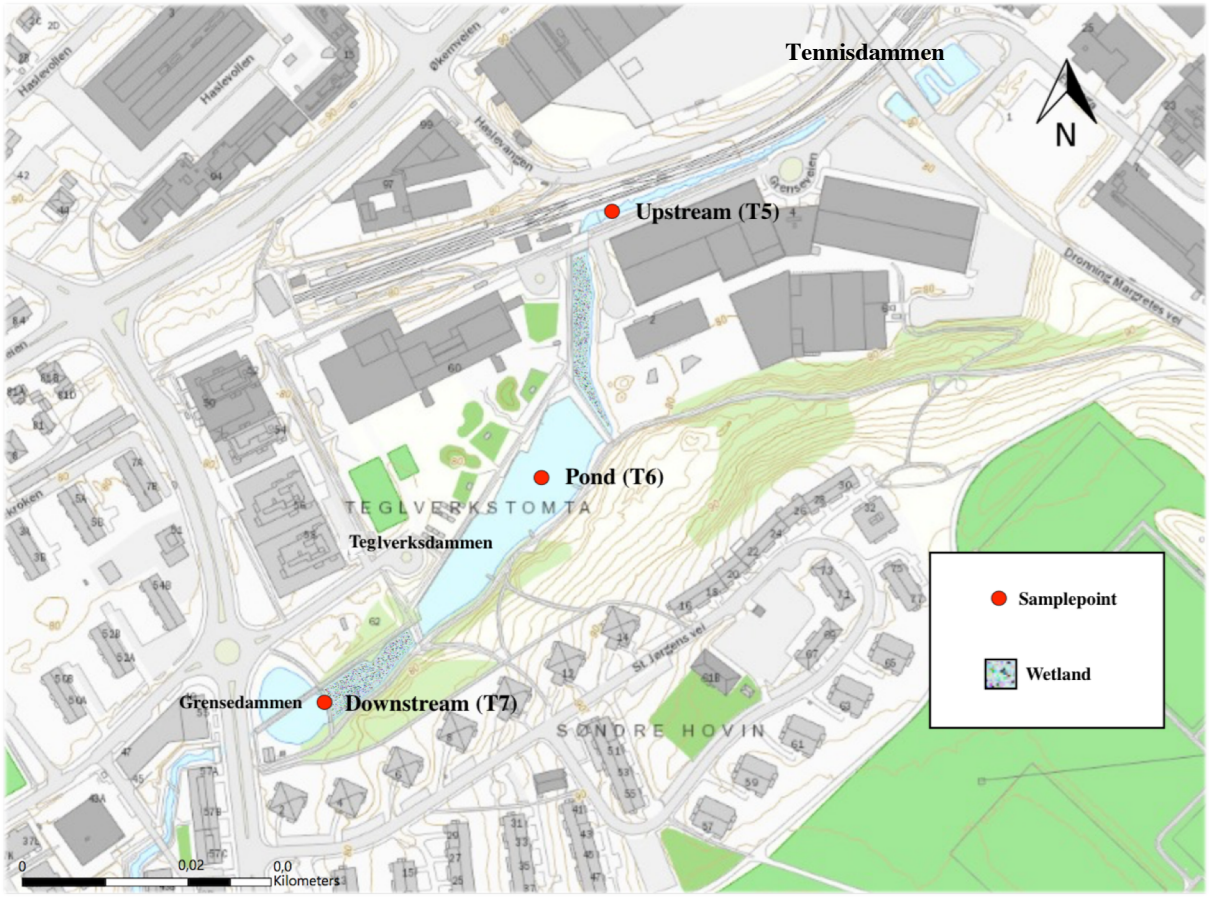

Figure 2. Teglverksdammen and the restored reach of Hovinbekken. Samples sites and wetlands are marked, original map generated from Norway's Water and Energy Directorate (Norges vassdrags- og energi direktorat) database NEVINA (2017).

2.1.1 Sample sites.

Three sample sites were used in this study (see Figure 2 earlier chapter); one upstream, one downstream and one within the pond. The upstream/downstream sites were sampled in the first small riffles found above and below the wetlands. For the pond, a point in the deepest area of the pond was selected. The names T5 and T7 corresponds to the site names used in two earlier master-theses on the stream ecology by Arnott (2016) and Myrstad (2017) (pictures and geographic coordinates of sampling sites in Appendix B).

2.1.2 Site events.

A few events at the sites may be of importance to the interpretation of the results. Due to maintenance work on the inlet-vent in August and September 2016 (11.08.2016 – 12.09.2016) there was no to minimal flow in the stream. During the sampling in September 2016 when the water had just been turned on again, the water table in the pond was approximately 0.5 - 0.9 m lower than normal, and no water was flowing out from the pond, leaving the downstream site dry. There was also a period with low flow due to operation problems with the vent in August 2017, starting around the 15th and lasting to the 21st. Sampling this month was on the 16th. Last, during sampling for other research projects in November 2016 at the first upstream non-culverted site, around 30 dead fish were found in the area before where the water enters the culvert.

2.2 Sampling, in Situ Measurements and Data Collection

Samples and in situ measurements were collected with a monthly interval from May to October the first two growth seasons after opening of the reach, 2016 and 2017. All fieldwork was done between approximately 10 am and 13 pm. At an inspection in April 2017, additional samples of floating patches of cyanobacteria was collected in the littoral zone of Teglverksdammen.

Stream discharge measurements were obtained from Oslo VAV and were recorded near the inlet of the restored reach. Data on temperature, incoming shortwave solar irradiation (SI) and precipitation is from Blindern metrological station (station no: 18700) and collected from the online database eKlima by the Norwegian Meteorological Institute (2018).

2.2.1 Sampling and in situ measurements in the stream.

For all chemical analyses of water samples, a one litre plastic bottle was filled, marked with station name and date and stored in a cooler bag until delivery a few hours later to the

laboratory at Oslo's water and sewage department (Oslo VAV, Norwegian name: Oslo Vann og Avløp) for further analysis. The bottle was placed in the middle of the streamflow so that the water sample collected was well-mixed and representative for the stream. All bottles were rinsed three times in the stream water before collecting the final sample. Care was taken so that the sediment upstream had not been disturbed, and so that the water samples were not contaminated through contact with skin or un-rinsed equipment. The bottles used were provided by Oslo VAV and the Norwegian Institute for Water Research (NIVA) and of a standard approved for limnological analyses. In September 2016 after the maintenance period, the stream below the pond was dry, and the sample from here (T7) could not be collected.

2.2.2 Sampling and in situ measurements in the pond.

Sampling and in situ measurements in the pond was done from a rowing boat in the deepest part (approximately 3 meter) of the pond. First, a two-meter Ramberg sampler was used to collect an integrated 0-2 meter vertical water column sample from which to extract mixed samples for chemical analysis, a chlorophyll a reading and phytoplankton-analysis. A multi-parameter sonde (YSI EXO2) was then used to record a profile over the water column. Last, the secchi-depth was recorded and a plankton net with a 25 µm mesh was used to collect a concentrated live sample of phytoplankton.

The mixed integrated sample was collected by lowering the Ramberg-sampler to just below the surface and then lifting it up by the attached rope before transferring the water to a mixing bucket. The Ramberg sampler used is a PVC tube, designed with a weighted bottom so that it stays vertical in the water, and with an open top and a one-way float-valve on the bottom so that water flows freely through it during descent but locks inside during ascent. All sampling equipment and instruments were disinfected with Virkon S between uses and sampling equipment rinsed a minimum of three times in the pond water before sampling. The turbulence from the transfer between the tube-sampler and the mixing bucket was sufficient to ensure that the water in the container was well mixed before the separate samples were extracted. The water was extracted on as un-disturbed water columns as possible, a little bit away from where the equipment was rinsed. As an anchor was sometimes needed to fix the boat, care was also taken so that sampling was not done just above the anchoring point but rather a little to the side or upstream.

The samples for chemical analysis, the chlorophyll a reading and the phytoplankton-analysis were taken from the integrated water sample. Using the multiparameter sonde (described further below), a chlorophyll a reading was recorded from a subsample using a

separate cup that was covered so the sensors were in the dark. A minimum of five chlorophyll a values were registered and a mean was noted. For the phytoplankton sample, a 100 ml glass bottle was filled with water from the mixed sample, and approximately 0.5-1 ml acidic Lugol's iodine 1% was added. The bottle was marked with station name and date and stored in the dark to avoid oxidation of the preservative. Water samples for chemical analysis were also taken from the mixed sample and were handled according to the same protocol as for the samples in the stream water. The sampling procedure used here was in accordance with the Norwegian standard NS 9459:2004, which recommend phytoplankton to be sampled from the same depths as other biological and chemical parameters examined. An integrated sampling depth of two meters is also in accordance with the standard NS-EN 16698:2015 which recommend not sampling the bottom 0.5-1 m of a lake, but otherwise the whole water column for a polymictic lake or whichever is larger in a stratified lake; the whole euphotic or epilimnic zone. Following these standard protocols allows for the variables from the phytoplankton and water chemistry samples to be used in WFD assessments.

The multiparameter sonde was further used to record profiles over the vertical water column. The sonde has a range of sensors, including a combined conductivity-temperature sensor, two combined depth-level and pressure sensors, an optical dissolved oxygen sensor, electrochemical cell pH sensor, a dual-channel fluorescence algae sensor and a turbidity sensor. The instrument software converts the sensor data inputs into a range of units of which chlorophyll a (RFU/ μg/L), temperature (°C), conductivity (μS/cm), salinity (PSU), depth (m), dissolved oxygen (% saturation, mg/L), turbidity (FNU) were used in this study. The corresponding standards and uncertainties of measurements are given in Appendix C. The profiles were recorded on a vertical decent and on an as undisturbed profile as possible. As the sonde needed recalibration for chlorophyll a fluorescence in June 2016, the chlorophyll a values from this month are missing. The sonde was recalibrated at NIVA before the next fieldwork in July 2016 (1-point calibration against distilled water). For comparison, an additional water sample for laboratory analysis of chlorophyll a was collected from the mixed sample during the fieldwork this month. It was collected in a dark 1-litre plastic bottle provided by NIVA, and stored in a dark and cool container before delivered to the laboratory at NIVA where the analysis was done.

Secchi depth was recorded after the sonde profile recording and water chemistry samples as it might otherwise disturb the water profile for the other recordings and samples. The secchi disc was lowered into the water until the disc was no longer visible. The disc was then pulled up until just visible again, and the length from the water surface to the disc was

measured. The secchi disc used had a diameter of approximately 20cm, a white surface and was weighted with lead underneath to stay vertical in the water column. The secchi depth (Z_{SD}) is an indirect measurement of the light conditions for photosynthetic activity in the pond and usually show a linear correlation with the depth of the euphotic zone (defined as the depth where approximately 1% of surface light remains). A rule of thumb is that the euphotic zone can be found by multiplying Z_{SD} by a factor of 3, but the exact relationship depends on the water's properties, light conditions, observers eyesight and to a lesser extent also the area of the disc (Cole, 1979).

Concentrated live plankton samples were also collected with regular intervals. A plankton nylon net with a 25 µm mesh was weighted with a glass bottle at the bottom and pulled up and down in the water column 2-4 times. The concentrated phytoplankton sample was stored in a glass bottle and examined under microscope within a few days. The purpose of live samples is to ease identification of species that might be difficult to identify in preserved samples.

2.3 Sample Analysis

2.3.1 Phytoplankton.

The quantitative analysis of phytoplankton on the Lugol's iodine preserved samples was done microscopically and included counting, identification to lowest possible taxonomic rank and calculation of taxon-specific and total biovolume. The procedure used is in accordance with the standard NS-EN 16695:2015 and meets the requirements for use of phytoplankton in determining ecological status is in accordance with WFD standards.

The analysis was carried out using the following equipment:

- Inverted microscope, Leica DMi 8 with phase contrast and DIC, fitted with;
    - 10x magnification binocular eyepieces
    - Objectives with 10x, 20x and 40x and 63x magnification
    - Digital camera connected to the visual software Leica Application Suite
- 10 ml round counting chamber with 25mm diameter
- Bottom and cover glass for the chamber
- Distilled water for cleaning or topping up the chamber when water had condensed

The preparation of the samples included acclimatization of equipment and the sedimentation of phytoplankton in the chamber. Before sedimentation, samples and the sedimentation chamber were acclimatized in room temperature, as is important for even

distribution in the chamber. For an even mix of the sample, the bottle was turned 100 times before the 10 ml counting chamber was filled. The chamber was marked and stored in a Styrofoam box for minimum 24 hours to let the sample sediment.

The counting procedure was carried out in three steps. Large and rare taxa were counted in the whole chamber surface on low magnification (100X). Intermediate taxa were counted on two random chamber transects on 200X magnification or four transects on 400X magnification (min. 5% of the chamber area examined), or less transects when the number of counting units for the size group exceeded 400 (for an evenly distributed sample n=400 gives a 5% precision for number of counting units in that step). Smaller taxa, not counted at lower magnification, were counted using large magnification of 640X and random transects and additional random fields of view until the number of counting units had exceeded 400.

Biovolume was estimated using measurements of visible dimensions and estimates of hidden dimensions. Measurements of visible dimensions were taken on 640X magnification using the eye-piece ruler or the digital ruler in the visual software. Measurements were noted in micrometres with 1-2 decimal points. For numerous taxa, mean biovolume of minimum 20 individuals was used. This normally gives a biovolume standard error of <10%. Taxa with very variable sizes were divided into further size groups before a mean was calculated. For large taxa like filamentous algae all counting units were measured. Hidden dimensions and geometrical shape were estimated using suggested dimensions for the species described in the standard. When the geometrical shape and hidden dimension relations for a taxon were not given in the standard, this had to be estimated from photos, literature or using estimates for taxa with similar geometrical shape. Biovolume for each counting unit was then calculated, and the estimates in each separate counting step was multiplied by a factor determined by how large sample volume is represented by the chamber area in that counting step. The biovolume estimate was noted in mm^3/l. The standard used assumes that phytoplankton density is on average approximately the same as for water and therefor that $1mm^3/l = 1mg/l$ (wet weight). These units are therefore interchangeable where the WFD classification system refers to biomass.

The identification of taxa was done to lowest possible taxonomic rank. Identification was done according to literature and keys by Guiry (2003), Cox (1996) and the Süßwasserflora von Mitteleuropa series (Pascher, 2005), as well as with assistance from phytoplankton expert Birger Skjelbred at NIVA. The online database Algaebase (Guiry & Guiry, 2017) was used to find latest updates on currently accepted names and taxonomic status of individual taxa.

2.3.2 Water sample analysis.

The water samples were analysed by Oslo VAV for total organic carbon (TOC), calcium, total nitrogen (TN), nitrate (NO3-N), ammonium (NH4-N), total phosphorus (TP) and phosphate(PO4-P). Standards followed, and accuracy of procedures are given in Appendix C.

2.4 Data Treatment and Statistics

2.4.1 Statistics.

Statistical analyses were conducted in R-cmdr version 2.3-2 and graphs were plotted using Prism 7. Correlation test are Pearson's product-moment and mean values arithmetic. Level of significance in this paper was set to 5%. Bonferroni correction was applied in multiple test (>5) so that α for significance was set to 0.05/n, were n is number of multiple comparisons. Results with a p-value>0.05/n was discussed as non-significant but low p-values were discussed further as indications of trends. On analyses that compared linear associations or paired differences, only data points that hold all variables of interest were included. This is relevant for paired t-tests on water chemistry upstream and downstream the pond, all correlation tests and the principal components analyses (PCA). PCAs were further done on standardised variables. All relevant pairwise linear associations with low p-values were checked with basic diagnostic plots, and data points that had a Cook's distances $D_i > 4/n$, where n was the number of data points, were discussed as influential.

2.4.2 Pond residence time.

The pond residence time reflects the theoretical mean time the water has resided in the pond. Here, this is given either as residence time at the day of sampling (days since a volume of water equal to the pond volume, 13000 m^3, had passed through the system), or as a global mean for the whole period discharge data exists. For the global mean residence time, the pond volume was simply divided by the mean daily discharge for the whole period. The residence time at day of sampling was found using the daily mean discharge data for relevant data points. The data was first used to calculate the corresponding daily volumes that passed. The residence time at day of sampling was then found as the number of days it would have taken for 13000m^3 if water to pass through, given the historical discharge before sampling. Half the

corresponding daily volume was used for the data point at the day of sampling, and the needed fraction for the first relevant data point.

2.4.3 Water quality parameters.

Both the water chemistry data from analyses at Oslo VAV and data from in situ measurements were used to describe the development in the pond. The data was plotted to show the development in the variables, and it was tested whether there was a correlation between the residence time at time of sampling and the water chemistry.

Data from the pond recorded with the multiparameter sonde is given here, unless otherwise stated, as a mean value of the readings in the top 0-2 m of the profile. This is relevant for the data; temperature, pH, turbidity (FNU), and conductivity from the pond when it is just given as a single value.

2.4.4 Pond profiles, Temperature oxygen and conductivity.

The sonde data is used to graphically display the temperature (°C), conductivity (μS/cm) and dissolved oxygen (% saturation).

2.4.5 Phytoplankton biovolume, chlorophyll a.

The phytoplankton total biovolume is used to describe the pond's trophic development. Different intervals and boundary values for phytoplankton biovolume have been used to define different trophic states historically. A rough subdivision into the three trophic state levels oligotrophic, mesotrophic and eutrophic are commonly used (Brettum & Andersen, 2005). In this study, a system which uses both mean and maximum biovolume to place a lake into one of seven trophic levels was used. The boundary levels and intervals are the same as used by the Norwegian Institute for Water Research (NIVA) in their report "The use of Phytoplankton as indicators for Water Quality" (Brettum & Andersen, 2005), and originally defined by Brettum (1989). The trophic states are, from lowest to highest; ultraoligotrophic, oligotrophic, oligomesotrophic, mesotrophic, eutrophic, polyeutrophic and hypereutrophic (intervals as in Appendix D).

Laboratory chlorophyll a analyses are often conducted to get a second estimate for the biomass of phytoplankton. The sonde chlorophyll a readings in this study were only semi-quantitative but should be fairly linear with the chlorophyll a concentration in water of similar quality. Here, the chlorophyll a readings were used only for comparison with the biovolume estimate to reveal any problems. The estimate for chlorophyll a was found through linear

interpolation and the relative fluorescence unit (RFU) output of the sonde measured at the 2 m mixed sample. The sonde was 1-point calibrated at NIVA using distilled water with regular intervals. The linear interpolation was done manually using the chlorophyll a value from the June 2016 laboratory analysis and the RFU output. The first point was defined as (0,0) and the second point as the lab result for chlorophyll a and the RFU output. The equation found was further applied to the RFU values from other readings to get estimates for the other months. Finally, the sonde chlorophyll a values were plotted against the biovolume, and a correlation test was done.

Last PCAs were conducted with the phytoplankton biovolume and the environmental variables that may be important for growth, as well with the residence times from 2016. It was further tested whether phytoplankton correlated with these variables individually. The SI values used in these tests was cumulative incoming solar irradiation four days previous to sampling (from noon day of sampling). The time duration of four days was chosen as this was the global mean residence time.

2.4.6 Phytoplankton assemblages.

The development in phytoplankton assemblages was plotted as relative composition using taxonomic groups at phyla level. The developmental patterns observed for the largest phyla were further described qualitatively to see if any overall patterns could be recognised.

2.4.7 Determining ecological status as in the water framework directive.

The biological quality element (BQE) phytoplankton and the supporting quality element (QE) total phosphorus were used in overall classification of ecological status. The indicated status for the QEs total nitrogen and oxygen in hypolimnion were also found. The QEs phytoplankton, phosphorus and to some extent nitrogen measure the environmental pressure eutrophication, and oxygen levels in hypolimnion measures the effect of high organic loading, which may be a result of eutrophication. Ecological status in the WFD system is classified on a scale using the ecological status classes high, good, moderate, poor and bad. All protocols followed in this chapter are as stated in the Norwegian classification guidelines (Sandlund et al., 2015).

Typification of the water body.

The ecological status indicated by a quality element is found using different reference states for different types of lakes. The pond was typified as LN-1 / 10 based on its characteristics; situated in the lowland, calcareous (as indicated by mean Ca), clear (as indicated by mean TOC). Typification is needed to establish which class intervals that should be used to determine the ecological class of individual QEs. The LN-1-type was needed for classification of ecological status for phytoplankton and oxygen saturation in hypolimnion, and the 10-type was needed for classification of nutrients. Neither type fitted the pond perfectly, but as recommended in the classification guide the closest type was chosen. It should be taken into consideration when interpreting the results that the pond is smaller and shallower than what is typified as a lake in the WFD system.

The biological quality element phytoplankton.

The BQE phytoplankton combines indices for biomass, assemblages and cyanobacteria to measure the environmental pressure eutrophication. The three indices are used to generate ecological quality rations (EQR) that are then normalised and used to find an overall indicated status for phytoplankton.

The assemblage index, known as the Phytoplankton Trophic Index (PTI) was determined for each sample. The index is based on phosphorus optimum (log) values for different taxa, which are given in the classification guide. The PTI for each sample is found as in Equation 1. The value used in the index is the yearly mean.

Equation 1: $$PTI = \frac{\sum_j^n ajsj}{\sum_j^n aj}$$

Where
a_j is proportion of jth taxon in the sample and
s_j is the optimum of jth taxon in the sample.

The biomass index is normally based on a mean EQR from chlorophyll a and total biovolume to get higher certainty. But since the chlorophyll a reading from the sonde is only semi-quantitative and not the standard way to measure chlorophyll a, it was not included. The biomass index is therefore simply the mean biovolume.

The last index, cyanobacteria, is simply based on the yearly maximum biovolume of cyanobacteria observed.

To determine what status was indicated by the phytoplankton QE, the indices were combined through finding their EQR values. These were further normalised into nEQR values that were finally combined in an overall nEQR for the quality element. The EQR for each index was found as in Equation 2, and normalised to nEQR values as in Equation 3. The three indices were further combined into one mean nEQR for phytoplankton. However, the nEQR for cyanobacteria should only be included in the mean if it indicates worse ecological status than either of the other indices. This is because the cyanobacteria index can only be used to lower the final status. The indicated status was found using the combined nEQR – class intervals as in Table 1.

Equation 2: EQR Cyano max, mean PTI and biovolume $= \frac{Obs - Max}{ref - Max}$

Where
obs= observed value
max= maximum value for the index*
ref= reference value for the index*
*reference and maximum values as in Norwegian classification guidelines for the lake type

Equation 3: $nEQR = \left\{ \left(\frac{EQR - lowerEQRclassborder}{upperEQRclassborder - lowerEQRclass\ border} \right) \times 0{,}2 \right\} + lowerEQRclassborder_n$

Where
nEQR= normalized EQR
lowerEQRclassborder* = lower non-normalized EQR border for the relevant class
upperEQRclassborder *= upper non-normalized EQR border for the relevant class
lowerEQRclassborder*n = lower normalized EQR border for the relevant class
0,2 = standardized class-with for the normalized scale
*Class borders as in as in Norwegian classification guidelines for the lake type

Table 1. Status class and normalised EQR class boundaries.

	nEQR Boundaries
High	>0.8
Good	>0.6, <0.8
Moderate	>0.4, <0.6
Poor	>0.2, <0.4
Bad	<0.2

Supporting quality elements total phosphorus and total nitrogen.

The indicated ecological status from the supporting chemical quality elements total phosphorus and nitrogen was found using yearly mean values. Total nitrogen was not needed in the overall classification as it is normally used only when nitrogen limitation is suspected.

The indicated status from both these nutrients is however useful on its own, as it says something about whether the observed values are high or low for the water body type.

Quality element dissolved oxygen in hypolimnion.

The QE dissolved oxygen in hypolimnion is normally used as a supporting element for the BQE fish. The QE was here used only as a reference for whether observed oxygen levels (mg/l) indicated poor conditions for biota. Oxygen levels in the 50^{th} and 5^{th} percentile of hypolimnion the month with lowest oxygen levels were used and compared to reference values for ecological classes. It should be noted the QE is developed for lakes, and normally not used in classification of smaller water bodies.

Combination of quality elements for classification of overall ecological Status.

The overall ecological status was determined using the QEs phytoplankton and total phosphorus. It was found for each year separately and for both years combined. As the September 2016 sample was taken after a long stagnation period and had high phytoplankton biovolume, it was tested whether removal of the data point for the phytoplankton quality element changed the resulting status. To find the ecological status for both years, the mean of the annual nEQR for phytoplankton was used.

When a biological quality element indicates less than good ecological status, the supporting element is not needed for classification. However, when the biological quality element indicates good or high status, a supporting element can downgrade the status to moderate. Therefore, when phytoplankton indicated good or high ecological status and phosphorus indicated a lower status, the overall status was set to moderate.

When interpreting the result, it should be noted that the WFD classification guidelines recommend classifying the ecological status on at least three years, and that the reference lake type and class intervals for the QEs are based on lakes with average depth >3m and a surface area > 0,5 km^2. Smaller water bodies like Teglverksdammen are normally classified as parts of a river. The BQE is however still useful in assessing the development in the pond, as the indices measure relevant parameters.

2.4.8 Assessment of the pond's potential for removal of nutrients and organic matter

To assess the potential for removal of nutrients and organic matter (measured as TOC) of Teglverksdammen and the adjoining wetlands, paired t-tests between the upstream and downstream concentrations were used. It was further tested through correlation tests whether the residence time or amount of phytoplankton influenced the change in concentrations.

3 Results

3.1 Development in Physio-chemical Conditions, Phytoplankton Total Biomass and Assemblages

3.1.1 Background data – Air temperature, precipitation and solar irradiation.

Monthly mean air temperatures and daily precipitation values are presented in Figure 3 and solar irradiation (SI) in Figure 4. Mean air temperature was slightly warmer than normal (1961-90) most months in the growth season 2016, and close to the normal in 2017.

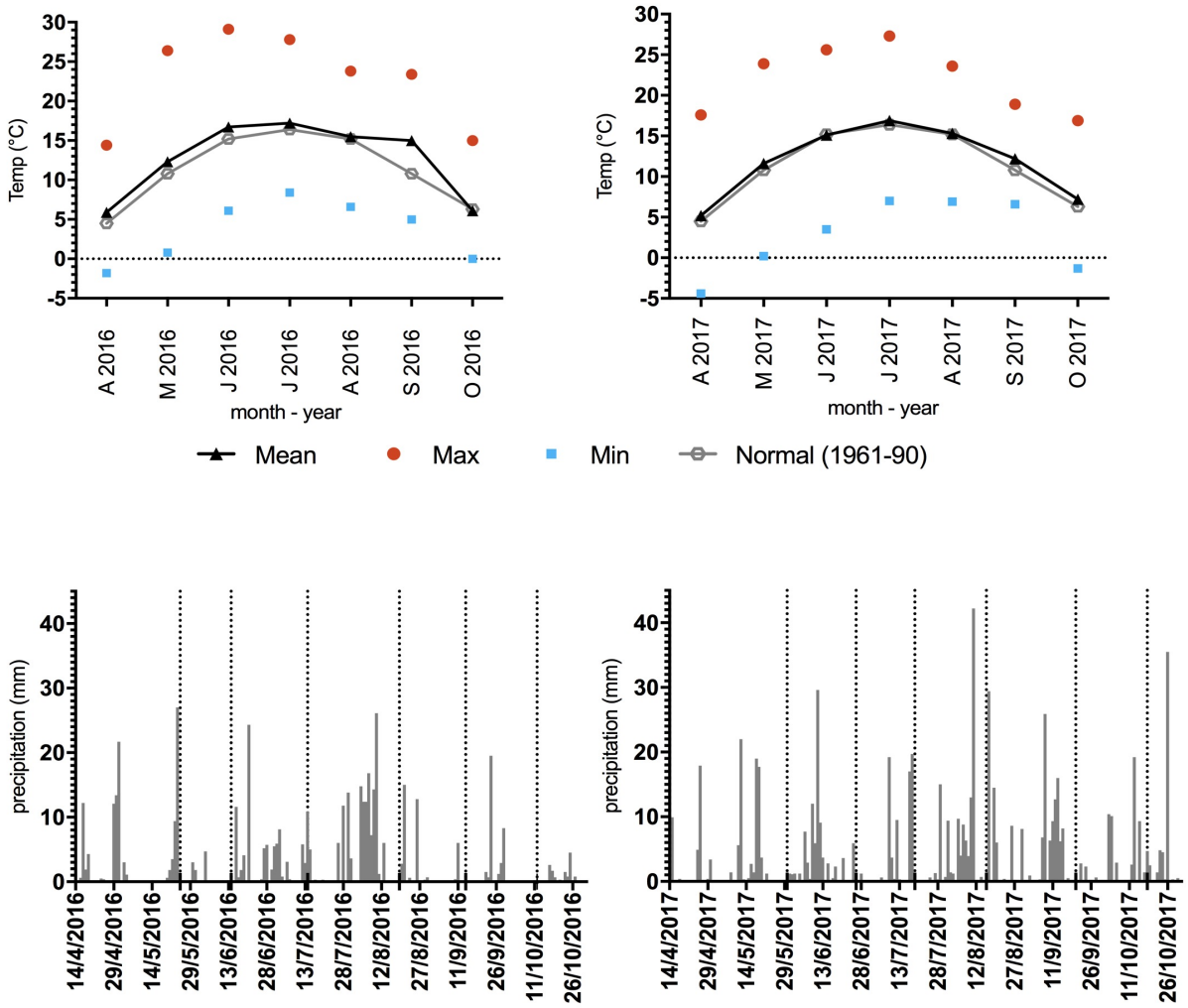

Figure 3. Mean temperature each month in the sampling season and daily precipitation. The mean temperature is shown together with monthly maximum and minimum, as well as the climate normal (1961-91) temperatures. The daily precipitation is shown from mid-April to end of October. Vertical dotted lines indicate sampling dates. Data from Blindern weather station (station no 18700), retrieved from: eklima.met.no.

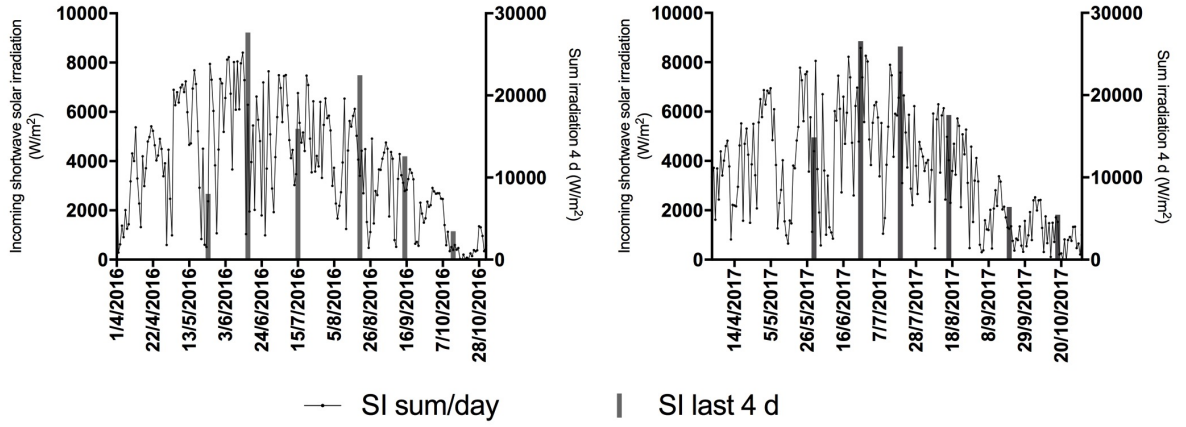

Figure 4. Daily incoming shortwave solar irradiation (SI), and accumulated SI the last 4 days at day of sampling (4x24 hours from 12 noon at day of sampling).

3.1.2 Stream discharge.

The stream discharge showed large variations throughout 2016 (Figure 5). The mean discharge for the whole period with recorded data was 0.038 m^3/s, or 0.058 m^3/s if the maintenance period (11.08.2016 – 12.09.2016) is excluded. The resulting pond residence times were shortest in the May to July samples, and longest in the August to October samples (precise values in Appendix E). The water table was also lower in September than the other months, by approximately 0.5 m. The global mean residence time, based on all water discharge data, was 3.70 days.

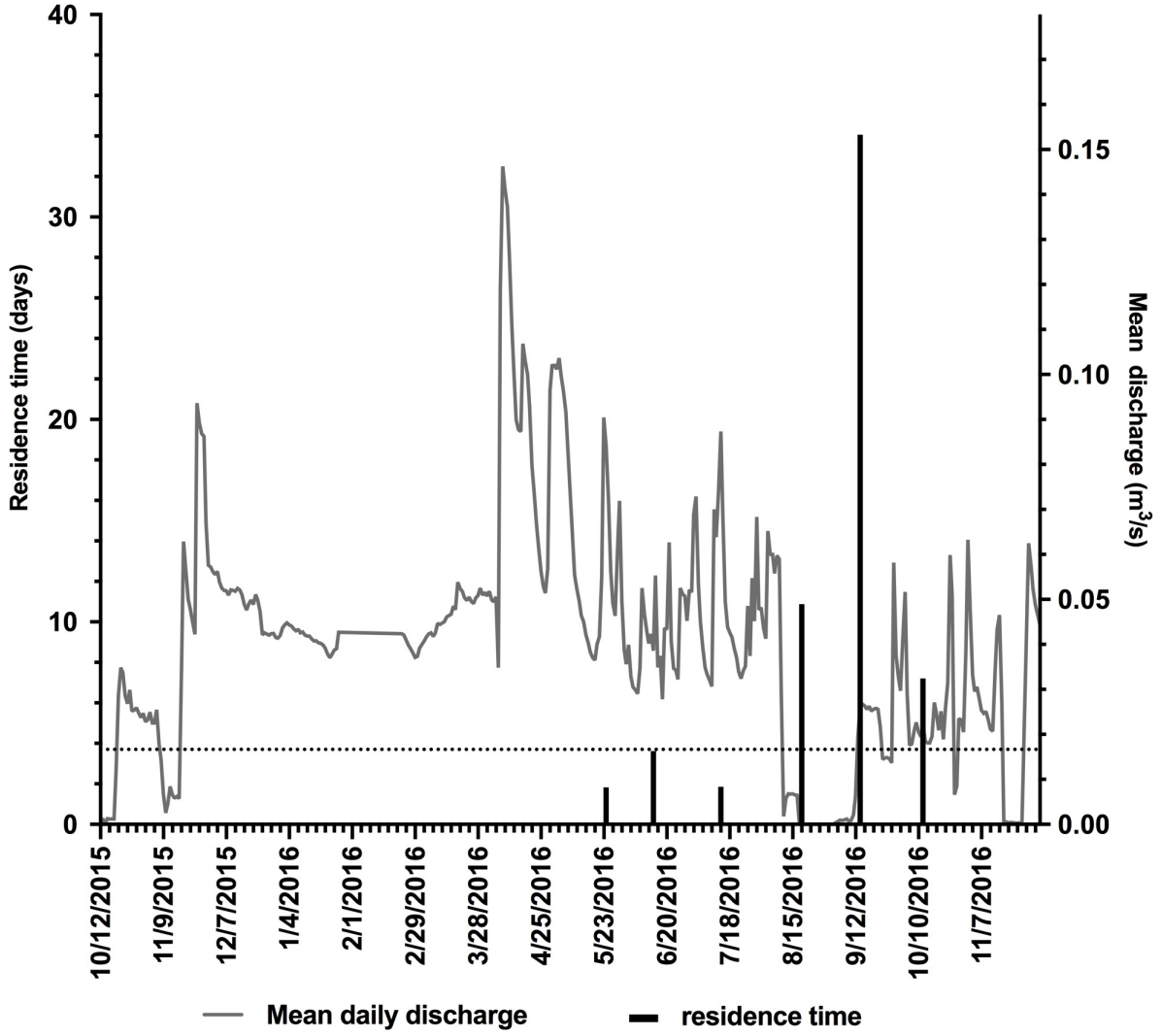

Figure 5. Mean daily stream discharge from opening to the end of 2016 shown together with the pond residence time at the time of sampling in 2016. The global men residence time for the pond, based on all stream discharge data, is shown as horizontal dotted line.

3.1.3 Water chemistry.

The water chemistry varied throughout the two sampling seasons without any apparent seasonality (nutrients, Figure 6)(Calcium, pH, TOC and conductivity, Figure 7). No seasonal patterns for ratios of dissolved nutrients to particulate-bound nutrients were observed either (precise values in Appendix F). However, the ratio of NO3-N to particulate nitrogen (PN) was lowest in the September 2016 sample (0.19), and next lowest in the August 2016 sample (0.91). In the other samples from both years, the ratios were all above 1.6. Further, the ratio of PO4-P to particulate phosphorus (PP) was also low in September and August 2016, but the variation outside these samples was much greater than for NO3-N:PN. The change in NO3-N:PN ratios during these months was not noted in the samples upstream (Appendix F).

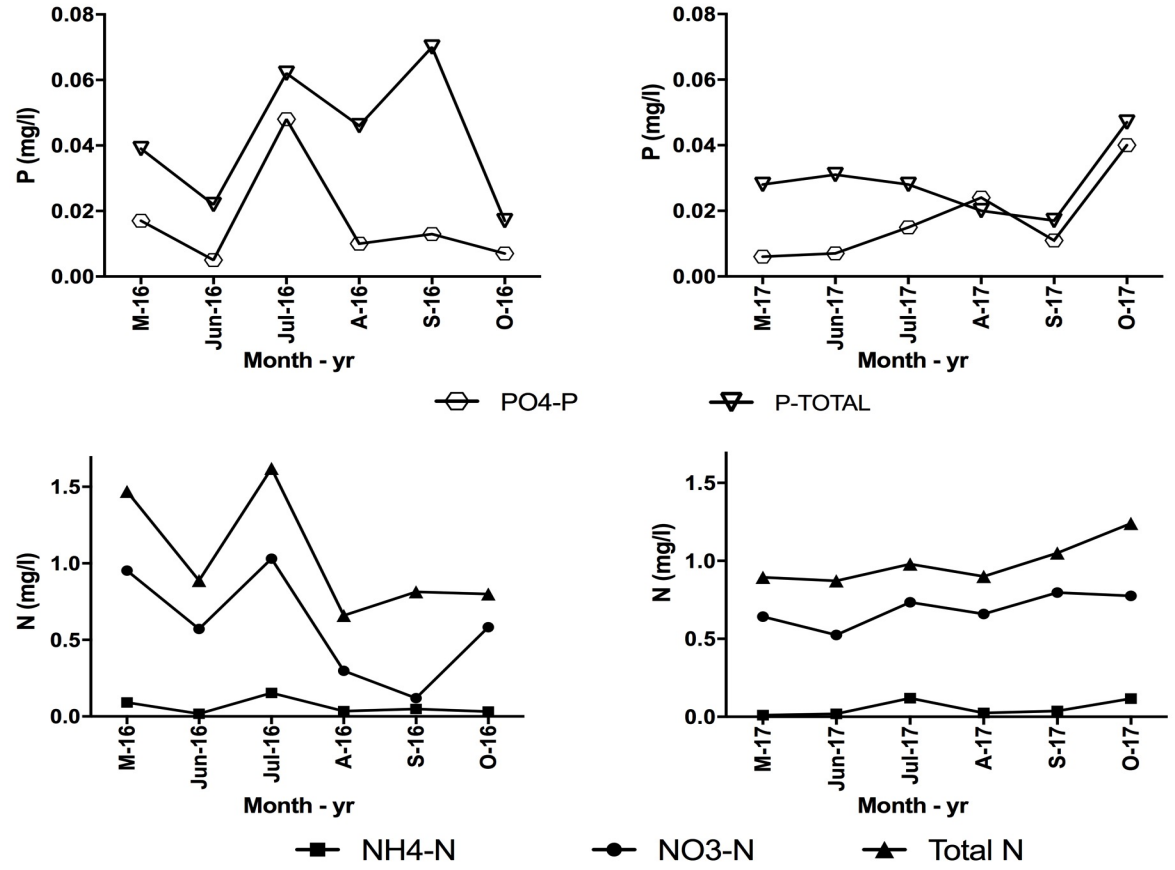

Figure 6. Total nitrogen and phosphorus, phosphate, nitrate and ammonium in Teglverksdammen (T6). The left graphs show the nutrient concentrations in 2016, and the right graphs the concentrations in 2017.

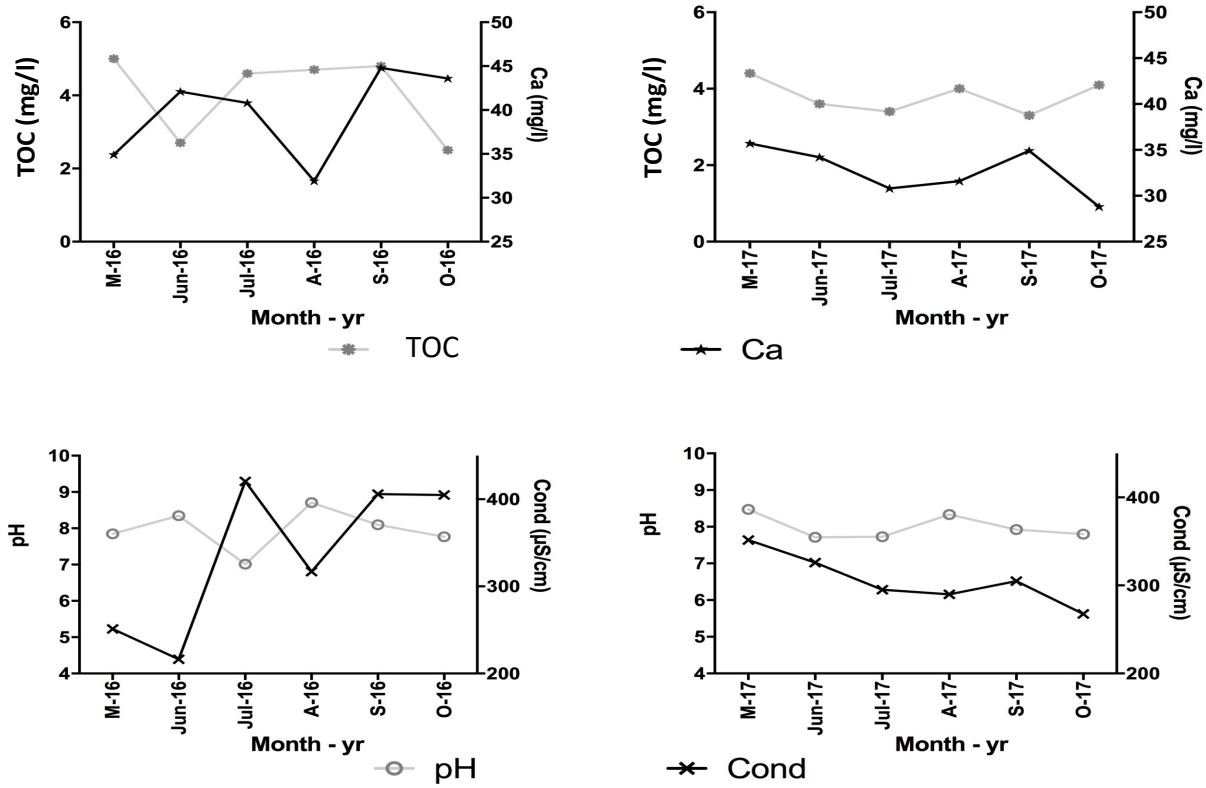

Figure 7. TOC, Ca, pH and conductivity in Teglverksdammen. The left graphs show the levels in 2016, and the right graphs the levels in 2017.

There was no significant difference in mean annual values for the water chemistry, except a small one for calcium (Table 2). However, nutrient and conductivity values were more stable the second year. The stabilisation of nutrient concentration was also noted in the samples upstream of Teglverksdammen (T5), although the concentrations were less stable than in the pond (Appendix G).

Table 2. Yearly means for water chemistry in Teglverksdammen, confidence intervals (CI) for means and Students t-test for difference in mean nutrient concentration

	unit	Mean 2016	CI for mean 2016	Mean 2017	CI for mean 2017	p for difference 2016-2017
TN	mg/L	1.04	0.62, 1.46	0.99	0.84, 1.14	0.770
NH4N	mg/L	0.06	0.01, 0.12	0.05	0.00, 0.11	0.780
NO3N	mg/L	0.59	0.22, 0.97	0.69	0.58, 0.80	0.547
TP	mg/L	0.04	0.02, 0.06	0.03	0.01, 0.04	0.183
P-PO4	mg/L	0.02	0.00, 0.03	0.02	0.00, 0.03	0.954
Ca	mg/L	39.68	34.2, 45.1	32.67	29.8, 35.5	0.019*
TOC	mg/L	4.05	2.86, 5.24	3.8	3.34, 4.26	0.631
Conductivity	µS/cm	336.0	243.7, 428.3	305.9	275.9, 336.7	0.455
pH		7.97	7.64, 8.34	7.99	7.65, 8.33	0.920

Note: * Significant at a 5% test level, but not after Bonferroni correction.

The water chemistry variables in the pond showed no significant correlation with the residence time in 2016 after Bonferroni correction (Appendix H). For NO3-N however, there was a negative correlation with p-value <0.05 (correlation= -0.819, p=0.046) with the residence time. This also held after the influential September sample (Di>2) was removed (-0.906, p=0.034). However, if both samples with longer residence times due to the maintenance period was removed, the p-value was much higher.

Nitrate also correlated with the mean daily discharge at the upstream site (T5) (correlation= 0.892, p=0.017), and this was also near-significant (correlation= 0.859, p=0.062) when the August 2016 sample taken when there was only minimal flow (due to the inlet-vent being closed) was removed. Total nitrogen also correlated with the mean daily discharge at the upstream site (T5) (correlation= 0.915, p=0.011) (August 2016 sample removed: correlation= 0.865, P=0.058).

3.1.4 Temperature, oxygen and conductivity profiles.

The gradients for temperature, conductivity and oxygen saturation indicate that some stratification occurs in the pond (Figure 8 and Figure 9). Only October 2016 and September 2017 show homogenous oxygen, temperature and conductivity profiles. In the other months, the temperature profile varies from near-homogenous (May and September 2016), to showing a steep gradient between an upper and lower temperature-separated stratum (June and August 2016) to more gradual or complex (June and July 2017). From June to September in 2016 and May to August in 2017 there was an overall decrease in oxygen saturation and increase in conductivity with depth in the deeper strata. In 2016, the registered oxygen saturation near the bottom was lower for each consecutive month up until August when it reached a minimum below 10%. The following year, oxygen saturation was below 10% already in May, and stayed low until August. For both years, the lowest registered oxygen levels in the deepest stratum (August 2016 and June 2017) would indicate the ecological status bad if used as a supporting QE for the BQE fish (Appendix I). The May 2016 profile showed another pattern all together, with little temperature change with depth but a sharp gradient for increasing oxygen saturation and conductivity around two meters depth.

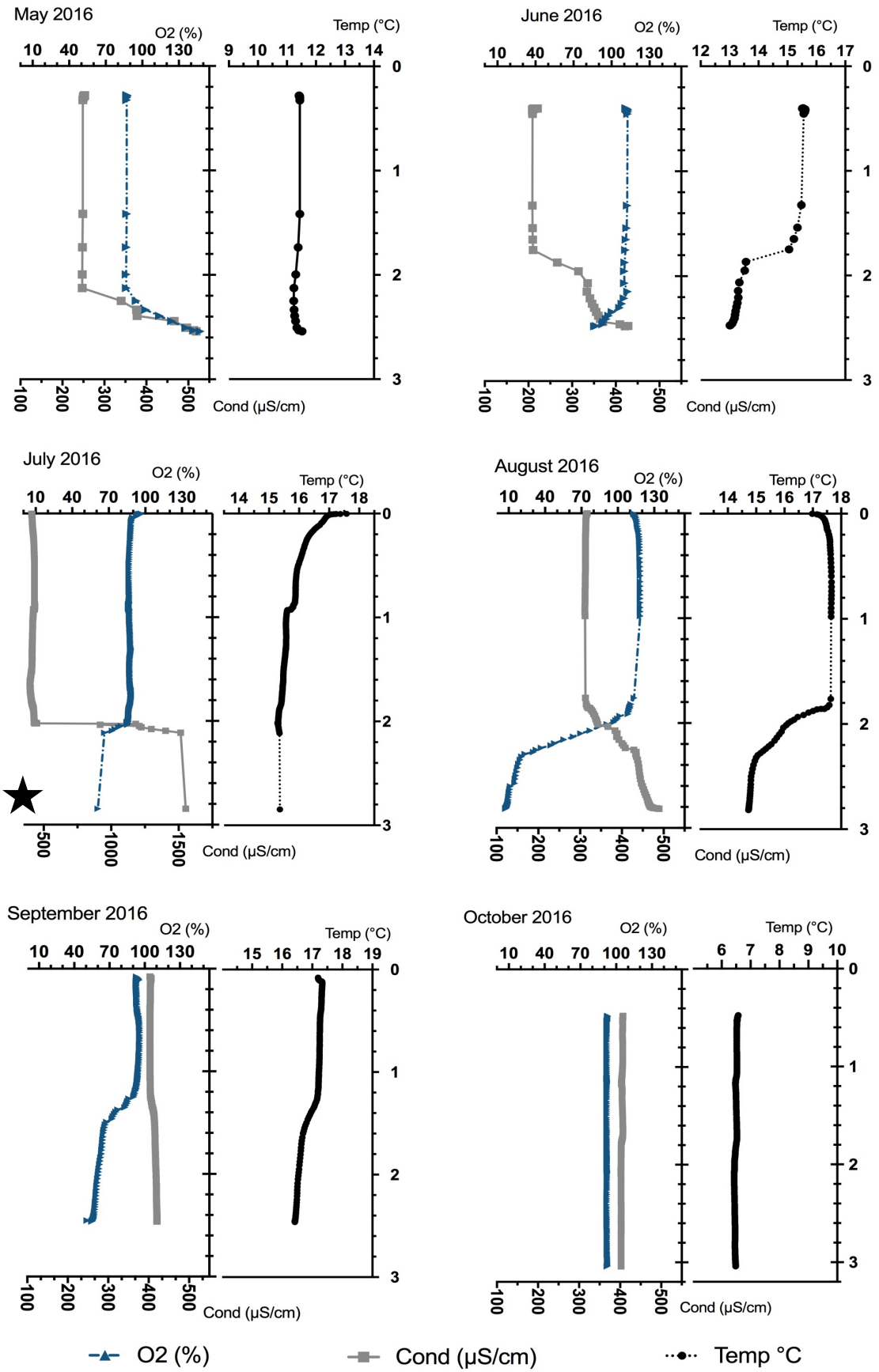

Figure 8. Temperature, oxygen saturation and specific conductivity profiles from the pond (T6) 2016. Note the different scale on conductivity in the July profile. A second profile recorded in July is shown in Appendix J.

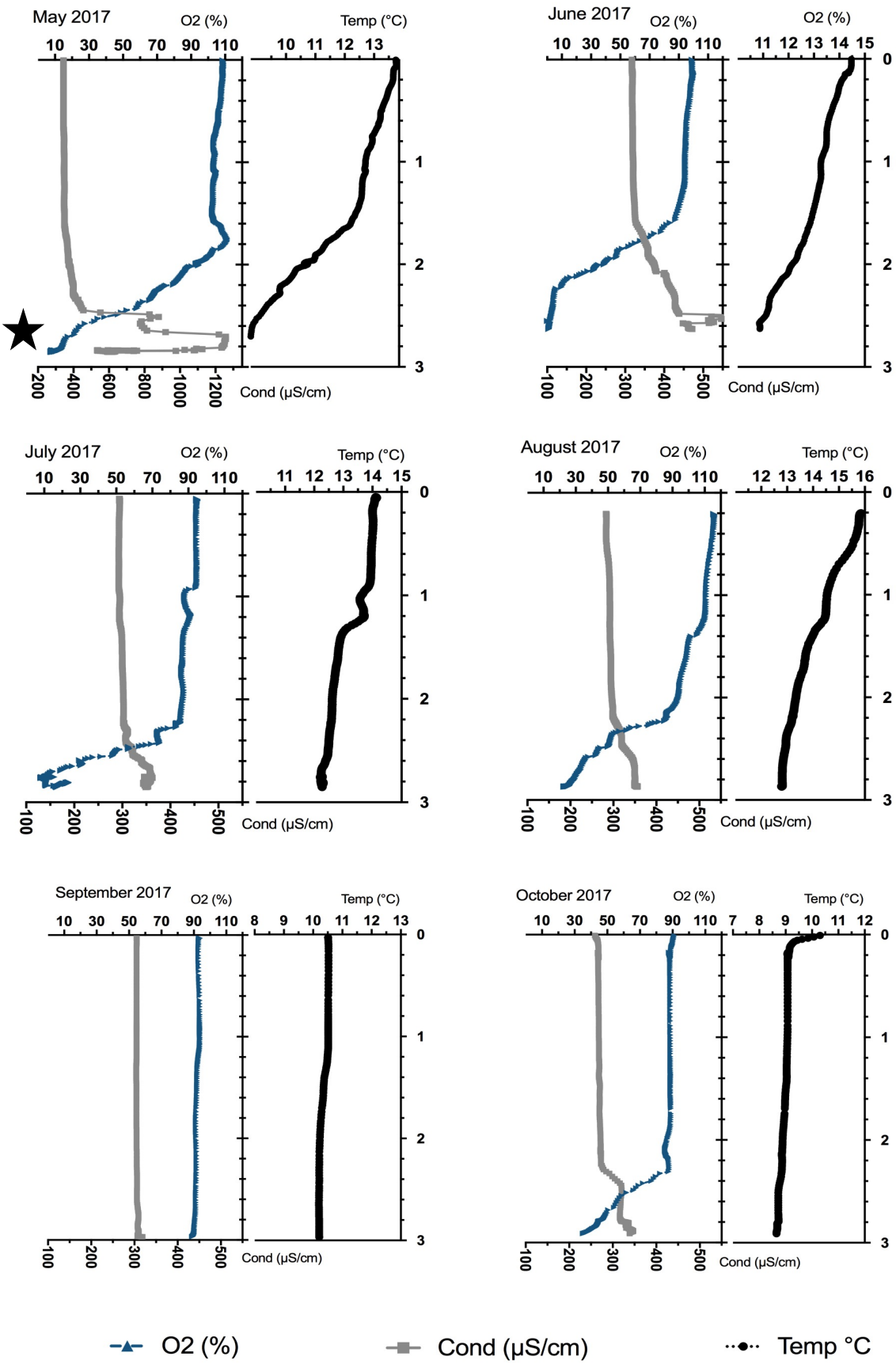

Figure 9. Temperature, oxygen saturation and specific conductivity profiles from the pond (T6) 2017. Note the different scale on conductivity axis for May profile.

3.1.5 Water Transparency and Colour.

The water transparency varied from very clear with a secchi depth equal to the depth of the pond, turbid and a secchi depth of only 0.4 m (Table 3). The water colour mostly varied from clear to very grey. The water was also slightly brown in July 2016, and a hint of green was also registered in the samples in June and September 2016, the same two months that also had the highest phytoplankton biovolume. There was no correlation between the residence time and the turbidity in the pond.

Table 3. Water colour, secchi depth and turbidity. Secchi depths >3m indicate that the secchi disc was still clearly visible at the depth of the pond.

	Secchi depth m	Water colour	Turbidity [b] FNU
May 2016	0.6	Grey	17.90
June 2016	1.5	Grey, a little green	4.45
July 2016	0.4	Grey, a little brown	27.53
August 2016	>3.0	Clear	1.83
September 2016	1.2	Clear, little grey + green	3.65
October 2016 [a]	>3.0	Clear	1.24
May 2017	1.9	Clear	3.13
June 2017	1.6	Clear	4.01
July 2017	0.9	Clear (+ hint of grey)	9.41
August 2017	>3.0	Clear	2.03
September 2017	3.0	Clear	2.60
October 2017	0.4	Grey	24.49

a) Clear, but the water just entering the pond was dark-grey, and heavily clay loaded

3.1.6 Total Phytoplankton Biovolume Concentration.

The total phytoplankton biovolume concentration showed large variations throughout the two growth seasons (Figure 10). In 2016 there was a peak in productivity in June and a larger in September, the sample taken after almost a month with no or minimal inflow to the pond. In 2017 the phytoplankton production was highest in the start of the growth season, with a peak in June and then gradually flattening out.

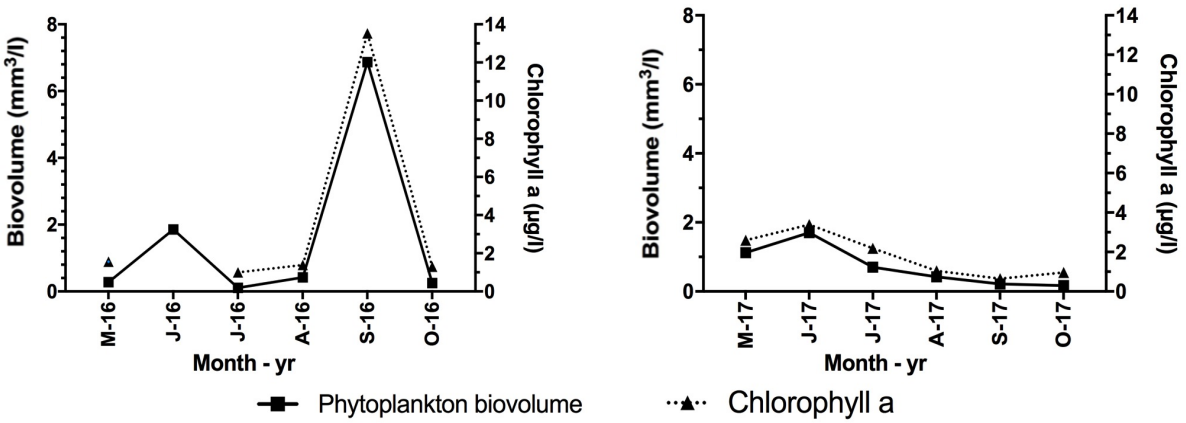

Figure 10. Total biovolume and chlorophyll a for 2016 and 2017. The peak in September 2017 is the sample taken at the end of the stagnation period.

There was a strong significant correlation between phytoplankton biovolume and the chlorophyll a (correlation =0.997, p = 2.409e-11). This also held when the September sample (D_i>2.5) was removed (correlation=0.950, p=2.493e-05).

The pond's trophic state as indicated by annual mean and maximum phytoplankton biovolume was polyeutrophic in 2016 and mesotrophic in 2017. When the 2016 September sample was taken out, the indicated trophic state was mesotrophic also for 2016 (see figure in Appendix K). However, the difference in mean biovolume the two years was not significant (see also chapter 3.2.1).

3.1.7 Relationship between environmental variables and Phytoplankton Biovolume.

Biplots from PCAs on standardised variables visualize how the phytoplankton biovolume concentration was associated with growth related variables (plot B, all 12 samples) as well as residence time (plot A: samples from first year) (Figure 11). Nutrient concentrations, temperature, secchi-depth, turbidity and incoming solar irradiation the four last days (SI4d) is represented as well as the pond residence time (plot A). Plot A indicates phytoplankton biovolume concentration was positively associated with the residence time, SI4d and temperature. Plot B, on all data points but excluding residence time also indicates a positive association with SI4d and temperature. Both plots indicate a negative association between with phytoplankton and NO3-N. It should be noted both plots also indicate the September 2016 (S16) sample may have been influential for these associations.

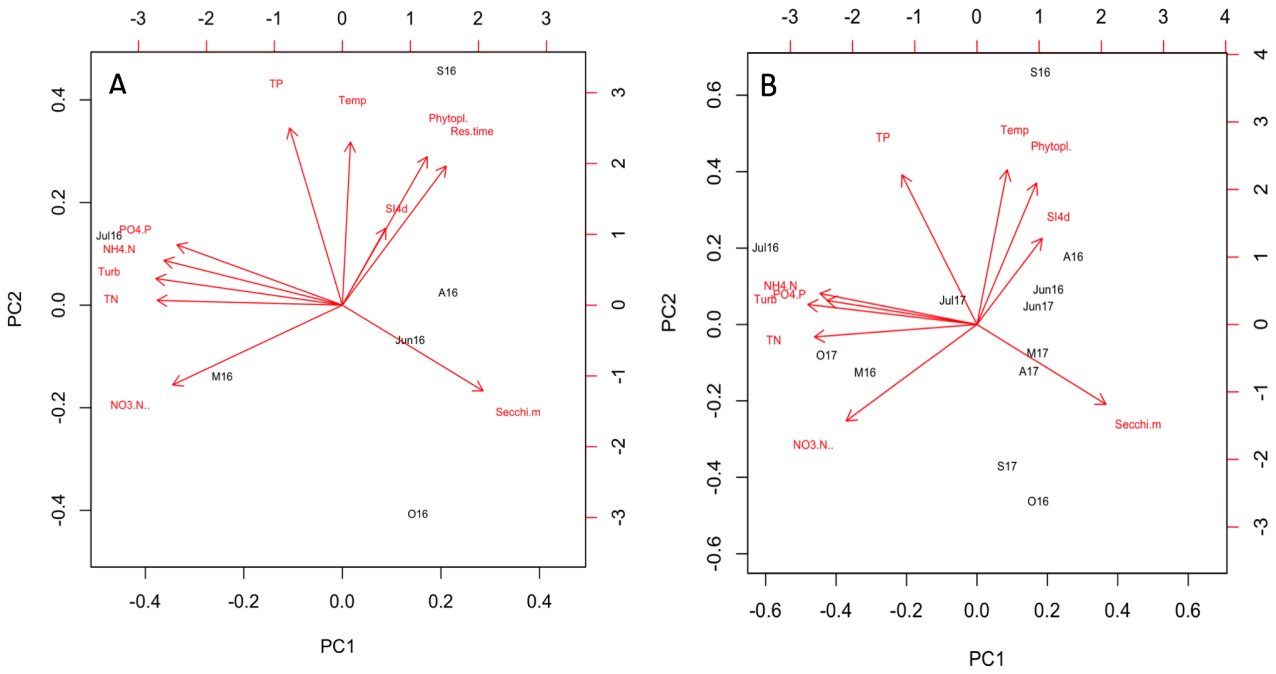

Figure 11. Biplots for the PCAs on standardized variables visualize how the phytoplankton biovolume concentration (Phytopl.) was associated with growth related variables (plot B) as well as residence time (plot A). The black marks represent each sample, and the red arrows relative loading on the different components. The cumulative proportion of variation captured by the first two components is 80.7%(A) and 76.7% (B). SI4d: cumulative incoming solar irradiation previous 4 days. Turb: turbidity. Temp: temperature, Res.time: Residence time.

The phytoplankton biovolume concentration did not significantly correlate with these variables individually after Bonferroni correction. There was however an initial negative correlation between NO3-N and phytoplankton concentration with $p<0.05$ (Table 4) , but when the influential September 2016 sample ($D_i > 3.5$) was removed, the correlation yielded much higher p-value. There was further an initial correlation with the residence time in 2016 (correlation=0.923, p=0.008), but when the influential September 2016 data point ($D_i>15$) was removed this was not significant (p=0.89). If the growth related variables were further split into the two years, NO3-N and phytoplankton biovolume also showed a negative correlation with $p <0.05$ in 2017 (correlation = -0.906, p=0.0127)(not significant after Bonferroni correction).

Table 4. Phytoplankton Biovolume Correlations with Growth Related Environmental Variables. TP : total phosphorus, TN: total nitrogen. Turbidity, and temperature are integrated values from the sonde profile. SI 4 days: incoming solar irradiation 4days before sampling.

	correlation	p-value
Temperature	0.4614	0.1311
TP	0.5056	0.0936
PO4-P	-0.2635	0.408
NO3-N	-0.7196	0.0083
TN	-0.3435	0.2743
NH4-N	-0.2310	0.4701
Secchi	-0.1512	0.6391
Turbidity	-0.2841	0.3708
SI 4 days	0.1710	0.5951

For the nutrient salts ratios to particulate-bound nutrients, there were no significant correlations with phytoplankton biovolume (Appendix L). There was however a near-significant negative correlation between phytoplankton biovolume and the NO3-N:PN ratio (correlation = -0.5403, p= 0.0698).

For phytoplankton biovolume and nutrient concentrations at the upstream site, no significant correlation was found (Appendix M). Splitting the data into the two seasons did not yield any significant correlations either.

3.1.8 Phytoplankton assemblages in Teglverksdammen.

There was large variation in phytoplankton assemblages (Figure 12) and the seasonal development in phytoplankton total biovolume and assemblages was different the two years. The global mean fraction of total biovolume was largest for the phyla *Cryptophyta* followed by *Bacillariophyta* and then *Ochrophyta*, the fraction for unknown phytoplankton excluded. The large fraction of unknown phytoplankton in May and July 2017 to a large extent consisted of phytoplankton with a diameter of <4 µm.

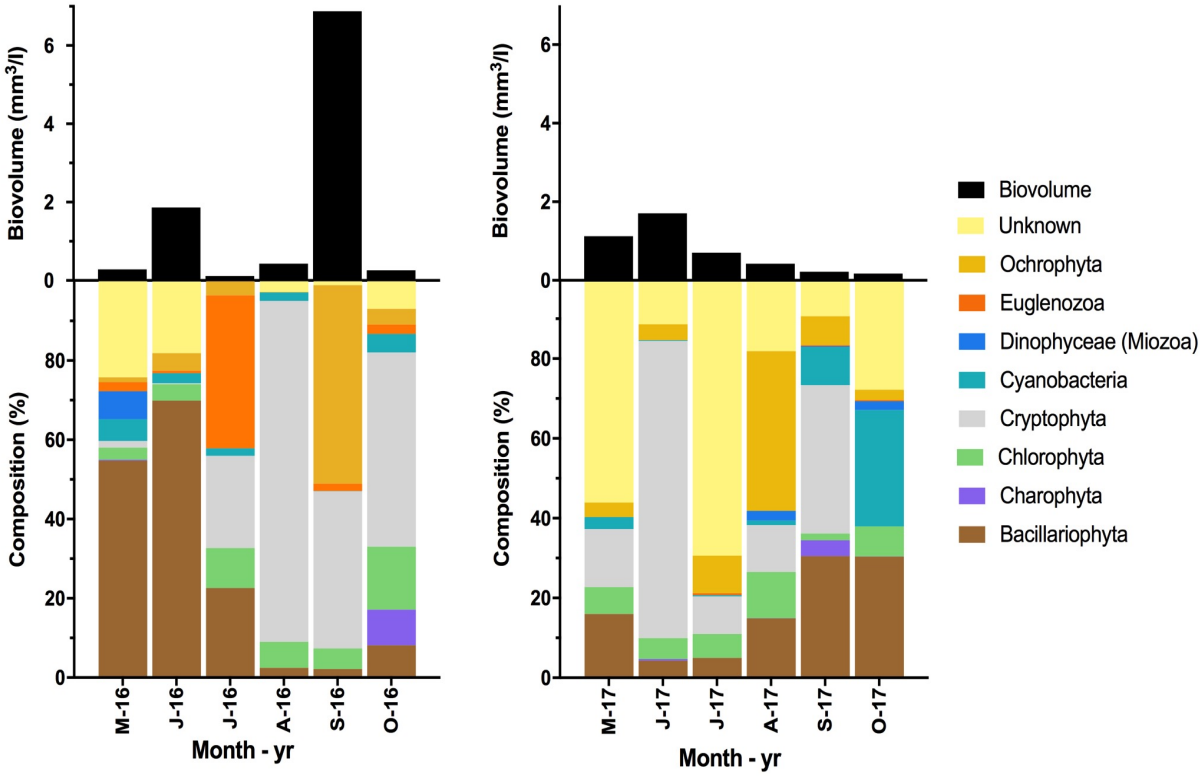

Figure 12. Phytoplankton biovolume and relative composition of the different phyla.

Cryptophyta were dominant in August 2016 and June 2017, representing 86% and 75% respectively. The phylum also represented 49% and 40% respectively in October and September in 2016. No significant correlation was found between the phylum and the environmental variables after Bonferroni correction (Appendix N), but correlations that yielded $p<0.05$ were with NO3-N (correlation= -0.735, p=0.007), and residence time (correlation= 0.985, p=0.0003). The p-value for the correlation with residence increased to 0.01 when the influential sample (September 2016: $D_i > 20$) was removed. For NO3-N, the September 2106 sample was only semi-influential measured by Cook's distance alone (D_i=2.9) but the residual vs leverage further indicated high influence. When the sample was taken out, the p-value was higher (p=0.15).

Bacillariophyta was dominant in May 2016, representing 55% of the total biovolume. The same phylum dominated also in June before the fraction decreased throughout the summer months. In May the following year the fraction of *Bacillariophyta* was lower, only 16%. The phylum also increased in September 2017, when the homogenous pond profiles indicated that the pond was well-mixed. No significant correlation was found between the phylum and the environmental variables after Bonferroni correction, nor any correlations that yielded a p-value<0.05.

The *Ochrophyta* fraction represented 50% of the biovolume in September 2016 of which 99% was *Mallomonas sp.,* and 40% in August 2017, of which 83% was *Chrysophyceae*. Both samples are from late summer to the beginning of autumn. No significant correlation was found between the phylum and the environmental variables after Bonferroni correction, but two correlations yielded initial p-values<0.05, NO3-N (correlation= - 0.652, p=0.022) and residence time (correlation=0.957, p=0.003). This did not hold up when the influential samples were removed (September 2016: $D_i > 4$ for NO3-N, $D_i > 25$ for residence time).

The *Cyanobacteria* fraction was largest in October 2017, representing 29%. The next largest fraction, 10% was recorded in the month before. No significant correlation was found with environmental variables after Bonferroni correction, nor any correlations that yielded a p-value<0.05.

3.1.9 Observations of algal growth in Teglverksdammen, not represented in pelagic sample.

Large dark patches (approximately 1-10cm in diameter) of floating algae were observed in the restored reach and the pond throughout the two growth seasons (Figure 13) to a varying extent (Table 5). A few of these patches were sampled and the algae was identified as the cyanobacterium *Oscillatoria sancta*. *O. sancta* from the pond was also cultivated and tested negative for microcystin at NIVA (Personal communication Birger Skjelbred, 2017). The patches were frequently observed as far up as the stream as the second sedimentation-pond below the inlet to the restored reach.

Figure 13. Patches of Oscillatoria in the pond and restored reach. Floating (left) and detaching from the bottom in the littoral zone (right). Pictures by Therese Fosholt Moe.

Table 5. Floating patches of Ocillatoria sancta in the restored reach and in the pond. A rough estimate where 0 = no patches visible, 1 = patches visible, 2 = plenty of patches visible.

	The whole restored reach	Teglverksdammen
May 2016	2	2
June 2016	1	1
July 2016	1	1
August 2016	2	2
September 2016	2	1
October 2016	1	1
May 2017	1	0
June 2017	2	1
July 2017	1	0
August 2017	1	0
September 2017	1	1
October 2017	1	1

In additional samples of *O. sancta* patches collected April 2017 near the inlet of the pond and in the littoral zone near the pond station (T6), several genera of phytoplankton associated with marine environments were found. These include *Thalassionema sp., Skeletonema sp., Scrippsiella sp., Rhizosolenia sp., Protoperidinium sp., Ditylum sp.* and *Chaetoceros sp.*

There were also a few events with extensive growth of filamentous green algae, *Spirogyra majuscula,* in the pond and adjacent wetlands. Extensive growth of the algae was first noted in the wetlands in July 2016. Park maintenance workers were also seen harvesting the algae masses during the fieldwork this month. The following month, August 2016, the algae was noted in the pond. At the time of sampling, there had been a stagnation period with no to minimal inflow over the duration of a week, the pond water was clear, and the bottom of the pond was covered with a thick mass of the filamentous green algae. The mass reached up approximately 1 m from the bottom in the deepest part of the pond. At the shorelines, the masses reached up to 0,5-1 m below the surface, where there was a clear divide to substrate with no algae attached (Figure 14). There was also extensive growth in the wetland below the pond this month. The following month only very small amounts remained of the algae mass in the pond, but some remained in the adjacent upstream wetland. No *Spirogyra* species were found in either of the pelagic samples in August and September 2016. Filamentous green algae were also noted growing in the restored reach in 2017, from July to September, but only in the wetlands and the last sedimentation pond.

Figure 14. Extensive growth of filamentous green algae, covering the bottom of the pond and approximately the bottom 1 m of the water column in the deepest part, and up to 0,5-1 m below the surface near the shoreline.

3.1.10 Relevant field observations.

A few field observations are worth noticing. Throughout both sampling seasons there were plenty of birds in the restored reach and in the pond in particular. It was common to observe over 40 birds in the pond during sampling. Second, a grab sample in the autumn of 2016 showed that only very little sediment had accumulated on the rocky substrate in the pond. The little sediment that was there, appeared to be predominantly organically derived. Last, fish was observed in the reach already in 2016. The types and amounts of fish is unknown, but members of the public did on several fieldwork trips also report having observed fish in the reach.

3.2 Ecological Status, WFD Classification

3.2.1 Phytoplankton.

The biological quality element "phytoplankton" indicated moderate ecological status in 2016 (Table 6) and good ecological status in 2017 (Table 8) . Changes in the individual indices were however not significant (Figure 15). Removing the September 2016 sample improved the indicated class of the biovolume index for the year from moderate to high. However, the status for 2016 remained moderate (Table 7). The cyanobacteria biovolume per litre never exceeded limits that would indicate a lower ecological status than high (0.16mg/l) and was in accordance with procedure not included in the combined nEQR for phytoplankton either year.

Table 6. 2016 Phytoplankton Indices for Determining Ecological Status in Teglverksdammen 2016 according to the WFD standard. Yearly values are mean for biovolume and PTI indices and maximum value for cyanobacteria biovolume.

	Total Biovolume mm^3/l	PTI	Cyanobacteria Biovolume mm^3/l
Yearly	1.631	2.708	0.047
EQR	0.764	0.676	0.995
nEQR	0.508	0.316	0.953
Ecological Status indicated	Moderate	Poor	High
nEQR Phytoplankton Ecological Status indicated			0.41 Moderate

Table 7. 2016a - Phytoplankton Indices for Determining Ecological Status in Teglverksdammen according to the WFD standard. - The influential September 2016 sample removed. Yearly values are mean for biovolume and PTI indices and maximum value for cyanobacteria biovolume.

	Total Biovolume mm^3/l	PTI	Cyanobacteria Biovolume mm^3/l
Yearly	0.585	2.770	0.047
EQR	0.947	0.644	0.995
nEQR	0.823	0.268	0.953
Ecological Status indicated	High	Poor	High
nEQR Phytoplankton Ecological Status indicated			0.54 Moderate

Table 8. 2017 Phytoplankton Indices for Determining Ecological Status in Teglverksdammen according to the WFD standard. Yearly values are mean for biovolume and PTI indices and maximum value for cyanobacteria biovolume.

	Total Biovolume mm^3/l	PTI	Cyanobacteria Biovolume mm^3/l
Yearly	0.723	2.426	0.050
EQR	0.923	0.824	0.995
nEQR	0.751	0.609	0.950
Ecological Status Indicated	Good	Good	High
nEQR Phytoplankton			0.68
Ecological Status indicated			Good

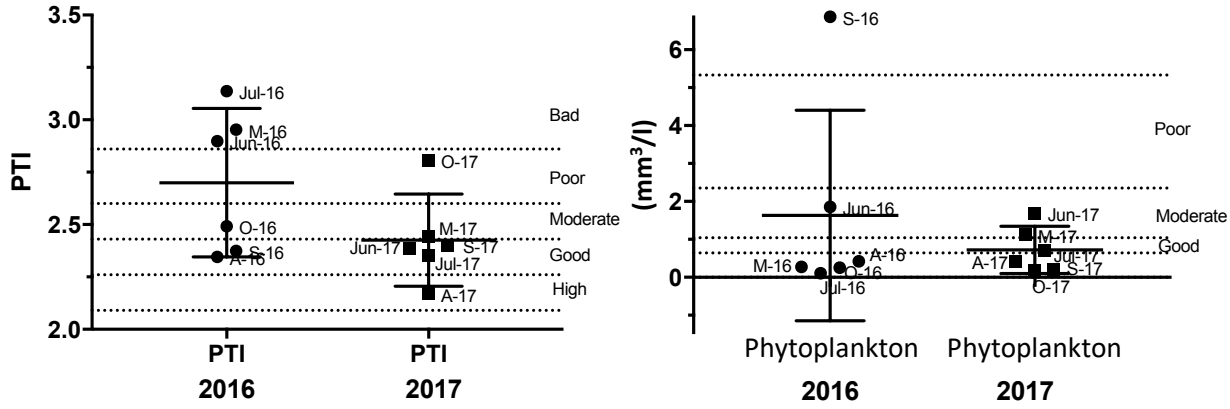

Figure 15. PTI and phytoplankton biovolume mm^3/l 2016 and 2017. Graphs show the mean PTI and phytoplankton biovolume concentration as the long middle bars, and the 95% confidence interval for the mean.

3.2.2 Nutrients.

According to the WFD class intervals for the water type, mean levels of total phosphorus indicate bad and poor ecological status in 2016 and 2017 respectively (Figure 16). The indicated class for total nitrogen was moderate both years.

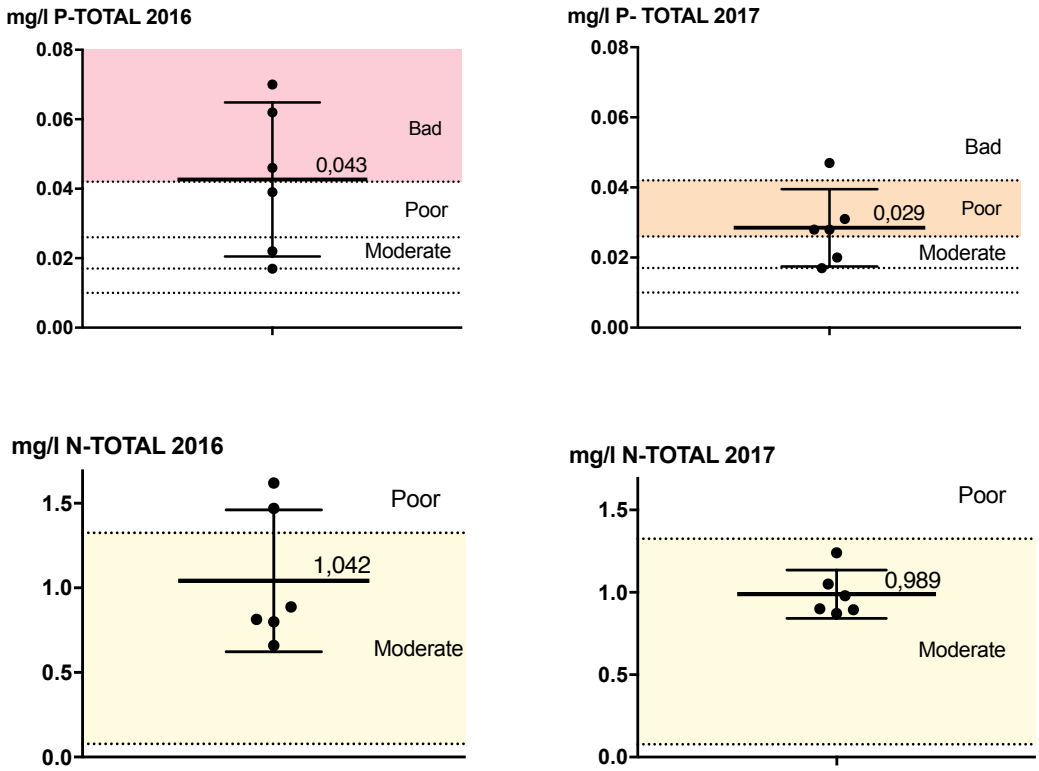

Figure 16. Total nitrogen and phosphorus 2016 and 2017. Graphs show the mean concentrations as the long middle bar and number, the 95% confidence interval for the mean, as well as the indicated ecological class as the coloured interval.

3.2.3 Overall ecological status.

The indicated ecological status was moderate for both years separately, as total phosphorus lowered the overall status for 2017. The ecological status for both years combined was moderate as indicated by the mean phytoplankton alone (nEQR =0.55).

1.1 Retention of Organic Matter and Nutrients

There was large variation in whether the TOC and nutrient concentrations increased or decreased downstream the pond, but the general trend indicate retention. After Bonferroni correction, there was no overall significant difference between upstream and downstream nutrient or TOC concentrations, but the p-values for reduction was 0.034 for NO3-N and 0.062 for PO4-P (Table 9). In the seasonal tests, NO3-N in spring also gave a p-value of 0.031 for reduction.

Table 9. Nutrient removal by the Teglverksdammen pond and adjacent wetlands, paired t-tests and confidence interval (CI) for difference. All samples: all samples in the growth season. Spring samples: May-June. Summer samples: July-August. Autumn samples: September-October (without September sample 2016).

	Nutrient	Mean difference upstream-downstream mg/l	95% CI for difference mg/l	p-value	
All samples	TOC	0.718	-1.498, 2.205	0.471	
All samples	NO3-N	0.139	0.011, 0.336	0.034	*
All samples	TP	0.028	-0.014, 0.052	0.100	
All samples	TN	0.131	-0.095, 0.275	0.185	
All samples	NH4-N	0.031	-0.040, 0.084	0.371	
All samples	PO4-P	0.010	-0.003, 0.017	0.062	
Spring samples	TOC	1.975	-5.601, 9.551	0.468	
Spring samples	NO3-N	0.238	0.040, 0.435	0.031	*
Spring samples	TP	0.017	-0.045, 0.079	0.445	
Spring samples	TN	0.250	-0.402, 0.902	0.309	
Spring samples	NH4-N	0.012	-0.192, 0.216	0.861	
Spring samples	PO4-P	0.005	-0.010, 0.020	0.385	
Summer samples	TOC	-0.850	-2.203, 0.503	0.140	
Summer samples	NO3-N	0.047	-0.353, 0.446	0.736	
Summer samples	TP	0.022	-0.039, 0.083	0.335	
Summer samples	TN	0.023	-0.354, 0.399	0.860	
Summer samples	NH4-N	0.093	-0.082, 0.267	0.191	
Summer samples	PO4-P	0.016	-0.021, 0.052	0.266	
Autumn samples	TOC	1.133	-4.430, 4.780	0.537	
Autumn samples	NO3-N	0.133	-0.280, 0.890	0.280	
Autumn samples	TP	0.054	-0.115, 0.171	0.412	
Autumn samples	TN	0.117	-0.270, 0.471	0.545	
Autumn samples	NH4-N	-0.027	-0.095, 0.065	0.467	
Autumn samples	PO4-P	0.009	-0.012, 0.024	0.350	

*Significant at a 5% significance level

The residence time and phytoplankton biovolume did not show any significant correlation with the difference in concentration of nutrients or TOC upstream and downstream of the pond (Appendix P). Neither was there any overall trend indicating that these variables affected the difference in concentrations.

4 Discussion

4.1 Development in Phytoplankton Total Biovolume and Assemblages, and the Relationship with Physio-Chemical Variables

4.1.1 Flow conditions.

There was large variation in the residence times at time of sampling. The maintenance period in 2016 resulted in an approximately 10-fold increased pond residence time in September, and three-fold increased residence time in August, compared to the global mean residence time (3.7 days). At the time the first three samples were taken, the residence time was marginally below the mean with <2 days in May and July and 3.6 days in June.

The residence times in this study are based on streamflow measurements and theoretical pond volume only and may therefore hold some errors. Evaporation is not accounted for here, which may have been an important factor in the warmer months with longer residence times. It should further be noted that there were indications of strata forming in the pond (discussed further in chapter 4.1.3.) and that the top part of the water column therefore may have been exchanged faster than the residence time would indicate.

The effects of low to minimal flow periods on the reach could be of interest for management purposes as there will likely be need for maintenance work on this and similar systems also in the future. There was a substantial increase of algae in the pond during the minimal flow period in 2016. The sample taken during the longest residence time, September 2016 had a phytoplankton biovolume approximately 12 times greater than the yearly mean without that sample. This sample increased the trophic state as indicated by mean and maximum phytoplankton biovolume by two classes in 2016. Peaks in phytoplankton production in spring and autumn are often observed in temperate lakes, but are usually associated with diatom blooms during circulation (Wetzel, 2001). August 2016 had the second longest residence time but did not show the same increase in planktonic algae. There was however extensive growth of filamentous green algae, occupying a large part of the water column. It should be noted that the water temperature also was high these months, but not much higher than July the same year. These results indicate that at least at some threshold, longer residence times can allow for increase in algae growth in the pond (discussed further in chapter 4.1.5.).

Two previous studies mention the ecological effect of the low to minimal flow period in the restored reach. A study by Myrstad (2017) on benthic algae showed that the composition of benthic algae in the restored reach changed after the dry period, but could not conclude whether this was due to seasonal changes or the disturbance. Arnott (2016) study on macroinvertebrates in the stream observed little to no effect on either biodiversity or population sizes after the low to minimal flow period, but further stated that this might be due to species assemblage consisting of mostly non-sensitive taxa. Overall, these studies could find little effect of the low to minimal flow period on the stream part of the restored reach. The results from this study however imply that stagnation periods could indeed have undesirable ecological effects on the pond part of the reach, in form of increase in total algal volumes. Further assessments of the impact of no flow periods that could be considered is the effect on fish. The feasibility study mentions that measures to prevent fish death during maintenance periods were likely unnecessary the first years as the initial fish populations were thought to be negligible (Norconsult, 2013). Since fish however were observed throughout the reach already the first year (see chapter 3.1.9), this might be something that could be assessed further.

4.1.2 Water chemistry.

The water chemistry varied without any seasonal patterns in Teglverksdammen and there was no significant change in the annual mean concentrations for the water chemistry. The exception being for calcium which was marginally lower in 2017.

The nutrient concentrations were slightly more stable the second year, as were calcium concentrations and conductivity. The stabilisation of nutrients in the pond in 2017 is likely a result of the more stable nutrient inputs, as the upstream samples showed a similar pattern.

Few impacts of flow conditions on the water chemistry in the pond were found, but there was a trend indicating a negative relationship between NO3-N and residence time. This is likely directly linked to the input since NO3-N correlated positively with mean daily discharge at the upstream site. The inlet vent should be fairly responsive to flow-conditions upstream, and therefore also to precipitation and runoff patterns in the catchment when the vent is operating normally (Personal communication Oslo VAV). When the August 2016 sample was taken, the streamflow was low due to the inlet-vent being almost closed. There had however also been little precipitation the week before sampling. When this sample was removed, the correlation between NO3-N and stream discharge was only near-significant ($p=0.06$). Leaching of NO3-N could be one explanation for the correlation. Positively charged

nutrient salts tend to be associated with negatively charged sites on clay minerals, and PO4-P adsorbs to minerals (VanLoon & Duffy, 2005). Nitrate mostly exists as a free salt and can therefore be more easily leached than other nutrients after periods with high rainfall. However, Krystad (2017) showed that there were higher concentrations of *E. coli* in the inlet of the restored reach after heavy rain. Stahl and May (1967) further states that wastewater can contribute to high rates of nitrifying bacteria. More influx of wastewater is therefore likely the most important explanation for correlation with streamflow. Total nitrogen also showed a correlation with streamflow at the upstream site, and these results are consistent with Hoare (1984) that found that concentrations of TN and NO3-N in urban streams often correlate mainly with wastewater inputs.

The effect on the NO3-N:PN ratio during the long residence times (in August and September 2016) was however only observed in the pond, not in the upstream samples. This indicates that longer pond residence times also can affect NO3-N concentrations. Mechanisms that may explain this might be that longer residence times combined with the observed high algal volumes results in NO3-N being assimilated faster than influx and nitrification make up for the difference. Longer residence times could further allow for more NO3-N to be lost though denitrification in the deeper anoxic stratum or in the sediment. Denitrification was not measured in this study, but in August 2016 the lowest 0.8 m or the water column was anoxic, which could have allowed for denitrification to be an important factor. Overall, the results indicate that stream discharge correlates with NO3-N and TN loading to the pond, and that longer residence times may further reduce the NO3-N:PN ratio.

4.1.3 Pond profiles.

The pond profiles show sharp inverse gradients for oxygen and conductivity during most summer months, indicating that there are periods with stratification despite of the pond's shallow depth. Oxygen levels also declined in the deepest stratum during the summer months.

The persisting low oxygen levels in the deeper strata, especially in 2017, indicate that stratification is stable through longer periods. According to Wetzel (2001) stable thermal stratification throughout the season is not expected in water bodies as shallow as Teglverksdammen. However, diel cycles of thermal and chemical stratification have been documented in fresh- and saltwater aquaculture ponds as shallow as 2 m, and is normally associated with warm weather and little wind (Losordo & Piedrahita, 1991). In Teglverksdammen, the temperature gradients alone did not indicate high stability of strata. The inverse sharp oxygen and conductivity gradients however still indicated that during most

summer months, the lowest stratum did not mix with the top stratum. During the months that indicated stratification, the temperature gradient varied from steep and well defined between two more temperature-homogenous strata, to only gradual or more complex. For none of the months the change was more than 4ºC. For some months, like July and August 2017, the slope in the temperature gradient was so small or gradual that it alone would indicate low stability of strata. Thermal stability may however not be the sole factor for strata not mixing. To begin with, the pond's situation below a hill and between relatively tall buildings likely contribute to effective sheltering from wind. It is further worth noticing the indications of chemical stability of strata. Several profiles show sharp conductivity and inverse oxygen gradients, independently of changes in the temperature gradient. This is most notable in May 2017, July 2017, October 2017 and July 2016. The largest recorded gradient in conductivity was in July 2016. The profile only has a few data points in the lower part of the pond, but the second profile recorded (Appendix J) also showed this gradient. The change in conductivity between 2.20 m and 2.30 meter in this sample is equivalent to a change in salinity from 0.21 to 0.59 PSU (direct output from the sonde data). That change in salinity at a constant temperature (4ºC) corresponds to a density change similar to when temperature changes from 4 to 10ºC at constant salinity (0.21 PSU) (calculation as in Maidment (1993)). It is therefore likely that salinity also contributed to stability of strata in Teglverksdammen. It is however not possible to establish what the causation of the increase in salinity is. Rise in conductivity in the lower stratum may follow periods with anoxic conditions as minerals can be reduced and mobilised from the substrate (mainly iron, manganese and calcium and magnesium carbonates) (Bowling & Tyler, 1990; Stahl & May, 1967; Tyler & Buckney, 1974). However, it may also be the case that influxes of high salt content water flowed to the bottom and contributed to a stable stratum. The findings of marine phytoplankton in the pond in spring 2017 support that the stream might indeed be salt polluted in periods. Road runoff during months when salt is used for de-icing is a probable source as Hovinbekken receives untreated road runoff upstream of the restored reach. A combination of effective sheltering, temperature- and salinity gradients likely contributed to little mixing of strata for longer periods.

Oxygen saturation declined towards the bottom during the months with indications of stratification. In 2016, oxygen levels sank gradually from June and reached a minimum below 10% in August. The following year, oxygen saturation in the lowest stratum stayed below 10% from May to August. The oxygen profile in May 2016 differed from the overall pattern with a sharp gradient for increasing oxygen saturation and conductivity at approximately 2 m,

and oversaturation of oxygen towards the bottom. This exception is likely a result of photosynthetic activity near the bottom (discussed further in chapter 4.1.6). Oxygen depletion in the lowest stratum of a lake is mainly linked to the decomposition of organic matter. It is therefore often considered as a secondary effect to the environmental pressure eutrophication as the increased productivity results in higher organic loading of the lowest stratum. But it can also be an effect of high organic loading in general (Sandlund et al., 2015; Wetzel, 2001). The TOC and phytoplankton concentrations were in general not very high, but there were events with higher phytoplankton volumes as well as events with extensive benthic growth. In addition, patches of *Oscillatoria* may have contributed to organic loading. The volume of the bottom stratum was also relatively small, possibly allowing for faster depletion of the oxygen reserve. Low oxygen levels near the bottom can have effects on both fish and benthic animals, and the oxygen levels were indeed so low that they would indicate bad ecological status if used as a supporting QE for the BQE "fish" in the WFD classification system. However, the word hypolimnion in the WFD is used to describe the lowest stratum of larger lakes. In Teglverksdammen, less than 1 m of the water column was affected, and it is not certain the ecological effect on fish is as relevant. Redox conditions are however further important for speciation of phosphates and anoxic conditions will in general result in dissolution of iron-phosphates from sedimented mineral material (VanLoon & Duffy, 2005). Whether anoxic conditions will result in net release of phosphorus however also depends on physiochemical factors like sulphate reduction rates, temperature and pH, and microbiological uptake and mineralisation further plays an important role (Hupfer & Lewandowski, 2008; Schindler, 1974; Wetzel, 2001). The fact that low oxygen levels in the lower stratum can have negative effects on ecology, and the further possible effect of phosphorus release, mean the development of such conditions should be considered undesirable. Efforts to reduce salt pollution and aeration of the pond may be measures to consider further.

4.1.4 Water Transparency and Colour.

The water transparency varied from very clear with a secchi depth equal to the depth of the pond (3 m), to turbid and a secchi depth of only 0.4 m. The water colour varied from clear to very grey.

The lowest secchi depths were associated with clay particles as indicated by the grey colour and high turbidity. Since the turbidity was low during the highest phytoplankton concentrations, and phytoplankton concentration further low on average, phytoplankton likely contributed little to the turbidity in the pond. There were two recorded events with high

turbidity grey water in 2016, and the water just entering the pond in October 2016 was heavily clay loaded by visual inspection. In 2017, the only month with grey and high turbidity water recorded was October. The event with heavily clay-loaded water just entering the beginning of the pond in October 2016, did not coincide with previous heavy rainfall. The clear separate colour between the water just entering the pond and the water a meter into the pond, indicate that the clay particles likely originated from a site specific-event rather than diffuse erosion. There was further no overall correlation between the residence time and the turbidity. Neither was there observed any pattern indicating that large peaks in discharge or rainfall in days previous to the sampling may have caused the events with grey water and low secchi. The exception being for May 2016 when a large rainfall had occurred the day before sampling. Overall, the results indicate that the more turbid water sometimes observed was due to higher concentrations of clay particles in the water some months, and that the loading of these particles likely originated from site specific events rather than general erosion. Such events may be building or drilling sites near the stream.

If the rule of thumb formula for the euphotic zone is used, multiplying Z_{SD} by a factor of 3, the whole or most of the top 2 m of water column existed in the euphotic zone during most of the months (see Table 3 in results). The lower secchi depths were mostly associated with grey water colour, indicating suspended clay particles. Inorganic particles can to a larger extent than organic material forward-scatter and refract light rather than absorb (Wetzel, 2001). The euphotic zone was therefore likely larger than the lowest secchi depths would suggest. This can explain how there despite low secchi depth appeared to be photosynthetic activity near the bottom in May 2016. Since the euphotic depth is defined as the depth only reached by 1% of the light, light may still have been limiting during some months and below certain depths for other taxonomic groups.

4.1.5 Phytoplankton total biovolume and assemblages.

Phytoplankton total biovolume and assemblages varied throughout the two growth seasons with few signs of seasonality (see Figure 12 in results). The trophic state of the pond as indicated by mean and maximum phytoplankton biovolume was mesotrophic both years if the 2016 September sample is not considered. Including this sample raised the trophic status of the pond to polyeutrophic in 2016. The strong correlation with the chlorophyll a readings indicate no problems with the accuracy of the total biovolume estimate.

Neither the total phytoplankton biovolume concentration nor succession in assemblages indicated seasonality. Both seasons did however show a peak in total biovolume

in June, but different phyla dominated. In large, deep, dimictic temperate lakes that are relatively undisturbed, seasonal patterns in phytoplankton assemblages can often be found (Salmaso, 2002; Wetzel, 2001). Physical and biotic factors that are important for the regulation of such seasonal patterns include changes in temperature, light availability, herbivore pressure, nutrient loading as well as spring and autumn mixing of the water column. Salmaso (2002) states that seasonal patterns in phytoplankton assemblages arise in deeper larger lakes due to these systems resilience against disturbances, and thus relatively predictable patterns in the mentioned physio-chemical conditions that affect the succession. Salmaso (2002) further states that this stands in contrast to smaller lakes where seasonal succession is often more unpredictable. The results from this study did indeed show very few trends that indicated total phytoplankton concentration or phyla succession correlated with specific environmental variables normally thought to be important for phytoplankton succession. Alternations in the recorded physio-chemical variables in the pond were also pronounced, although more stable the second year. The urban location and small dimensions of Teglverksdammen makes it especially susceptible to disturbances. Further, the fact that 2016 was the first growth season in Teglverksdammen likely influenced the alternations in biotic factors, such as zooplankton populations. Lack of seasonal periodicity in phytoplankton or correlation with growth related variables in such a system may therefore be considered normal.

The correlation tests and PCA indicated that phytoplankton biovolume concentration was, overall, to little extent dependent on the light availability. Both the PCA and the correlation tests indicated no correlation with water transparency in the pond (secchi depth or turbidity). The biplot did however indicate phytoplankton was positively associated with the cumulative incoming shortwave solar irradiation four days before sampling (SI4d), as well as with temperature. The biplot however further indicated the September 2016 sample, taken after the long stagnation period, might have been influential for this association. The SI4d and phytoplankton did also not show significant correlation. Growth may still have been limited by light during some conditions, but the results indicate the phytoplankton biovolume in the pond to a lesser extent depended on light availability.

Nutrient concentrations were generally high, and the variation in concentrations did not appear to be an important factor for the total phytoplankton biovolume in Teglverksdammen. If the phytoplankton community is nutrient limited, increase in the limiting nutrient will generally result in phytoplankton growth, although co-limitation also has been observed (Dzialowski, Wang, Lim, Spotts, & Huggins, 2005; Lewis William &

Wurtsbaugh Wayne, 2008; Schindler & Fee, 1974; Stahl & May, 1967). No significant correlations with nutrient concentrations and total phytoplankton biovolume was found in this study. However, NO3-N concentration sank during August and September 2016 when there was high algal volumes and the residence times were long. There was further a negative association between phytoplankton biomass and NO3-N concentrations in the pond in 2017, although the correlation was not significant after Bonferroni correction. This is not to say NO3-N limitation was observed, but rather that during some conditions, NO3-N appears to at least be assimilated faster than nitrification and influx are making up for the difference. Many methods for assessing whether a system is limited by a certain nutrient exist. The results from a bioassay study by Dzialowski et al. (2005) in 19 Kansas reservoirs indicated that in general, TN:TP ratios <18 (molar) indicated N-limitation, TN:TP ratios of 20-46 indicated co-limitation and, and TN:TP ratios > 65 generally indicated P-limitation. Different ratios have however also been suggested in other studies (Ptacnik, Andersen, & Tamminen, 2010). The mean annual concentrations of TN and TP in Teglverksdammen gives molar TN:TP ratios of 54 in 2016 and 75 in 2017 which would according to the mentioned study indicate that if there was nutrient limitation, the pond was somewhere between weakly N-P co-limited to P-limited. The sum of the following findings however indicate that nutrients were likely not the most important factor for controlling phytoplankton biovolume in the pond; (1) There was no overall correlation found between phytoplankton biovolume and nutrient concentrations in either the pond, nor in the upstream samples. It should however be considered that nutrient concentration in the upstream samples may not be representative of the recent input due to fluctuations, and the fact that birds may also be important for nutrient input. (2) The PCA biplots indicated no positive association between the phytoplankton concentration and either of the nutrients. (3) As indicated by the WFD status classes for the QE's TP and TN concentrations were generally high, and the PTI further indicate phytoplankton assemblages associated with high TP concentrations. Meanwhile the status class for the phytoplankton biovolume index indicated the phytoplankton biovolumes was on average low (see chapter 3.2.1). A correlation or lack thereof is not alone enough to establish whether nutrients are limiting for growth, but these results together is at least an indication that nutrient limitation was not an important factor for the resulting total phytoplankton concentration. It is still possible there was nutrient limitation at times, but that other factors such as loss of phytoplankton may be more important.

The indication that nutrient levels had little effect on phytoplankton is in accordance with Soballe and Kimmel (1987), that suggested that in water bodies with shorter residence

time, the response of phytoplankton to increased phosphorus concentration is small compared to waterbodies with longer residence times. As briefly discussed in 4.1.1. earlier, the results of this study indicated that the longer residence times may allow for higher phytoplankton concentrations. This effect was mainly seen after the longer stagnation period in September 2016. It should also be noted that there were also notably large algal volumes in August 2016 after a shorter (1 week) stagnation period, although the bulk of the algae was benthic and therefore not represented by the plankton sample. There were indications the water level had been lower previous to sampling this month (see chapter 3.1.9), and the filamentous algal masses had therefore likely occupied an even larger part of the water column than observed during sampling. It is therefore possible that the filamentous algal masses had inhabited the growth of phytoplankton this month, at least through shading. For phytoplankton, the most obvious effect of changes in residence times is the influence on the rate of phytoplankton loss through flushing (Elliott & Defew, 2012). This effect should logically be of relatively larger importance for small waterbodies with shorter residence times, as the ratio of loss through flushing to other losses (such as grazing, pathogens and sedimentation) should then be greater. This effect of residence time on phytoplankton loss, may explain why response to nutrients is generally low in water bodies with short residence time as in Soballe and Kimmel (1987). However, secondary effects of flow conditions and a low water level on turbidity, light availability and stratification could also be relevant. These variables did however show great variation independently of the residence time. Although it was beyond the scope of this study to examine losses of phytoplankton in the pond, it is reasonable to assume flushing rates were important for the resulting phytoplankton volumes in the pond. Eppley (1972) suggested that phytoplankton community maximum growth rates under perfect conditions (no light limitation 24h/day) could be estimated through the formula $\mu = 0.851(1.066)^T$ (doubling of mass/day where T= temperature in °C). For the warmest registered temperature in the pond (17°C, integrated from top 2m of profile) this would give a maximum possible doubling rate of 2.5/day. Even when assuming this theoretical maximum rate, it becomes obvious that if the whole pond volume is exchanged within a few days, this can drastically impact the amount of plankton in the pond.

Similarly as for total phytoplankton biovolume, the individual phyla groups showed no significant correlations with environmental variables after Bonferroni correction. The trend for a negative association between *Cryptophyta* and NO3-N that initially gave a low p-value, yielded a much higher p-value after the influential sample was removed. Cryptomonads are

however known to be able to utilize both NH4-N and organic sources of nitrogen, but not all species can utilize NO3-N (Graham, Graham, & Wilcox, 2009). It is therefore possible that the phylum had an advantage under low NO3-N conditions in the pond. This is however highly hypothetical, and the indication of a correlation might as well just be a coincidence. For *Bacillariophyta,* or diatoms, no statistically significant trends or trends with low p-value were observed. The phylum fraction was largest in spring 2016 and autumn 2017. No overall seasonal patterns for the phylum were observed in this study. However, Wetzel (2001) states that spring and autumn diatom maximums during lake mixing are common trends for phytoplankton community succession in lakes. Further, that in reservoirs it is also common with increases in diatom populations in summer with shorter residence times. It cannot be concluded that the spring maximum in 2016 followed pond mixing, but the residence times in the beginning of the year were short and the temperature gradient was small at least in May. No pronounced increase in diatoms was observed in autumn 2016, but the residence times were also longer than the global mean and the pond profiles did not indicate mixing until October. The 2017 maximum in the phylum was observed in September when the almost homogenous profiles indicated the water column had mixed. No spring maximum was observed in 2017 but the pond profiles also showed larger gradients indicating early establishment of strata this spring. It is therefore possible that mixing and short residence times had an effect on the amount of diatoms also in Teglverksdammen. There is however very little information to examine whether this was a real trend, and it cannot be rejected that this was coincidental. For the third largest phylum in the pond, *Ochrophyta,* no overall trends or seasonal patterns were observed. The different subordinate taxa within the phylum do also thrive under very different environmental conditions (Graham et al., 2009; Wetzel, 2001). This is also true if you move down in taxonomic rank to the two largest subordinate taxonomic groups observed, *Synurophyceae (Mallomonans sp*. dominated in September 2016*)* and *Crysophyceae* (dominated in August 2017) (Graham et al., 2009). Overall, the succession of the most dominant phyla indicated no seasonality and no single environmental factor could be identified as the most important for the abundance of either phyla. As briefly discussed earlier, the urban location, newly established system and short residence time are all factors that contribute towards low inertia against disturbances. This may explain why the phytoplankton assemblage development showed little order or predictable response to environmental variables, at least that could be detected on a monthly resolution.

The *Cyanobacteria* fraction was relatively small in the phytoplankton community in Teglverksdammen. It is however of particular interest from a water management standpoint as

several species can produce toxins harmful to humans (Graham et al., 2009). The relative fraction of the phylum in lakes commonly increase with increasing TP concentrations (Håkanson, Bryhn, & Hytteborn, 2007; S. Watson, B., McCauley, & Downing, 2003) and total phytoplankton biovolume (Brettum & Andersen, 2005). No such trends were observed in the pond. However, there were also substantial amounts of the cyanobacterium *O. sancta* throughout the system that was not represented within the pelagic samples (see chapter 3.1.9).

4.1.6 Algal growth not represented by Pelagic Samples.

Neither the extensive benthic growth of filamentous green algae nor the patches of the cyanobacterium *Oscillatoria sancta* were represented by the pelagic samples. Both the trophic state and the amount of cyanobacteria in the pond should therefore be considered higher than just the pelagic samples indicated in 2016. Last, the presence of marine phytoplankton in April 2017 is also worth noticing as this indicate a disturbance of the ecosystem.

Oscillatoria sancta was noted throughout the reach both seasons and the cyanobacteria were likely also growing at the bottom of the pond in May 2016. The oxygen profile from this month indicated photosynthetic activity at the bottom, and patches of bottom dwelling cyanobacteria were observed detaching and floating to the top in the littoral zone due to air bubble formation within the patches. *O. sancta* is also known to form thick microbial mats at substrates that later may detach and float (Komárek, 2005). There was low secchi depth this month, but as discussed in chapter 4.1.4 the euphotic zone might be deeper than the secchi depth indicate. Species of *Oscillatoria* are known to thrive at depths with lower light intensity but where there is often good access to nutrients (Wetzel, 2001). The species can be considered cosmopolitan, and as it has been found in environments like irrigation ditches (Komárek, 2005), rice paddies (Vijayan & Ray, 2015) and water channels with industrial waste effluent (Parikh, Shah, & Madamwar, 2006) it might also be considered a pioneer species. Samples of *O. sancta* from the pond were tested negative for microcystin at NIVA. The dark floating patches did however notably affect the aesthetic appeal of the pond when they were abundant. The estimate of the amount of the patches in the reach was rough. Nevertheless, there seemed to be less of *O. sancta* in the restored reach the second year. Although it cannot be concluded this was a true decline, it is possible that this development was an effect of secondary succession as the ecosystem in the restored reach established.

Other observed algal growth to notice in the pond was the thick mass of the filamentous green algae *Spirogyra majuscula* in August 2016. At the time of sampling, the mass covered approximately the lower half of the water column. The discharge had been low

a week before sampling (see chapter 3.1.2) and the clear divide above which no algae grew indicated the water table had likely been at least 0.5 m lower than normal previous to the fieldwork. The algae mass had therefore likely covered an even larger part of the water column the days previous to sampling. Stevenson, Bothwell, Lowe, and Thorp (1996) states that high biomass communities dominated by green algae such as the Spirogyra taxon is known to form under relatively nutrient rich, and low flow conditions. Graham et al. (2009) further states that such communities do not well tolerate the sheer stress of higher water velocities, and that species of Spirogyra generally requires relatively high temperatures (optimum around 25°C), and further good light conditions for growth as they self-shade. Low sheer stress related to water velocity was likely not a very important factor for the growth of S. majuscula in the pond during the stagnation period, as the masses were also able to form in the wetlands under medium flow conditions. However, secondary effects of low flow conditions on turbidity may have been of importance. The oxygen saturation did sink towards the bottom in the recorded profile, suggesting that even when the water was clear, there was little photosynthetic activity in the bottom layer of algae mass at the time of the fieldwork. It is therefore likely that the algal mass could only grow in the pond under the circumstances that the beginning of stagnation period offered; relatively warm temperature, low water table and good light conditions.

The last findings of algae growth not represented by the pelagic samples that is important to notice is the presence of marine phytoplankton in the pond in April 2017. The large amount of birds in the system is a probable pathway for the algae to reach the pond. The fact that several marine taxa were present, and that these were so numerous further indicate that there probably had been growth of these algae in the system. This indicate that at least in the early spring 2017, the system was salt polluted. Road-runoff is a probable source.

4.2 Ecological Classification

The overall ecological status for both years combined was moderate. As this was indicated by the BQE phytoplankton alone, the lower status (poor) of the supporting QE total phosphorus did not affect the overall status.

On an annual basis, the BQE phytoplankton indicated ecological status moderate for 2016 and good for 2017. The overall class for 2017 was however downgraded to moderate since the supporting QE total phosphorus indicated poor status. For 2016, removing the influential September sample improved the indicated status for the biomass index from moderate to high, but did not raise the overall indicated class for the QE. However, the nEQR improved from nearly indicating poor status to nearly indicating good status 2016 (see Table 6 and Table 7).

A few things should be taken into account when interpreting the indicated status classes. First, it is recommended to use data from at least three years to determine overall status due to natural variations between years. However, since in this case the overall status was the same for both years individually and combined, it is likely data from a third year would not change the overall status. It is also common practice to include chlorophyll a from laboratory analyses to get a more accurate mean nEQR for the biomass index. However, the strong correlation with the semi-quantitate chlorophyll a reading from the sonde indicate no obvious problems with the phytoplankton biovolume estimate. Furthermore, and most importantly, the class-intervals in the system are based on a lake type, while the Teglverksdammen is a pond that would normally be typified as part of the stream. The pelagic zone in the pond is not a lotic habitat, but neither is it truly lentic, and this will affect how well the phytoplankton indices reflect the trophic state. In general, a larger fraction of the water column will be in the euphotic zone in shallow waterbodies, and phytoplankton could therefore be less light limited in the average volume of water in a small pond than in a deeper lake. Therefore, using the BQE phytoplankton on smaller water bodies like Teglverksdammen could yield a worse ecological status than if a reference system for small lakes or ponds had existed. The problem could however also be reverse. As discussed in 4.1.5, shorter residence times can result in higher loss of phytoplankton through flushing. Further, as the May 2016 profile and August 2016 extensive benthic growth showed, the overall algal growth of the waterbody can be much higher than the phytoplankton QE indicates. Overall, the findings in this study with extensive benthic growth and increase in phytoplankton in the longest residence time sample, indicate that the use of the BQE phytoplankton likely underestimate

the environmental pressure “eutrophication” in the pond. The class intervals are therefore not accurate for the pond. The indices measured are however still relevant, and the WFD classification system is still a useful tool for monitoring the development. The PTI should reflect mean phosphorus concentrations, and in the range of phosphorus concentrations measured the index response should be fairly linear (Ptacnik, Solimini, & Brettum, 2009). Further, large phytoplankton and cyanobacteria concentrations are as undesirable in the pond as in a normal lake from an ecological viewpoint and can in addition reduce the aesthetic appeal of the pond.

For the reasons mentioned above, phytoplankton QE should not be relied upon as the only quality element for estimating the ecological status or the environmental pressure eutrophication. However, Myrstad (2017) determined the Periphyton Index of Trophic status (PIT) for the BQE benthic algae for six sites along the restored reach in 2016. For the September-samples (normally used in classification) in the lower part of the reach the individual samples all indicated moderate status, and one sample near the inlet indicated poor status (T5). The index reflects the environmental pressure eutrophication in streams and should like the PTI respond mainly to phosphorus concentrations. The fact that the results from Myrstad (2017) supported the ecological classification from the phytoplankton BQE, support that the classification may be relevant. Further investigations of the chemical status of the system should however also be considered in overall assessment of the reach, considering the urban location and input of road runoff.

4.3 Retention of Nutrients and Organic Matter

The results indicate that there is a large variation in whether the pond with adjacent wetlands acts like a sink or source for nutrients and TOC. There was however a general trend in reduction which was strongest for NO3-N.

It should be noted that the difference in nutrient and TOC concentrations between the upstream and downstream station is just an estimate for how the water quality change along the reach. Samples do not truly represent a change in water quality before and after the pond and wetland as it is not the same water tested twice. Fluctuations in concentrations in the stream input may have influenced the results. For example, as discussed earlier, concentrations of TN and NO3-N correlated with discharge and may be influenced by the amount of precipitation. Assuming that the water quality in the stream above the pond is relatively stable over a time duration similar to the mean residence time, paired t-test may

however still provide an estimate for how the water quality change along the reach. It should also be noted that this estimate represents the early ability of purification, and that the further development of the adjacent wetlands may improve purification.

Whether a reduction in nutrients and organic matter will be seen after a lake or pond depends on the dynamics of import to and loss of these substances from the waterbody. If the only import of nutrients and organic matter is through the stream input and at least some particles sediment in the pond, the concentrations should logically decrease, and longer residence time would allow for more retention. However, the dynamics of a natural system is more complex.

Nutrients and matter can be imported to the pond through both allochthonous and autochthonous input. Allochthonous input can be through the stream inflow or through fertilisation by birds. Migration of fish and insects are also theoretical pathways, but likely not of large relevance for this study. In the stream, nutrients travel either as dissolved nutrient salts, in organic particles, or adsorbed to inorganic particles, i.e. clay or silt particles. There can also be autochthonous input of organic matter and nitrogen through primary production and nitrogen fixation. The latter for which mainly cyanobacteria are responsible for in water. Further, nutrients and organic material previously exported to sediments can also be re-imported back into the water. This can happen through re-suspension, for example when the pond is in circulation, or due to bioturbation. Under anoxic conditions, phosphorus can also be released from the sediments (discussed in chapter 4.1.3). There is however yet little accumulated sediment on the rocky substrate from which to release nutrients. For nutrients, stream input and fertilisation by birds are likely the most relevant in the pond. Input from birds are further a source of organic matter, and so is the primary production in the pond. In Teglverksdammen, the large numbers of birds that was frequently observed likely contributed considerably to nutrient and organic loading in the pond. This input could also be of larger relevance with longer residence times.

The pond with the adjacent wetlands can act like a sink for nutrient in several ways. Nutrients can be removed from the water in a pond through sedimentation of organic particles, hereunder biota like phytoplankton, and through sedimentation of inorganic material with sorbed nutrients. The latter is often considered an important removal mechanism for phosphorus (Schindler, 1974; Wetzel, 2001). Nitrate can also be lost through denitrification in the deeper anoxic stratum. Nutrients can also be removed through assimilation by biota, like phytoplankton, and then lost through sedimentation. However, if little of the phytoplankton have time to sediment the larger phytoplankton populations should not affect the difference in

total nutrient concentration upstream and downstream, but TOC should increase. Organic material may be lost through decomposition in addition to sedimentation. In the feasibility study sedimentation of particles was estimated to be the most important factor for removal of nutrients, both in the pond and in the adjacent wetlands (Norconsult, 2013). A positive effect on water quality in form of reduced particle and nutrient (mainly phosphorus) concentration was expected to be seen with a residence time of at least 24h.

This study found no trends indicating that the change in concentrations between the upstream and downstream site was affected by either phytoplankton concentrations in the pond nor the retention time. Phytoplankton concentrations were on average moderate, and any effect on water quality may have been obscured by effects of other factors. The purification mechanism of sedimentation of particles should logically have been more relevant with longer residence times. However, since large numbers of birds were frequently observed, they likely contributed considerably to the pond's nutrient import. The fact that the pond is small means that there is little dilution of this input. Episodes with a large number of birds may therefore have substantially influenced the nutrient concentrations. The effect of birds on water quality could also have been larger with longer residence times, as this input will then be even less diluted. Overall, the fact that changes in residence time did not affect changes in concentrations of nutrients or TOC indicate that factors like nutrient circulation within the pond, direct fertilisation by birds and possibly nitrogen fixation was of larger importance than settling time for particles.

5 Conclusions

Phytoplankton biovolumes were generally moderate in the pond even if light availability and nutrient concentrations were generally high. Changes in nutrient concentrations did not appear to be an important factor for the resulting phytoplankton biovolumes in the pond, and the sum of the results indicate there was little overall nutrient limitation. Neither was there observed any significant response to changes in light conditions or temperature although these are important controlling factors for phytoplankton growth. As the residence times were generally short, variations in the rate of phytoplankton loss through flushing was likely of larger importance for the resulting phytoplankton concentrations than other important growth controlling variables; light conditions, nutrients or temperature. Longer residence time especially in the warmer periods may therefore result in un-desirable algal growth in the pond, as was observed in August and September 2016. It might therefore be a good idea try to limit the length of maintenance work that results in periods without water flow in the stream, or if possible, plan such work to colder periods when growth will naturally be slower.

The small pond volume, short residence time and situation of the pond makes it especially susceptible to disturbances. Large alternations in the physiochemical environment was also recorded. The pond residence time was drastically altered during maintenance periods when the water supply was cut off. There were also several events recorded with heavily clay loaded water entering the pond. Furthermore, some of the findings also indicate that the pond receives salt-polluted water in periods. The high conductivity in the lower stratum in the pond already in May both years, also without low oxygen concentration in 2016, is one such indication. The findings of marine phytoplankton in the pond in April 2017 further support this, as well as indicate that there might be a significant ecological impact of this salt pollution. Road runoff is a probable source of the salt.

The large alternations in the physiochemical environment and short residence time of the pond likely contributed towards there being few signs of seasonality in phytoplankton biomass or assemblages nor correlations with growth controlling variables. Diatoms did show a weak indication of increasing biovolumes during spring and autumn as is commonly observed in natural lakes, but as assemblage succession generally showed little order, it cannot be excluded that this was coincidental.

The pond showed signs of persistent stratification during the summer months and oxygen levels declined near the bottom during these periods. Although the volume affected by low oxygen values is small there may be negative effects on biota and the possible effect of phosphorus release from sediments. Aeration of the pond may therefore be a measure to consider for further management plans.

The phytoplankton BQE improved from indicating moderate status the first year to good status the second year, but as phosphorus concentrations were still high the overall status was moderate both years. The pond is smaller than the lake types the BQE phytoplankton was developed for, but the classification system is still useful as the indices measure relevant parameters, and it gives an indication of the ecological status of the system. The BQE does however not well reflect trophic state during episodes with extensive benthic growth in a system like Teglverksdammen. Means for phytoplankton biovolume, PTI, TP or TN were all lower in 2017 than in 2016, although none of the differences were significant on their own. As there were further episodes with extensive benthic growth in 2016, the pond could be considered to have been somewhat less trophic in 2017.

There was large variation in whether the pond acted like a sink or source for nutrients or organic matter, although there was a general trend for reduction. The paired t-test are only estimates of how the water quality change after the pond, and fluctuations in the input may have affected the quality of the estimates. Further development of the wetlands might improve nutrient retention by the reach in the future. The results from this early study however indicate no or little net purification effect by the pond and adjacent wetlands. The effect of nutrient and organic matter removal through particle sedimentation should logically be larger with longer residence times, but no effect of residence times on net nutrient retention was observed here. The large number of birds that visited the pond likely contributed substantially to nutrient loading, and the effect of visiting birds on nutrient concentrations could logically have been larger with longer residence times. As there further are indications that longer residence times may allow for more algae in the pond, reducing the inflow to improve nutrient retention through sedimentation would not be the best management option.

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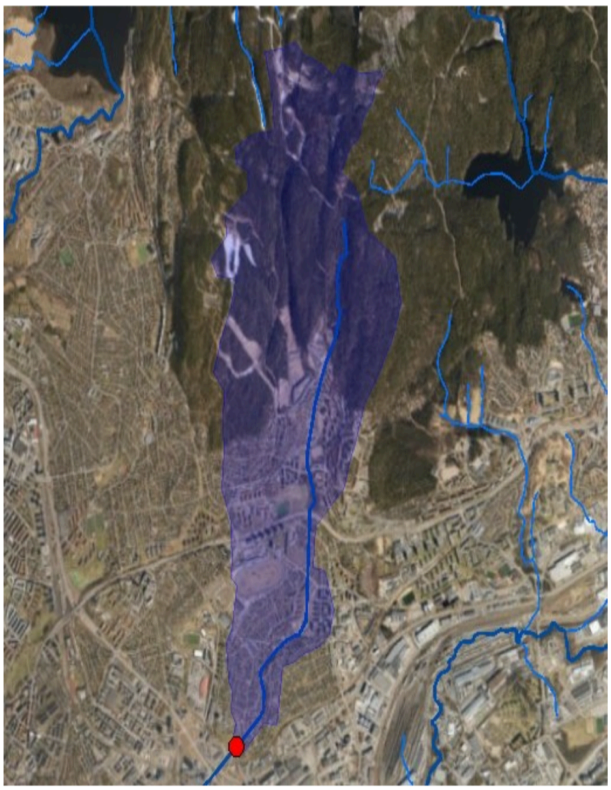
Appendices

APPENDIX A: CATCHMENT MAP AND REPORT



Lavvannskart

Vassdragsnr.:	006.A1Z
Kommune:	Oslo
Fylke:	Oslo
Vassdrag:	HOVINBEKKEN

Vannføringsindeks, se merknader		
Middelvannføring (61-90)	19,9	l/(s*km²)
Alminnelig lavvannføring		l/(s*km²)
5-persentil (hele året)		l/(s*km²)
5-persentil (1/5-30/9)		l/(s*km²)
5-persentil (1/10-30/4)		l/(s*km²)
Base flow	0,0	l/(s*km²)
BFI		

Klima		
Klimaregion	Ost	
Årsnedbør	828	mm
Sommernedbør	392	mm
Vinternedbør	436	mm
Årstemperatur	4,6	°C
Sommertemperatur	12,3	°C
Vintertemperatur	-1,0	°C
Temperatur Juli	15,0	°C
Temperatur August	13,9	°C

Feltparametere	
Areal (A)	4,8 km²
Effektiv sjø (S_{eff})	0,1 %
Elvelengde (E_L)	4,0 km
Elvegradient (E_G)	-23,8 m/km
Elvegradient$_{1085}$ (G_{1085})	43,0 m/km
Feltlengde(F_L)	4,9 km
H_{min}	101 moh.
H_{10}	132 moh.
H_{20}	160 moh.
H_{30}	189 moh.
H_{40}	216 moh.
H_{50}	250 moh.
H_{60}	281 moh.
H_{70}	300 moh.
H_{80}	318 moh.
H_{90}	337 moh.
H_{max}	377 moh.
Bre	0,0 %
Dyrket mark	1,6 %
Myr	0,3 %
Sjø	0,8 %
Skog	57,5 %
Snaufjell	0,0 %
Urban	20,1 %

1) Verdien er editert

Norges vassdrags- og energidirektorat

NVE

Kartbakgrunn: Statens Kartverk

Kartdatum: EUREF89 WGS84

Projeksjon: UTM 33N

Det er generelt stor usikkerhet i beregninger av lavvannsindekser. Resultatene bør verifiseres mot egne observasjoner eller sammenlignbare målestasjoner.

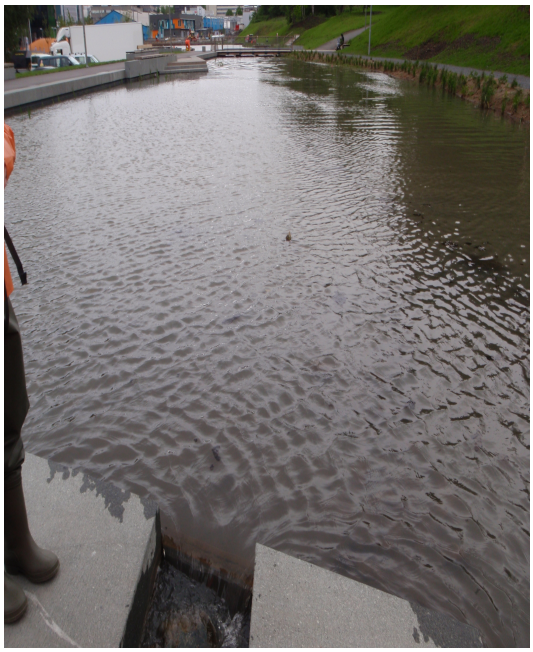
I nedbørfelt med høy breprosent eller stor innsjøprosent vil tørrværsavrenning (baseflow) ha store bidrag fra disse lagringsmagasinene.

Nedbørfeltgrenser, feltparametere og vannføringsindekser er automatisk generert og kan inneholde feil. Resultatene må kvalitetssikres.

Figure. Catchment map and report. Dyrket mark: farmed land, Myr: bog land, Skog: forest land, Urban: Urban land.

APPENDIX B: SAMPLING POINTS, PICTURES AND COORDINATES

T5



T7

T6

Figure. Sample sites upstream of Teglverksdammen (T5), downstream of Teglverksdammen(T7) and in Teglverksdammen (T6). Pictures from first year May to Jul.

Table. Teglverksdammen geographical coordinates	
T5	E: 265130.97 N: 6650485.18
T6	E: 265099.55 N: 6650330.40
T7	E: 264966.26 N: 6650195.46

APPENDIX C: STANDARDS AND UNCERTAINTIES

Table C. Standards and uncertainties

	unit	Standard code	Uncertainty
Chlorophyll a, sensor	µg/l	-	Estimate
Chlorophyll a, NIVA laboratory	µg/l	NS 4767:1983	20%
pH		NS-EN ISO 10523:2012	±0.2 pH units for entire temp range (-5 to 50 °C)
Phytoplankton Biovolume	mm^3/l	NS-EN 16695:2015	Estimate
Conductivity	$\mu S/cm$		±1% of reading or 2 µS/cm whichever is greater
Secchi	m	NS-EN 16698:2015	Estimate
Dissolved oxygen	mg/l, % saturation	ISO 17289	0-200%: ±1%
Total Organic Carbon	mg/l	NS-EN 1484	1-5 mg C/L: ±0,6 mg C/L 5-25 mg C/L: ±12 %
Total phosphorous	mg/l	NS-EN-ISO 15681-1 & 2, : 2015	0,005-0,010 mg/L: ± 0,002 mg/L 0,010-1,00 mg/L: ± 20 % 1,00-5,00 mg/L: ± 10 %
Phosphate	mg/l	NS-EN ISO 6878 -1 :2004	0,002-0,007 g/L: ± 0,0011 mg/L 0,007 -0,300mg/L: ± 16 %
Total Nitrogen	mg/	NS - EN ISO 11905-1 PART 1:1997	0,140-0,45 mg/L: ± 0,070 mg/l 0,45 -8,00 mg/L: ± 15 %
Ammonium	mg/	NS - 4746 1. Ed: 1975	0,005-0,030 mg/L: ± 0,004 mg/L 0,030-30 mg/L: ± 15 %
Nitrate	mg/	NS - EN ISO 13395 - 1996	0,008 – 0,025 mg/l: ± 0,003 mg/l 0,025 - 1,00 mg/l: ± 12,00 %

APPENDIX D: BOUNDARY VALUES FOR TROPHIC STATE INTERVALS, PHYTOPLANKTON

Table 1. The boundary values for trophic state as in Brettum (1989).

	Ultra-oligotrophic	Oligo-trophic	Meso-trophic	Oligomeso-trophic	Eutrophic	Poly-eutrophic	Hyper-eutrophic
Max biovolume	0-0.2	0.2-0.7	0.7-1.2	1.2-3	3-5	5-10	>10
Mean biovolume	0-0.12	0.12-0.4	0.4-0.6	0.6-1.5	1.5-2.5	2.5-5	>5

APPENDIX E: DISCHARGE AND RESIDENCE TIMES

Table 2. Discharge and residence times[a] for 2016.

	Residence time
May	1.83
June	3.62
July	1.85
August	10.87
September	34.07
October	7.20
Global mean residence time *	**3.70 days**

Note* Based on all discharge data.m

APPENDIX F: WATER CHEMISTRY

Table 1. Water chemistry in the pond (T6). Dissolved nitrogen; DN, Particulate Nitrogen; PN, particulate phosphorous; PP.

	NO3-N mg/l	P mg/l	N mg/l	PO4-P mg/l	NH4-N mg/l	C mg/l	Ca mg/l	pH	Turb FNU	Cond µS/cm	NO3/PN	NH4/PN	PO4/PP	DN/PN
M16	0,9530	0,0390	1,4700	0,0170	0,0920	5,0000	34,9000	7,8496	17,8989	251,1720	2,2424	0,2165	0,7727	2,45882353
Jun16	0,5710	0,0220	0,8880	0,0050	0,0170	2,7000	42,1000	8,3488	4,4515	216,3360	1,9033	0,0567	0,2941	1,96
Jul16	1,0300	0,0620	1,6200	0,0480	0,1530	4,6000	40,8000	7,0102	27,5350	420,6180	2,3570	0,3501	3,4286	2,70709382
A16	0,2980	0,0460	0,6600	0,0100	0,0350	4,7000	31,9000	8,7066	1,8287	316,9740	0,9113	0,1070	0,2778	1,01834862
S16	0,1200	0,0700	0,8140	0,0130	0,0490	4,8000	44,8000	8,0960	3,6491	405,9910	0,1860	0,0760	0,2281	0,2620155
O16	0,5830	0,0170	0,8000	0,0070	0,0320	2,5000	43,6000	7,7687	1,2428	404,9960	3,1514	0,1730	0,7000	3,32432432
M17	0,6430	0,0280	0,8940	0,0060	0,0100	4,4000	35,7000	8,4711	3,1273	351,6610	2,6680	0,0415	0,2727	2,70954357
Jun17	0,5250	0,0310	0,8720	0,0070	0,0190	3,6000	34,2000	7,7116	4,0075	325,8730	1,6006	0,0579	0,2917	1,65853659
Jul17	0,7340	0,0280	0,9800	0,0150	0,1190	3,4000	30,8000	7,7285	9,4089	295,1430	5,7795	0,9370	1,1538	6,71653543
A17	0,6590	0,0200	0,9000	0,0240	0,0250	4,0000	31,6000	8,3337	2,0265	289,8810	3,0509	0,1157	-6,0000	3,16666667
S17	0,7970	0,0170	1,0500	0,0110	0,0380	3,3000	34,9000	7,9239	2,6039	304,8210	3,7070	0,1767	1,8333	3,88372093
O17	0,7760	0,0470	1,2400	0,0400	0,1170	4,1000	28,8000	7,7974	24,4867	267,6940	2,2363	0,3372	5,7143	2,57348703

Note: The negative ratio of PO4-P to PP in Jul 2017 is likely related to the accuracy of the laboratory tests.

Table 2. Water chemistry upstream the pond (T5). Particulate Nitrogen; PN, particulate phosphorous; PP.

	NO3-N mg/l	P mg/l	N mg/l	PO4-P mg/l	NH4-N mg/l	C mg/l	NO3-N:PN	PO4-P:PP	NH4-N:PN
M16	1,2200	0,0390	1,5200	0,0220	0,0410	4,6000	4,7104	1,2941	0,1583
Jun16	0,8380	0,0310	1,1700	0,0080	0,0350	2,8000	2,8215	0,3478	0,1178
Jul16	1,0100	0,1050	1,9500	0,0700	0,3450	4,5000	1,6975	2,0000	0,5798
A16	0,4610	0,0280	0,7600	0,0170	0,0250	4,0000	1,6825	1,5455	0,0912
S16	0,8320	0,0510	1,2300	0,0110	0,0480	2,9000	2,3771	0,2750	0,1371
O16	0,8510	0,1770	1,2400	0,0280	0,0390	6,5000	2,4314	0,1879	0,1114
M17	0,9260	0,0860	1,6700	0,0310	0,1910	12,1000	1,6745	0,5636	0,3454
Jun17	0,6010	0,0140	0,7420	0,0080	0,0130	2,5000	4,6953	1,3333	0,1016
Jul17	0,7450	0,0160	0,9540	0,0090	0,1130	2,9000	7,7604	1,2857	1,1771
A17	0,0380	0,0490	0,4230	0,0240	0,1010	4,2000	0,1338	0,9600	0,3556
S17	0,8490	0,0170	1,0200	0,0140	0,0310	3,1000	6,0643	4,6667	0,2214
O17	0,8300	0,0370	1,1600	0,0340	0,0540	3,7000	3,0072	11,3333	0,1957

Table 3. Water chemistry downstream the pond (T7). Particulate Nitrogen; PN, particulate phosphorous; PP.

	NO3-N mg/l	P mg/l	N mg/l	PO4-P mg/l	NH4-N mg/l	C mg/l	NO3:PN	PO4-P:PP	NH4:PN
M16	0,907	0,052	1,510	0,026	0,172	5,000	2,104	1,000	0,399
Jun16	0,538	0,019	0,858	0,005	0,031	2,800	1,862	0,357	0,107
Jul16	0,988	0,027	1,770	0,022	0,093	4,500	1,434	4,400	0,135
A16	0,089	0,024	0,522	0,009	0,010	6,000	0,210	0,600	0,024
S16									
O16	0,539	0,019	0,797	0,005	0,028	2,300	2,343	0,357	0,122
M17	0,641	0,013	0,869	0,013	0,010	3,000	2,940	0,000	0,046
Jun17	0,549	0,018	0,864	0,006	0,018	3,300	1,848	0,500	0,061
Jul17	0,713	0,024	1,000	0,014	0,088	3,400	3,583	1,400	0,442
A17	0,278	0,035	0,704	0,013	0,023	5,100	0,690	0,591	0,057
S17	0,787	0,016	1,050	0,012	0,036	3,500	3,467	3,000	0,159
O17	0,806	0,035	1,220	0,033	0,142	4,100	2,963	16,500	0,522

Note: No samples from September 2016, as the downstream site was dry due to the maintenance period.

APPENDIX G: NUTRIENT CONCENTRATION UPSTREAM OF TEGLVERKSDAMMEN (T5)

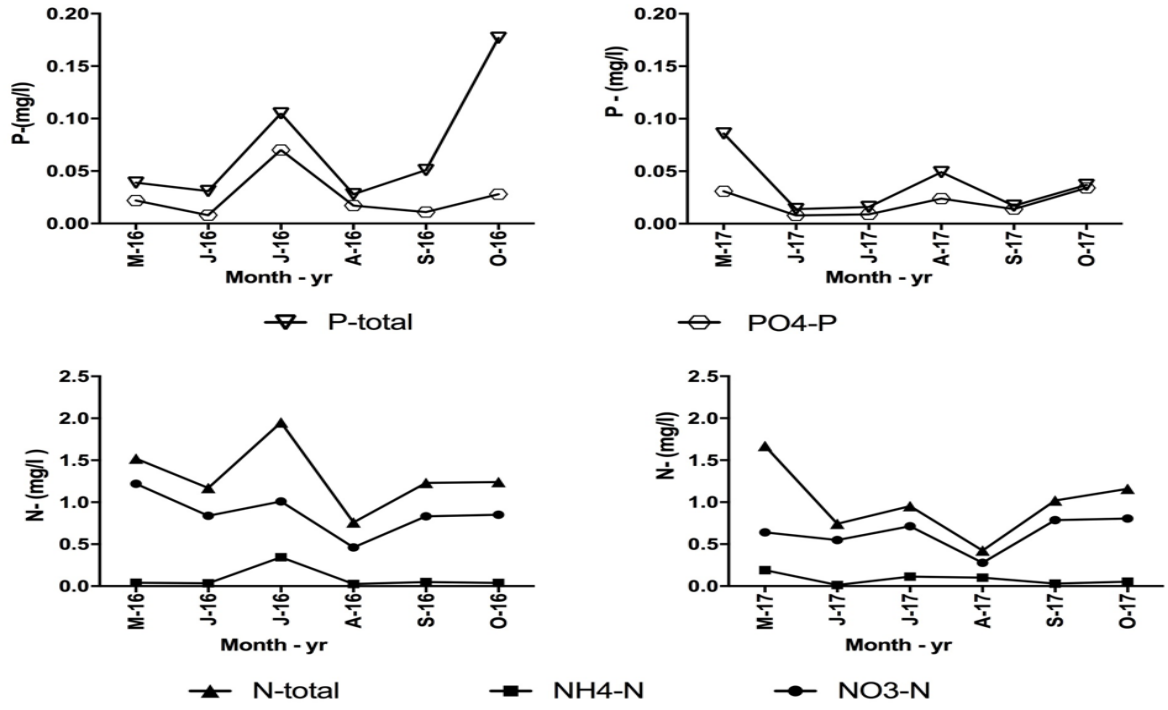


FIGURE. Nutrient concentrations upstream of Tegelverksdammen (T5). Nutrients were more stable the second year also upstream of the pond.

Table. Yearly means for water chemistry in the sample above Tegelverksdammen (T5), and Students t test for difference in mean nutrient concentration

	unit	Mean 2016	CI for mean 2016	Mean 2017	CI for mean 2017	p for difference 2017-2016
TN	mg/L	1.31	0.90, 1.73	0.99	0.56, 1.43	0.208
NH4N	mg/L	0.09	-0.04, 0.22	0.08	0.02, 0.15	0.932
NO3N	mg/L	0.87	0.61, 1.13	0.66	0.32, 1.00	0.255
TP	mg/L	0.07	0.01, 0.13	0.04	0.01, 0.07	0.224
P-PO4	mg/L	0.02	0.00, 0.04	0.02	0.01, 0.03	0.580

APPENDIX H: WATER QUALITY IN TEGLVERKSDAMMEN; CORRELATIONS WITH RESIDENCE RIME

```
Pearson correlations:
             RESIDENCETIME
C......       0.2765
Ca.....       0.3782
Cond          0.4130
N......      -0.5018
NH4.N        -0.3023
NO3.N..      -0.8194
P......       0.5653
PO4.P..      -0.2548
Turb         -0.4621
pH            0.2971

 Pairwise two-sided p-values:
             RESIDENCETIME
C......       0.5958
Ca.....       0.4597
Cond          0.4157
N......       0.3105
NH4.N         0.5603
NO3.N..       0.0460
P......       0.2424
PO4.P..       0.6261
Turb          0.3562
pH            0.5675
```

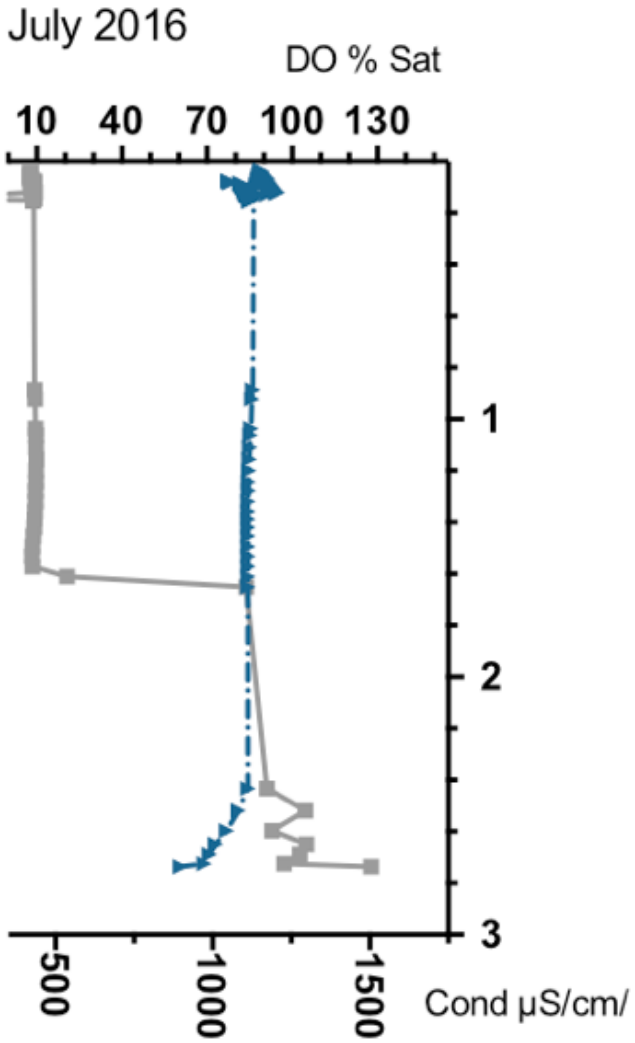
APPENDIX I: OXYGEN IN THE DEEPEST STRATUM

Table. Ecological status (WFD) indicated by oxygen levels in hypolimnion, based on tolerance limits for fish

	Month w. least O_2 in hypolimnion	Hypolimnion start, depth m	O_2 in 50th percentile mg/l	O_2 in 5th percentile mg/l	Ecological status indicated
2016	August	2.0	1.51	0.89	Poor
2017	June	2.6	0.42	0.17	Poor

Note: The word hypolimnion refers to the deepest strata of a stratified lake. It is not normally used when describing ponds, as they are not expected to stratify.

APPENDIX J: JULY POND PROFILE, SECOND READING



APPENDIX K: PHYTOPLANKTON CONCENTRATIONS AND TROPHIC STATE

Table. Phytoplankton Yearly Mean and Maximum Biovolume

	Mean Biovolume mm3/l	Max. Biovolume mm3/l	95% CI for mean	Indicated Trophic state level[b]
2016	1.631	6.872	1.631±2.777	polyeutrophic
2016a [a]	0.584	1.858	0.584±0.81679	mesotrophic
2017	0.729	1.703	0.729±0.619	mesotrophic

a) September sample taken out (table 1)
b) Trophic state intervals as defined by Brettum (1989)

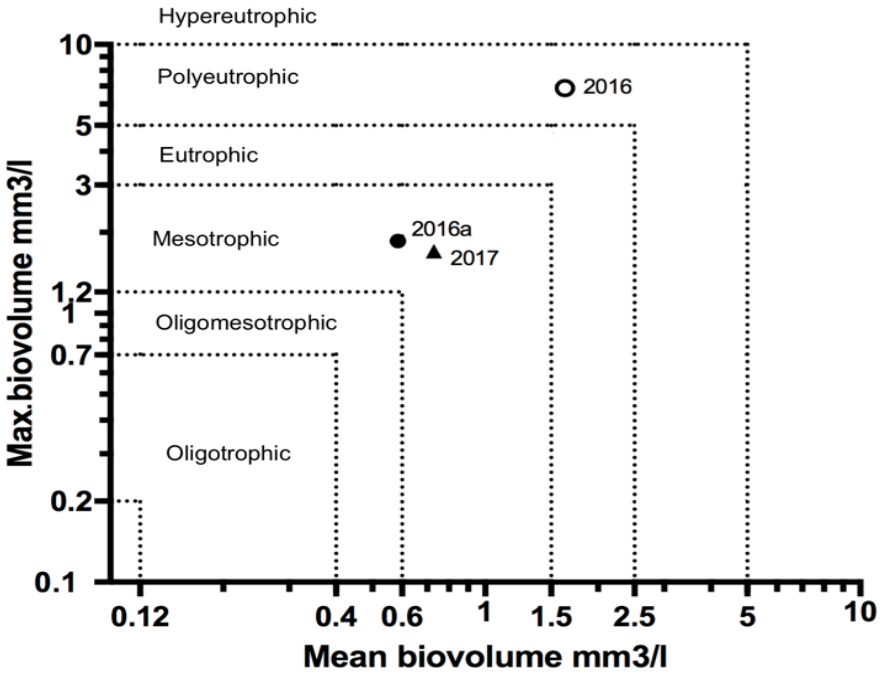


FIGURE. Trophic state as determined by yearly mean and maximum biovolume. Trophic state intervals as in Brettum (1989). The point for 2016a shows the mean and maximum values with the September sample taken out.

APPENDIX L: CORRELATIONS BETWEEN PHYTOPLANKTON CONCENTRATIONS AND DISSOLVED AND PARTICULATE-BOUND NUTRIENTS

Table. Pearsons correlation between phytoplankton biovolume and nutrient concentrations, as well as nutrient salt fractions to total N/P (DN- dissolved nitrogen, PN – particulate nitrogen=total-salts, PP particulate phosphorous)

```
Pearson correlations:
              DN:PN    NH4:PN   NH4:PN   NO3:PN   PO4:PP
BIOVOLUME    -0.5158  -0.2310  -0.2619  -0.5403  -0.1253

Pairwise  two-sided  p-values:
           DN.PN   NH4.N   NH4.PN  NO3.PN  PO4.PP
BIOVOLUME  0.0861  0.4701  0.4108  0.0698  0.6981
```

APPENDIX M: CORRELATIONS BETWEEN PHYTOPLANKTON CONCENTRATIONS AND UPSTREAM NUTRIENT CONCENTRATIONS

Table. Correlations between phytoplankton biovolume and upstream nutrient concentration

Pearson correlations:

	BIOVOLUME
BIOVOLUME	1.0000
T5...mg.l.C.....	-0.1856
T5...mg.l.NO3.N..	0.0372
T5...mg.l.P.....	-0.1180
T5..mg.l.N.....	0.0005
T5..mg.l.NH4.N	-0.1927
T5..mg.l.PO4.P..	-0.3677

Pairwise two-sided p-values:

	BIOVOLUME
BIOVOLUME	
T5...mg.l.C.....	0.5636
T5...mg.l.NO3.N..	0.9086
T5...mg.l.P.....	0.7150
T5..mg.l.N.....	0.9987
T5..mg.l.NH4.N	0.5484
T5..mg.l.PO4.P..	0.2397

APPENDIX N: PHYLA CORRELATIONS WITH ENVIRONMENTAL VARIABLES

Pearson correlations:

	Bacillariophyta	C.....	Ca.....	Charophyta	Chlorophyta
Bacillariophyta	1.0000	-0.3970	0.3696	-0.1975	0.1025
Charophyta	-0.1975	-0.6359	0.3456	1.0000	-0.1393
Chlorophyta	0.1025	0.2152	0.5279	-0.1393	1.0000
Cryptophyta	-0.0999	0.2819	0.4128	-0.0501	0.9299
Cyanobacteria	0.5346	-0.2245	-0.1288	-0.1338	-0.2189
Dinophyceae	-0.0953	0.3991	-0.2878	-0.2134	-0.2424
Euglenozoa	0.0081	0.3624	0.6336	-0.1838	0.8792
Ochrophyta	-0.0135	0.3204	0.4994	-0.1629	0.9618

	Cond	Cryptophyta	Cyanobacteria	Dinophyceae	Euglenozoa
Bacillariophyta	-0.5158	-0.0999	0.5346	-0.0953	0.0081
Charophyta	0.3588	-0.0501	-0.1338	-0.2134	-0.1838
Chlorophyta	0.3759	0.9299	-0.2189	-0.2424	0.8792
Cryptophyta	0.4331	1.0000	-0.3637	-0.2475	0.8174
Cyanobacteria	-0.5350	-0.3637	1.0000	-0.0188	-0.2927
Dinophyceae	-0.4428	-0.2475	-0.0188	1.0000	-0.1800
Euglenozoa	0.5251	0.8174	-0.2927	-0.1800	1.0000
Ochrophyta	0.4006	0.9006	-0.2512	-0.1432	0.9425

	N.....	NH4.N	NO3.N..	Ochrophyta	P.....	PO4.P..
Bacillariophyta	-0.1297	-0.3116	-0.1002	-0.0135	-0.2219	-0.3118
Charophyta	-0.2491	-0.2950	-0.0298	-0.1629	-0.4696	-0.3350
Chlorophyta	-0.3825	-0.2498	-0.7416	0.9618	0.4935	-0.2531
Cryptophyta	-0.3509	-0.2196	-0.7345	0.9006	0.5486	-0.2298
Cyanobacteria	0.0615	-0.0958	0.1501	-0.2512	-0.1991	0.0249
Dinophyceae	0.4483	0.1603	0.4073	-0.1432	-0.0516	0.1785
Euglenozoa	-0.0027	0.1249	-0.4734	0.9425	0.7325	0.1080
Ochrophyta	-0.2383	-0.0835	-0.6520	1.0000	0.5971	-0.0968

	Secchi.m	Temp	Turb	X1xretentiontime	pH
Bacillariophyta	-0.0969	0.2254	-0.1501	-0.1946	0.1995
Charophyta	0.4942	-0.6914	-0.3478	-0.1155	-0.0086
Chlorophyta	-0.0972	0.4307	-0.3159	0.9518	0.2433
Cryptophyta	-0.0795	0.4102	-0.2808	0.9854	0.2377
Cyanobacteria	-0.1662	-0.3483	0.1800	-0.3714	-0.2985
Dinophyceae	-0.1903	-0.1629	0.2809	-0.3207	-0.1598
Euglenozoa	-0.2851	0.4456	0.0394	0.8613	0.2249
Ochrophyta	-0.1362	0.3994	-0.1839	0.9574	0.1297

Pairwise two-sided p-values:

	Bacillariophyta	C.....	Ca.....	Charophyta	Chlorophyta
Bacillariophyta		0.2013	0.2370	0.5384	0.7513
Charophyta	0.5384	0.0263	0.2711		0.6659
Chlorophyta	0.7513	0.5018	0.0777	0.6659	
Cryptophyta	0.7573	0.3747	0.1823	0.8770	<.0001
Cyanobacteria	0.0733	0.4830	0.6899	0.6784	0.4943
Dinophyceae	0.7683	0.1988	0.3643	0.5055	0.4477
Euglenozoa	0.9801	0.2470	0.0270	0.5675	0.0002
Ochrophyta	0.9668	0.3100	0.0983	0.6130	<.0001

	Cond	Cryptophyta	Cyanobacteria	Dinophyceae	Euglenozoa
Bacillariophyta	0.0861	0.7573	0.0733	0.7683	0.9801
Charophyta	0.2520	0.8770	0.6784	0.5055	0.5675
Chlorophyta	0.2285	<.0001	0.4943	0.4477	0.0002
Cryptophyta	0.1596		0.2451	0.4379	0.0012
Cyanobacteria	0.0731	0.2451		0.9538	0.3559
Dinophyceae	0.1494	0.4379	0.9538		0.5757
Euglenozoa	0.0796	0.0012	0.3559	0.5757	
Ochrophyta	0.1969	<.0001	0.4310	0.6570	<.0001

	N.....	NH4.N	NO3.N..	Ochrophyta	P.....	PO4.P..
Bacillariophyta	0.6880	0.3242	0.7566	0.9668	0.4881	0.3238
Charophyta	0.4350	0.3519	0.9266	0.6130	0.1234	0.2872
Chlorophyta	0.2198	0.4336	0.0058	<.0001	0.1030	0.4274
Cryptophyta	0.2634	0.4929	0.0065	<.0001	0.0647	0.4724
Cyanobacteria	0.8494	0.7672	0.6414	0.4310	0.5350	0.9389
Dinophyceae	0.1438	0.6187	0.1888	0.6570	0.8735	0.5788
Euglenozoa	0.9933	0.6990	0.1201	<.0001	0.0067	0.7383
Ochrophyta	0.4557	0.7965	0.0216		0.0404	0.7647

	Secchi.m	Temp	Turb	X1xretentiontime	pH
Bacillariophyta	0.7645	0.4813	0.6416	0.7117	0.5342
Charophyta	0.1024	0.0128	0.2680	0.8275	0.9789
Chlorophyta	0.7638	0.1622	0.3172	0.0034	0.4461
Cryptophyta	0.8061	0.1854	0.3766	0.0003	0.4570
Cyanobacteria	0.6057	0.2673	0.5756	0.4686	0.3459
Dinophyceae	0.5537	0.6129	0.3764	0.5355	0.6199
Euglenozoa	0.3690	0.1466	0.9033	0.0275	0.4822
Ochrophyta	0.6729	0.1983	0.5671	0.0027	0.6879

APPENDIX O: CHANGE IN WATER QUALITY BETWEEN THE UPSTREAM AND DOWNSTREAM STATION, INDIVIDUAL DATA POINTS.

Table. Change in waterquality between upstream and downstream (concentrations T5-T7). Values for September 2016 are missing because of no streamflow at the point below the pond. Positive values indicate a reduction downstream. *The additional sample outside the growth season, November 2016, is not included in the tests.

Month/yr	NO3-N	TP	TN	PO4-P	NH4-N	TOC
5-16	0,313	-0,013	0,010	-0,004	-0,131	-0,400
6-16	0,300	0,012	0,312	0,003	0,004	0,000
7-16	0,022	0,078	0,180	0,048	0,252	0,000
8-16	0,372	0,004	0,238	0,008	0,015	-2,000
9-16						
10-16	0,312	0,158	0,443	0,023	0,011	4,200
11-16 *	-0,100	-0,023	-0,320	-0,019	-0,074	-0,600
5-17	0,285	0,073	0,801	0,018	0,181	9,100
6-17	0,052	-0,004	-0,122	0,002	-0,005	-0,800
7-17	0,032	-0,008	-0,046	-0,005	0,025	-0,500
8-17	-0,240	0,014	-0,281	0,011	0,078	-0,900
9-17	0,062	0,001	-0,030	0,002	-0,005	-0,400
10-17	0,024	0,002	-0,060	0,001	-0,088	-0,400

APPENDIX P: CORRELATIONS BETWEEN NUTRIENT AND TOC RETENTION WITH PHYTOPLANKTON BIOVOLUME AND RESIDENCE TIME.

Correlations, phytoplankton concentration (BIOVOLUME)and residence time with nutrient and TOC difference between upstream (T5)- downstream (T7)concentrations.

Pearson correlations:

	BIOVOLUME
BIOVOLUME	1.0000
TOC.....	0.1442
TN.....	0.2009
NH4.N	0.0185
NO3.N..	0.1833
TP.....	-0.1967
PO4.P..	-0.2776

	Residence time
BIOVOLUME	0.9206
TOC.....	-0.0544
TN.....	0.4932
NH4.N	-0.1220
NO3.N..	0.5941
TP.....	0.1086
PO4.P..	-0.1545
X1xretentiontime	1.0000

Pairwise two-sided p-values:

	BIOVOLUME
BIOVOLUME	
TOC.....	0.6723
TN.....	0.5536
NH4.N	0.9569
NO3.N..	0.5895
TP.....	0.5621
PO4.P..	0.4085

	Residence time
TOC.....	0.9308
TN.....	0.3985
NH4.N	0.8451
NO3.N..	0.2907
TP.....	0.8620
PO4.P..	0.8040

APPENDIX Q: PHYTOPLANKTON ASSEMBLAGE AND PHYLA BIOVOLUMES

Table. Phytoplankton taxa biovolumes in **mm³/l,** fraction of total biovolume in monthly sample, fraction within phylum and PTI-value of taxon. Number of decimals do not reflect the accuracy of the estimates. UK: Unknown. The notation "type..." does not refer to a taxonomically correct name, but rather note a taxon not identified to species level, different from other unidentified taxa in the sample but most likely the same as taxon in other samples marked with the same "type".

May 2016	Bacillariophyta	Cymbella	Diatoma vulgaris	Fragilaria sp.(type capucina)	Fragilaria nanana	Meridion circulare	Navicula sp.	Nitzschia sp.	Surirella sp.	Unknown
Total	**0,150545123**	**0,01840095**	**0,002385647**	**0,001955136**	**0,027375057**	**0,016134729**	**0,025963858**	**0,016043152**	**0,02658592**	**0,01570068**
Fraction of total biovolume	0,547528189	0,066923715	0,008676528	0,007110773	0,099562278	0,058681536	0,094429787	0,058348471	**0,0966922**	0,0571029
Fraction within phylum		0,122228803	0,015846723	0,012987045	0,181839548	0,10717537	0,172465617	0,106567063	0,17659767	0,10429216
PTI	-	-	-	2,54	2,981	-	2,862	3,917	-	-

May 2016	Charophyta	Closterium sp.
Total	**0,000554916**	**0,000554916**
Fraction of total biovolume	0,002018213	0,002018213
Fraction within phylum		1
PTI		2,81

May 2016	Chlorophyta	Carteria sp.	Chlamydomonas sp.	Monoraphidium sp.	Monoraphidium minutum
Total	**0,008524211**	**0,002222501**	**0,006138062**	**0,000123055**	**4,05925E-05**
Fraction of total biovolume	0,031002305	0,008083172	0,022323953	0,000447547	0,000147634
Fraction within phylum		0,260728092	0,720073957	0,014435924	0,004762026
PTI		2,19	2,36	-	3,643

May 2016	Cryptophyta	Cryptomonas sp.
Total	**0,004568623**	**0,004568623**
Fraction of total biovolume	0,016615948	0,016615948
Fraction within phylum		1
PTI	-	2,364

May 2016	Cyanobacteria	Oscillatoria sancta	Rivularia sp..	UK (likely sp.irulinaceae)	Pseudoanabema sp..	Unknown	UK type Rodshaped
Total	**0,015110428**	**0,005315575**	**0,000910236**	**0,001404826**	**0,003370493**	**0,001416892**	**0,002692407**
Fraction of total biovolume	0,054956183	0,019332589	0,0033105	0,00510931	0,012258383	0,005153195	0,009792205
Fraction within phylum		0,351781881	0,060238905	0,092970622	0,22305739	0,093769152	0,178182051
PTI	-	-	-	-	-	-	-

May 2016	Dinophyceae (Miozoa)	UK
Total	**0,019409989**	**0,019409989**
Fraction of total biovolume	0,070593561	0,070593561
Fraction within phylum		1
PTI	-	-

May 2016	Euglenozoa	Euglena sp.
Total	**0,006318509**	**0,006318509**
Fraction of total biovolume	0,02298023	0,02298023
Fraction within phylum		1
PTI	-	3,799

May 2016	Ochrophyta	UK
Total	**0,003278211**	**0,003278211**
Fraction of total biovolume	0,011922758	0,011922758
Fraction within phylum		1
PTI	-	

May 2016	Unknown	UK different	Picoplancton
Total	**0,066644095**	**0,056759044**	**0,009885051**
Fraction of total biovolume	0,242382614	0,206430975	0,035951639
Fraction within phylum		0,851674019	0,148325981
PTI	-	-	1,911

June 2016	Bacillariophyta	Cymbella sp.	Diatoma vulgaris	Fragilaria sp. (type capuci	Fragilaria nanana	Melosira varians	Navicula sp.	Melosira sp.	Ulnaria ulna	Unknown
Total	**1,299514345**	**0,003035558**	**0,032508415**	**0,001825934**	**1,057281351**	**0,00254469**	**0,00126482**	**0,00147566**	**0,00070245**	**0,19887548**
Fraction of total biovolume	0,699248963	0,001633388	0,017492285	0,000982507	0,568907062	0,001369259	0,00068058	0,00079403	**0,00037798**	0,10701188
Fraction within phylum		0,002335917	0,025015819	0,001405089	0,813597291	0,001958185	0,0009733	0,00113554	0,00054055	0,15303831
PTI	-	-	-	2,54	2,981	3,272	2,862	-	2,769	-

June 2016	Chlorophyta	Chlamydomonas :	Microspora	Tetradesmus dimorphus	UK
Total	**0,076203299**	**0,02033405**	**0,000100138**	**0,005524319**	**0,050244792**
Fraction of total biovolume	0,04100384	0,010941444	5,38829E-05	0,002972552	0,02703596
Fraction within phylum		0,266839502	0,001314094	0,072494484	0,65935192
PTI		2,36			

June 2016	Cryptophyta	Cryptomonas sp.	Croomonas/Komma
Total	**0,005205154**	**0,002990774**	**0,002214379**
Fraction of total biovolume	0,002800814	0,00160929	0,001191524
Fraction within phylum		0,574579429	0,425420571
PTI	-	2,364	2,106

June 2016	Cyanobacteria	Clorococcum	Mersismopedia glauca	Pseudoanabema sp.	Unknown type "Rodshaped"
Total	**0,046916414**	**0,044194551**	**0,000305815**	**0,000943996**	**0,001472052**
Fraction of total biovolume	0,025245011	0,023780418	0,000164555	0,00050795	0,000792089
Fraction within phylum		0,941984855	0,006518299	0,020120796	0,03137605
PTI	-	-	-	-	-

June 2016	Euglenozoa	Euglena	Phacus
Total	**0,011785214**	**0,006187237**	**0,005597976**
Fraction of total biovolume	0,006341445	0,003329258	0,003012186
Fraction within phylum		0,525	0,475
PTI	-	3,799	3,689

June 2016	Ochrophyta	Chrysophyceae	Mallomonas sp.	Ochromonas	Pedinellaceae
Total	**0,082747686**	**0,007812611**	**0,00741352**	**0,001604099**	**0,065917457**
Fraction of total biovolume	0,044525275	0,004203847	0,003989103	0,000863141	0,035469184
Fraction within phylum		0,094414852	0,08959187	0,019385418	0,79660786
PTI	-	1,924	2,125	1,893	1,869

June 2016	Unknown	UK different	Picoplancton
Total	**0,336070896**	**0,335**	**0,001070896**
Fraction of total biovolume	0,180834653	**0,18025842**	0,000576233
Fraction within phylum		0,996813481	0,003186519
PTI	-	-	1,911

July 2016	Bacillariophyta	Diatoma vulgaris	Fragilaria sp.(type capucina)	Melosira varians	Navicula sp.	Nitzchia sp.	UK
Total	**0,024687053**	**0,010693375**	**4,22168E-05**	**0,001073787**	**0,01200888**	**0,0008146**	**5,4E-05**
Fraction of total biovolume	0,22585262	0,097829691	0,000386226	0,009823672	0,10986471	0,00745245	0,0005
Fraction within phylum		0,433157212	0,00171008	0,043495938	0,48644426	0,03299696	0,0022
PTI	-	-	2,54	-	2,862	3,917	-

July 2016	Chlorophyta	Chlamydomonas sp.	Clorococcaceae	Keratococcus sp.	Schroederia	UK
Total	**0,011025052**	**0,000693618**	**0,001345182**	**0,000745274**	**0,00068585**	**0,00755513**
Fraction of total biovolume	0,100864076	0,006345651	0,012306567	0,006818237	0,0062746	0,06911903
Fraction within phylum		0,062912892	0,122011394	0,067598269	0,06220843	0,68526901
PTI	-	2,36	-	-	-	-

July 2016	Cryptophyta	Cryptomonas sp.	Croomonas/Komma sp.
Total	**0,025436188**	**0,021138086**	**0,004298102**
Fraction of total biovolume	0,232706176	0,193384445	0,039321731
Fraction within phylum		0,831024121	0,168975879
PTI	-	2,364	2,106

July 2016	Cyanobacteria	Pseudoanabema sp.	Unknown type "Rodshaped"	UK (likely sp.irulinaceae)
Total	**0,002058234**	**0,000235761**	**0,001748391**	**7,40813E-05**
Fraction of total biovolume	0,018830009	0,002156892	**0,015995374**	0,000677743
Fraction within phylum		0,114545481	0,849461835	0,035992683
PTI	-	0	0	0

July 2016	Euglenozoa	Euglena sp.
Total	**0,042204949**	**0,042204949**
Fraction of total biovolume	0,386117296	0,386117296
Fraction within phylum		1
PTI	-	3,799

July 2016	Ochrophyta	Mallomonas sp.
Total	**0,00388741**	**0,00388741**
Fraction of total biovolume	0,035564462	0,035564462
Fraction within phylum		1
PTI	-	2,125

July 2016	Unknown	Picoplankton
Total	**7,14438E-06**	**7,14438E-06**
Fraction of total biovolume	6,53613E-05	6,53613E-05
Fraction within phylum		1
PTI	-	1,911

August 2016	Bacillariophyta	Eunotia sp.	Fragilaria capucina	Navicula sp.	Nitzschia sp.	Rhoicosp.henia
Total	**0,010291504**	**0,000827363**	**0,00599335**	**0,000919191**	**0,000770486**	**0,001781115**
Fraction of total biovolume	0,024273833	0,001951442	0,014136085	0,002168029	0,00181729	0,004200987
Fraction within phylum		0,080392821	0,582358983	0,089315478	0,074866223	0,173066495
PTI	-	-	2,54	2,862	3,917	-

August 2016	Chlorophyta	Ankyra judayi	Carteria sp.	Chlamydomonas sp.	Gyromitus cordiformis	Monomastix sp.	Oocystis	UK
Total	**0,027960526**	**0,018688049**	**0,003514285**	**0,000677221**	**0,00160653**	**0,001147522**	**0,00159378**	**0,000733**
Fraction of total biovolume	0,065948488	0,044078163	0,125687363	0,001597313	0,003789208	**0,002706577**	**0,00375913**	0,001729
Fraction within phylum		0,668372611	0,125687363	0,02422061	0,05745708	0,041040771	0,05700107	0,02622
PTI		3,417	2,19	2,36	2,058	0	0	

August 2016	Cryptophyta	Cryptomon sp.	Chroomonas/rhodomonas sp..
Total	**0,36480548**	**0,253280843**	**0,111524636**
Fraction of total biovolume	0,860440538	0,597395372	0,263045166
Fraction within phylum		0,694290129	0,305709871
PTI	-	2,364	2,106

August 2016	Cyanobacteria	Crococculales	UK
Total	**0,008972346**	**0,007627265**	**0,001345081**
Fraction of total biovolume	0,021162429	0,017989884	0,003172546
Fraction within phylum		0,850085938	0,149914062
PTI	-	-	-

August 2016	Ochrophyta	Chrysamoeba
Total	**0,00051001**	**0,00051001**
Fraction of total biovolume	0,001202923	0,001202923
Fraction within phylum		1
PTI	-	-

August 2016	UK	UK different	Picoplancton
Total	**0,011435371**	**0,011307869**	**0,000127502**
Fraction of total biovolume	0,02697179	0,026671059	0,000300731
Fraction within phylum		0,988850174	0,011149826
PTI	-	-	1,911

September 2016	Bacillariophyta	Fragilaria sp..(type ca	Meridion circulare	Navicula sp.	Nitzschia sp.	Ulnaria ulna	UK (Mabye synedra)
Total	**0,147294671**	**0,00637867**	**0,02671345**	**0,000451469**	**0,009261852**	**0,09811333**	**0,0063759**
Fraction of total biovolume	0,021446294	0,000928743	0,003889513	6,57345E-05	0,001348538	0,01428543	0,000928339
Fraction within phylum		0,043305507	0,181360601	0,003065073	0,062879752	0,66610237	0,043286696
PTI	-	2,54	-	2,862	3,917	2,769	-

September 2016	Charophyta	Closterium
Total	**4,58665E-06**	**4,58665E-06**
Fraction of total biovolume	6,67822E-07	6,67822E-07
Fraction within phylum		1
PTI		2,81

September 2016	Chlorophyta	Ankyra Judai	Carteria sp.	Characium	Chlamydomonas sp.	Chlorella	Dictyosp.haerium	Microspora sp.	Monoraphidium	Oocystis marsonii	Scenedesmus	Radiococcus polycoccus	Unknown
Total	**0,357848993**	**0,193425168**	**0,011620077**	**0,007172887**	**0,041036825**	**0,05101699**	**0,003930198**	**0,006546236**	**0,000503872**	**0,00573831**	**0,017350589**	**0,002225043**	**0,01728279**
Fraction of total biovolume	0,052103276	0,540521761	0,032472013	0,020044452	0,005975015	0,00742814	0,000572242	0,00095314	7,33643E-05	0,000835505	**0,002526268**	**0,000323969**	0,0025164
Fraction within phylum		0,540521761	0,032472013	0,020044452	0,114676374	0,14256571	0,01098284	0,01829329	0,001408056	0,016035562	0,048485785	0,006217826	0,04829633
PTI	-	3,417	2,19	-	2,36	-	-	-		2,731	3,119	-	-

September 2016	Cryptophyta	Cryptomonas sp..	Chroomonas/rhodomonas sp..
Total	**2,724262703**	**2,689667282**	**0,03459542**
Fraction of total biovolume	0,39665617	0,391619032	0,005037138
Fraction within phylum		0,987300997	0,012699003
PTI	-	2,364	2,106

September 2016	Cyanobacteria	UK (likely spirulinacea	Unknown type "Rodshaped"
Total	**0,003219758**	**0,001551751**	**0,001668007**
Fraction of total biovolume	0,000468801	0,000225937	0,000242864
Fraction within phylum		0,481946318	0,518053682
PTI	-	-	-

September 2016	Euglenozoa	Euglena sp.	Euglena oxyuris f. minor
Total	**0,12852923**	**0,127517992**	**0,001011238**
Fraction of total biovolume	0,018714022	0,018566784	0,000147238
Fraction within phylum		0,992132236	0,007867764
PTI	-	3,799	3,964

September 2016	Ochrophyta	Mallomonas caudta	Mallomonas sp.	Pedinellaceae
Total	**3,441605504**	**2,032822199**	**1,370719185**	**0,038064121**
Fraction of total biovolume	0,501102209	0,295981539	**0,199578485**	0,005542185
Fraction within phylum		0,590661014	0,398278996	0,011059989
PTI	-	2,349	2,125	1,869

September 2016	Unknown	UK different	Picoplancton
Total	**0,06530546**	**0,06139833**	**0,003907131**
Fraction of total biovolume	0,009508559	0,008939676	0,000568883
Fraction within phylum		0,940171457	0,059828543
PTI	-	-	1,911

October 2016	Bacillariophyta	Achnantes sp..	Navicula sp.	Nitzschia acicularis	Nitzschia sp.. (type palea)	Nitzschia sp.	Ulnaria ulna
Total	**0,020618623**	**0,001100842**	**0,008092282**	**0,005320876**	**0,000505381**	**0,001558159**	**0,004041082**
Fraction of total biovolume	0,081531136	0,004353001	0,031998885	0,021040061	0,001998403	0,006161346	0,015979439
Fraction within phylum		0,053390656	0,39247442	0,258061669	0,02451092	0,075570471	0,195991865
PTI	-	1,84	2,862	3,917	3,917	3,917	2,769

October 2016	Charophyta	Closterium moniliferum
Total	**0,022742658**	**0,022742658**
Fraction of total biovolume	0,089930097	0,089930097
Fraction within phylum		1
PTI	-	2,81

October 2016	Chlorophyta	Ankistrodesmus fı	Carteria sp.	Chlamydomonas sp..	Microspora sp.	Schroederia setigera	Scourfieldia sp..	Paramastix conifera
Total	**0,040155627**	**0,008999511**	**0,014605622**	**0,011040096**	**0,000305363**	**0,004976775**	**0,000152234**	**7,60265E-05**
Fraction of total biovolume	0,158785285	0,224115811	0,36372541	0,043655272	0,00120748	0,019679399	0,00060197	0,000300628
Fraction within phylum		0,224115811	0,36372541	0,274932732	0,007604483	0,123937173	0,003791093	0,001893297
PTI	-	-	2,19	2,36	-	2,964	-	1,927

October 2016	Cryptophyta	Cryptomonas sp..	Chroomonas/rhodomonas sp..
Total	**0,124007098**	**0,113846572**	**0,010160527**
Fraction of total biovolume	0,490354742	0,450177507	0,040177235
Fraction within phylum		0,91806496	0,08193504
PTI	-	2,364	2,106

October 2016	Cyanobacteria	Ocillatoria sancta	Unknown - Pseu	Unknown type "Rodshaped"
Total	**0,011715317**	**0,010868654**	**0,000126861**	**0,000719802**
Fraction of total biovolume	0,046325261	0,042977346	**0,000501642**	0,002846273
Fraction within phylum		0,927730253	0,010828682	0,061441065
PTI	-	-	-	-

October 2016	Euglenozoa	Euglena
Total	**0,005980202**	**0,005980202**
Fraction of total biovolume	0,023647196	0,023647196
Fraction within phylum		1
PTI		3,799

October 2016	Ochrophyta	Crysophyceae	Mallomonas sp..	Mallomonas caudata
Total	**0,010107826**	**0,002661088**	**0,006517071**	**0,000929667**
Fraction of total biovolume	0,039968846	0,010522601	**0,025770111**	0,003676134
Fraction within phylum		0,26327008	0,644754942	0,091974978
PTI	-	1,924	2,125	2,349

October 2016	UK	UK different	Picoplancton
Total	**0,017565274**	**0,012758161**	**0,004807113**
Fraction of total biovolume	0,069457437	0,050448923	0,019008514
Fraction within phylum		0,726328596	0,273671404
PTI	-	-	1,911

May 2017	Bacillariophyta	Cymbella sp.	Diatoma vulgaris	Fragilaria sp..	Fragilaria sp..(type capucina)	Navicula sp.	Nitzschia acicularis	Nitzschia sp.	Rhoicosp.henia abbreviata	Ulnaria ulna	Unknown
Total	**0,179691273**	**0,024231127**	**0,038430103**	**0,00113541**	**1,31894E-05**	**0,01925612**	**0,001426809**	**0,00693754**	**0,003597624**	**0,08223993**	**0,00242342**
Fraction of total biovolume	0,159768969	0,021544631	0,034169372	0,001009527	1,17271E-05	0,0171212	0,001268619	0,006168377	**0,003198756**	**0,07312202**	0,00215473
Fraction within phylum		0,134848657	0,213867389	0,00631867	7,34004E-05	0,10716226	0,007940333	0,038608105	0,020021137	0,4576735	0,01348655
PTI	-	-	-	2,54	2,54	2,862	3,917	3,917	-	2,769	-

May 2017	Chlorophyta	Chlamydomonas sp.	Monoraphidium minutum	Radiococcus polycoccus	Schroederia setigera	Unknown	Scourfieldia sp.	Ulothrix
Total	**0,075559012**	**0,020044247**	**0,000194844**	**0,001940438**	**0,002029626**	**0,01775923**	**0,026671822**	**0,006918808**
Fraction of total biovolume	0,067181813	0,017821949	0,000173242	0,001725302	0,001804602	0,01579027	0,023714727	0,006151722
Fraction within phylum		0,265279374	0,002578701	0,025681093	0,026861467	0,23503784	0,352993257	0,09156827
PTI	-	2,36	3,643	-	2,964	-	-	-

May 2017	Cryptophyta	Cryptomonas sp.	Croomonas/Komma
Total	**0,16404535**	**0,116541121**	**0,047504229**
Fraction of total biovolume	0,145857704	0,103620251	0,042237453
Fraction within phylum		0,710420142	0,289579858
PTI	-	2,364	2,106

May 2017	Cyanobacteria	UK (type Komvophoron)	Romeria sp.	UK (likely sp.irulinaceae)	Rhabdoderma linearis
Total	**0,033525481**	**0,000443848**	**0,028382723**	**0,000426221**	**0,004272688**
Fraction of total biovolume	0,029808523	0,000394639	0,025235941	**0,000378966**	0,003798977
Fraction within phylum		0,013239131	0,846601521	0,012713358	0,12744599
PTI	-	-	-	-	-

May 2017	Ochrophyta	Chrysophyceae	Mallomonas sp.	Tribonema	Pedinellaceae
Total	**0,040747599**	**0,003298478**	**0,01791906**	**0,003216097**	**0,016313964**
Fraction of total biovolume	0,036229928	0,002932777	0,015932381	0,00285953	0,014505241
Fraction within phylum		0,08094901	0,439757443	0,078927279	0,400366267
PTI	-	1,924	2,125	-	1,869

May 2017	Unknown	Picoplancton
Total	**0,631125737**	**0,009814572**
Fraction of total biovolume	0,561153063	0,008726434
Fraction within phylum		0,015550898
PTI	-	1,911

June 2017	Bacillariophyta	Diatoma vulgaris	Fragilaria sp..(type capucina)	Navicula sp.	Nitzschia acicularis	Nitzschia sp.	Ulnaria ulna	Unknown
Total	**0,072527398**	**0,016263868**	**0,000743301**	**0,007657601**	**0,000755158**	**0,026033636**	**0,009747524**	**0,01132631**
Fraction of total biovolume	0,042724132	0,00958065	0,00043786	0,004510907	0,000444846	0,015335784	0,00574203	0,00667205
Fraction within phylum		0,224244471	0,010248552	0,10558218	0,010412044	0,358948985	0,134397818	0,15616595
PTI	-	-	2,54	2,862	3,917	3,917	2,769	-

June 2017	Charophyta	Cosmarium subcostatum
Total	**0,007645979**	**0,007645979**
Fraction of total biovolume	0,004504061	0,004504061
Fraction within phylum		1
PTI		3,214

June 2017	Chlorophyta	Chlamydomonas sp..	Monoraphidium minitum	Monoraphidium	Closteriopsis longissi	Scourfieldia sp.
Total	**0,088571041**	**0,083052293**	**0,004167499**	**0,00081185**	**0,000539399**	**0,014234443**
Fraction of total biovolume	0,052175053	0,048924093	0,002454972	0,000478241	0,000317747	0,008385165
Fraction within phylum		0,937691284	0,047052609	0,009166093	0,006090014	0,160712159
PTI		2,36	3,643	-	-	-

June 2017	Cryptophyta	Cryptomonas sp..	Chroomonas/komma
Total	**1,265179618**	**1,209859484**	**0,055320134**
Fraction of total biovolume	0,745286647	0,712698897	0,03258775
Fraction within phylum		0,956274877	0,043725123
PTI	-	2,364	2,106

June 2017	Cyanobacteria	Ocillatoria sancta	Pseudoanabema sp..
Total	**0,003065594**	**0,00292011**	**0,000145484**
Fraction of total biovolume	0,001805867	0,001720166	8,57009E-05
Fraction within phylum		0,9525431	0,0474569
PTI	-	-	-

June 2017	Ochrophyta	Mallomonas sp.	Pedinellaceae	Synura sphagnicola
Total	**0,06753616**	**0,052325448**	**0,00675358**	**0,008457132**
Fraction of total biovolume	0,039783915	0,030823653	0,00397837	0,004981891
Fraction within phylum		0,774776772	0,099999472	0,125223756
PTI	-	2,125	1,869	2,566

June 2017	Unknown	UK different	Picoplancton	UK (type long sideflagella)
Total	**0,193048725**	**0,176092037**	**0,002722244**	**0,014234443**
Fraction of total biovolume	0,113720325	0,103731551	0,001603608	0,008385165
Fraction within phylum		0,912163691	0,014101332	0,073734977
PTI	-	-	1,911	-

July 2017	Bacillariophyta	Achnanthes sp.	Fragilaria sp.	Nitzschia sigmoidea	Navicula sp.	Ulnaria ulna
Total	**0,034982103**	**0,008066431**	**0,007315951**	**0,00336501**	**0,009459629**	**0,006775083**
Fraction of total biovolume	0,049669648	0,011453193	0,010387617	0,004777839	0,013431337	0,009619661
Fraction within phylum		0,230587367	0,209134107	0,096192329	0,270413376	0,193672822
PTI	-	1,84	2,54	3,917	2,862	2,769

July 2017	Chlorophyta	Chlamydomonas sp.	Microspora sp.	Monoraphidium minutum	Oedogonium sp.	Oocystis lacustris	Scourfieldia sp..	Unknown
Total	**0,042115846**	**0,007459721**	**0,000882473**	**0,000378864**	**0,002426079**	**0,021480208**	**0,006569223**	**0,00291928**
Fraction of total biovolume	0,059798555	0,010591751	0,001252988	0,000537933	0,00344469	0,030498862	0,009327369	0,00414496
Fraction within phylum		0,177123854	0,020953476	0,008995747	0,057604903	0,510026742	0,155979832	0,06931545
PTI		2,36	-	3,643	-	3,013	-	-

July 2017	Cryptophyta	Cryptomonas sp..	Chroomonas/Komma
Total	**0,066779767**	**0,05785787**	**0,008921897**
Fraction of total biovolume	0,094817841	0,082150006	0,012667834
Fraction within phylum		0,866398197	0,133601803
PTI	-	2,364	2,106

July 2017	Cyanobacteria	Mersismopedia glauca	Rhabdoderma (type lineare)	Unknown, other
Total	**0,002112011**	**6,15815E-05**	**0,000479499**	**0,00157093**
Fraction of total biovolume	0,002998758	8,7437E-05	**0,000680821**	0,002230499
Fraction within phylum		0,029157754	0,227034378	0,743807868
PTI		-	-	-

July 2017	Euglenozoa	Euglena
Total	**0,003044439**	**0,003044439**
Fraction of total biovolume	0,004322673	0,004322673
Fraction within phylum		1
PTI	-	3,799

July 2017	Ochrophyta	Uk chrysophyceae	Mallomonas sp..	Pedinellaceae
Total	**0,066782656**	**0,035504429**	**0,008443244**	**0,022834983**
Fraction of total biovolume	0,094821942	0,050411276	**0,011988214**	0,032422452
Fraction within phylum		0,531641465	0,126428692	0,341929843
PTI	-	1,924	2,125	1,869

July 2017	Unknown	UK different
Total	**0,488478559**	**0,488478559**
Fraction of total biovolume	0,693570583	0,693570583
Fraction within phylum		1
PTI	-	-

August 2017	Bacillariophyta	Fragilaria sp..(type capucina)	Fragilaria sp.	Nitzschia acicularis	Ulnaria ulna	Unknown
Total	**0,063282786**	**0,001060325**	**0,045279935**	**0,001241474**	**0,013849424**	**0,001851628**
Fraction of total biovolume	0,148981371	0,002496235	0,106598764	0,002922698	**0,032604542**	0,004359133
Fraction within phylum		0,01675535	0,715517405	0,019617873	0,21884979	0,029259582
PTI	-	2,54	2,54	3,917	2,769	-

August 2017	Charophyta	Closterium venus
Total	**7,90215E-05**	**7,90215E-05**
Fraction of total biovolume	0,000186034	0,000186034
Fraction within phylum		1
PTI		2,81

August 2017	Chlorophyta	Carteria sp.	Chlamydomonas sp.	Gonium pectorale	Monoraphidium	Scourfieldia sp.	UK
Total	**0,049186912**	**0,007590755**	**0,004229453**	**0,014179797**	**0,001051798**	**0,020826204**	**0,0013089**
Fraction of total biovolume	0,115796634		0,009957047	0,033382311	0,00247616	0,049029391	0,00308144
Fraction within phylum		0,154324689	0,085987363	0,28828395	0,0213837	0,423409471	0,02661083
PTI		2,19	2,36		3,643		

August 2017	Cryptophyta	Cryptomonas sp.
Total	**0,050018747**	**0,050018747**
Fraction of total biovolume	0,117754953	0,117754953
Fraction within phylum		1
PTI	-	2,364

August 2017	Cyanobacteria	Pseudoanabema sp.	Unknown (type Rodshaped)	Unknown other type
Total	**0,00476912**	**0,000863343**	**0,003064339**	**0,000841439**
Fraction of total biovolume	0,01122754	0,002032495	0,007214117	0,001980928
Fraction within phylum		0,181027662	0,64253758	0,176434758
PTI	-	-	-	-

August 2017	Dinophyceae (Miozoa)	Tyrannodinium edax	Uk Dinoflaggelata
Total	**0,010449783**	**0,007374996**	**0,003074788**
Fraction of total biovolume	0,024601051	0,017362335	0,007238716
Fraction within phylum		0,705755838	0,294244162
PTI	-	2,09	0

August 2017	Ochrophyta	Chrysophyceae (Chromulina?)	Mallomonas heterospina
Total	**0,169969134**	**0,141030773**	**0,028938362**
Fraction of total biovolume	0,400144119	0,332016954	**0,068127165**
Fraction within phylum		0,82974343	0,17025657
PTI	-	1,924	2,125

August 2017	Unknown	UK different	Picoplancton
Total	**0,077014288**	**0,064548532**	**0,012465756**
Fraction of total biovolume	0,181308298	0,151961212	0,029347087
Fraction within phylum		0,838137102	0,161862898
PTI	-	-	1,911

September 2017	Bacillariophyta	Cocconeis pediculus	Cocconeis placentula	Diatoma vulgaris	Fragilaria sp..(type capucina)	Fragilaria sp..	Navicula sp.	Nitzschia sp.	Rhoicosphenia	Ulnaria ulna	Unknown
Total	**0,066157286**	**0,004603518**	**0,003840837**	**0,01939743**	**0,002431242**	**0,0010279**	**0,01450907**	**0,001587203**	**0,016578149**	**0,00081182**	**0,00137012**
Fraction of total biovolume	0,305045998	0,021226455	0,017709795	0,08944001	0,011210265	0,00473955	0,06690018	0,007318465	**0,076440531**	**0,00374323**	0,00631753
Fraction within phylum		0,069584442	0,058056145	0,293201717	0,036749424	0,01553715	0,21931176	0,02399135	0,250586899	0,01227102	0,02071009
PTI	-	-	-	-	2,54	2,54	2,862	3,917	-	2,769	-

September 2017	Charophyta	Closterium	Cosmarium punctulatum
Total	**0,008602663**	**0,000246016**	**0,008356646**
Fraction of total biovolume	0,039666195	**0,001134362**	0,038531833
Fraction within phylum		0,028597705	0,971402295
PTI		2,81	2,452

September 2017	Chlorophyta	Chlamydomonas sp.	Monoraphidium minutum	Sphaerellopsis fluviatilis	Tetraëdron incus	Treubaria seti	Ulotrichales	Scourfieldia sp.
Total	**0,003677277**	**0,000198168**	**0,000136988**	**0,000451506**	**8,03374E-05**	**0,00022139**	**0,0018795**	**0,000709382**
Fraction of total biovolume	0,016955631	0,000913735	0,000631639	0,002081857	0,000370429	0,00102082	0,00866624	0,003270903
Fraction within phylum		0,053889757	0,037252459	0,12278265	0,021846976	0,06020556	0,51111306	0,192909532
PTI		2,36	3,643	-	-	-	-	-

September 2017	Cryptophyta	Cryptomonas sp.	Komma/rhodomonas/plagioselmis
Total	**0,080580884**	**0,079211442**	**0,001369441**
Fraction of total biovolume	0,371552062	0,365237677	0,006314385
Fraction within phylum		0,983005382	0,016994618
PTI	-	2,364	2,106

September 2017	Cyanobacteria	Mersismopedia sp..	Pseudoanabema sp..	UK (likely sp.irulinaceae)	UK
Total	**0,020730344**	**0,000747555**	**0,013775277**	**0,002575555**	**0,003631957**
Fraction of total biovolume	0,09558597	0,003446915	0,063516707	0,011875681	0,016746667
Fraction within phylum		0,036060893	0,664498224	0,124240835	0,175200048
PTI	-	-	-	-	-

September 2017	Dinophyceae (Miozoa)	Dinophyceae (Miozoa)
Total	**0,00054795**	**0,00054795**
Fraction of total biovolume	0,002526556	0,002526556
Fraction within phylum		1
PTI	-	-

September 2017	Euglenozoa	Euglena chlamydophora
Total	**0,000535118**	**0,000535118**
Fraction of total biovolume	0,002467386	0,002467386
Fraction within phylum		1
PTI	-	3,799

September 2017	Ochrophyta	Characiopsis	Crysophyceae	Mallomonas caudata	Pedinellaceae
Total	**0,015708054**	**2,03345E-05**	**0,008913061**	**0,006075611**	**0,000699048**
Fraction of total biovolume	0,072428591	9,37607E-05	0,041097415	0,028014161	0,003223254
Fraction within phylum		0,001294526	0,567419783	0,386783188	0,044502503
PTI	-	0	1,924	2,349	1,869

September 2017	UK	UK different	Picoplancton
Total	**0,020336852**	**0,014431428**	**0,005905424**
Fraction of total biovolume	0,09377161	0,066542169	0,027229442
Fraction within phylum		0,709619557	0,290380443
PTI	-	-	1,911

October 2017	Bacillariophyta	Achnanthes sp.	Amphora sp.	Fragilaria sp. (type capucina)	Fragilaria sp.	Hippodonta capitata	Mastogloia sp.	Navicula	Nitzschia acicularis	Nitzschia sigmoidea	Nitzschia sp.	Rhoicosphenia abbreviata	Surirella sp.	Ulnaria ulna	Unknown
Total	**0,051948129**	**0,00152959**	**0,024126064**	**0,002943889**	**0,003158391**	**0,00239574**	**0,00052083**	**0,002624015**	**8,46738E-05**	**0,000122951**	**0,002379086**	**0,002350893**	**0,00564493**	**0,00313381**	**0,00093327**
Fraction of total biovolume	0,303795035	0,00894511	0,141090326	0,017215998	0,018470415	0,014010398	0,003045836	0,015345359	**0,000495176**	0,000719022	0,013913006	0,01374813	0,03301183	**0,01832664**	0,00545778
Fraction within phylum		0,029444556	0,464426044	0,056669782	0,060798936	0,04611793	0,010025958	0,050512212	0,001629967	0,0023668	0,045797344	0,045254623	0,10866483	0,06032567	0,01796535
PTI	-	1,84	-	2,54	2,54	-	-	2,862	3,917	3,917	3,917	-	-	2,769	-

October 2017	Charophyta	Closterium sp.
Total	**7,84001E-05**	**7,84001E-05**
Fraction of total biovolume	0,000458488	0,000458488
Fraction within phylum		1
PTI		2,81

October 2017	Chlorophyta	Carteria sp.	Coelastrum microporum	UK
Total	**0,012823814**	**0,000929049**	**0,011674962**	**0,000219804**
Fraction of total biovolume	0,074994251	0,005433119	**0,068275709**	0,001285424
Fraction within phylum		0,072447136	0,910412568	0,017140297
PTI		2,19	3,247	-

October 2017	Cyanobacteria	Cyanothece	UK (likely sp.irulinaceae)	Leptolyngbya sp.	Pseudoanabema catenata	Ocillatoria sp.	Phormidium	Pseudoanabema sp.	Synechococcaceae	Romeria okensis	Unknown
Total	**0,049950352**	**0,001270607**	**0,000313097**	**0,000866368**	**4,81316E-05**	**0,004204985**	**0,00063196**	**0,001344958**	**0,000490538**	**0,000251962**	**0,040527745**
Fraction of total biovolume	0,292111942	0,007430568	0,001831007	0,005066562	0,000281476	0,024590945	0,00369573	**0,007865377**	0,002868687	**0,001473486**	0,237008105
Fraction within phylum		0,025437398	0,006268167	0,017344589	0,000963589	0,084183292	0,012651761	0,026925902	0,009820505	0,00504425	0,811360548
PTI	-	-	-	-	-	-	-	-	-	-	

October 2017	Dinophyceae (Miozoa)	UK
Total	**0,003696665**	**0,003696665**
Fraction of total biovolume	0,021618263	0,021618263
Fraction within phylum		1
PTI	-	-

October 2017	Euglenozoa	Euglena
Total	**0,000512731**	**0,000512731**
Fraction of total biovolume	0,002998475	0,002998475
Fraction within phylum		1
PTI	-	3,799

October 2017	Ochrophyta	Chrysophyceae	UK (Poterioochromonas sp.. / Monas sp.. ?)
Total	**0,004433432**	**0,000952401**	**0,003481031**
Fraction of total biovolume	0,025926912	0,005569686	0,020357226
Fraction within phylum		0,214822581	0,785177419
PTI	-	1,924	-

October 2017	UK	UK different	Picoplancton
Total	**0,047553772**	**0,043103357**	**0,004450415**
Fraction of total biovolume	0,278096635	0,252070402	0,026026233
Fraction within phylum		0,906412989	0,093587011
PTI	-	-	1,911

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