



Preface

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Abstract

Two lakes situated in the same catchment in SE Norway were studied during the growing season of 2013. Lake Østensjøvannet is a small and shallow lake situated upstream from a larger and deeper recipient, Lake Årungen. The objective was to determine to what extent the water quality in Lake Årungen is influenced by processes in Lake Østensjøvannet.

The fieldwork was carried out during the growing season of 2013. The water samples were collected from vertical depth series in the lakes. Water samples were collected at an upstream and downstream location in the connecting stream, and discharge measurements were carried out at a turbulent section of the stream. Physical, chemical and biological analysis of the water quality was performed according to Norwegian standards.

The physical, chemical and biological differences were evaluated and were related to morphometric and hydrological characteristics and limiting factors for phytoplankton growth.

The phytoplankton composition was compared in the two lakes to determine the level of interconnection between the lakes. The nutrient retention capacity was evaluated in the upstream lake with focus on internal phosphorus loading. Retention of different water chemical variables in the interconnecting stream was evaluated at different discharge intensities.

The stability of the stratification regime was governed by lake size and volume. The mixing barrier in Lake Østensjøvannet was less resistant against wind force, which resulted in homogenous vertical concentration distribution of chemical parameters. Low precipitation during the second half of the season however resulted in steeper vertical gradients. The vertical mixing of the water column was determining for phytoplankton composition. At shallow mixing and low insolation, phytoplankton groups with abilities for vertical positioning were favored. Shallow mixing was limiting for diatom algae in both lakes at the first half of the season, while nutrient limitation caused the diatom decline in Lake Årungen. The phytoplankton growth in Lake Østensjøvannet was dominated by autotrophic picoplankton at the latter half of the season.

Transport of algae cells downstream was limited by increased retention time and low discharge. The upstream lake retain particles and nutrients, however the retention efficiency was increased as a result of prolonged retention time. Internal phosphorus loading in Lake Østensjøvannet was not supported by this study. Retention of particles and total phosphorus occurs in the combining stream during low flow. However, phosphate was increased downstream which could have consequences for the phytoplankton growth in the recipient.

Sammendrag

Vannkvaliteten i en innsjø kan påvirkes av innsjøer som ligger oppstrøms spesielt hvis de har kort hydraulisk oppholdstid. To eutrofe innsjøer i et nedbørsfelt i Akershus fylke sørøst i Norge ble studert under vekstsesongen 2013. Østensjøvannet er en grunn innsjø som drenerer til Årungen, en større og dypere innsjø. Målet med oppgaven var å vurdere i hvilken utstrekning vannkvaliteten i Årungen påvirkes av vannkvaliteten i Østensjøvannet. Vannprøver ble hentet fra flere dyp i innsjøene og fra en oppstrøms og nedstrøms lokalitet i Bølstadbekken som drenerer fra Østensjøvannet til Årungen. Vannføringsmålinger ble foretatt for å koble konsentrasjonsforandringer i sentrale vannkjemiske variable opp mot vannføringsintensitet. Fysiske, kjemiske og biologiske analyser ble foretatt i henhold til Norsk standard. Dynamikken i de to innsjøene ble sammenlignet med vekt på morfometriske og hydrologiske forskjeller og begrensende faktorer for algevekst. På grunnlag av algesammensetning ble graden av kobling mellom de to innsjøene vurdert i forhold til fytoplanktonmigrasjon. Mobilisering av fosfor ble vurdert i Østensjøvannet og Bølstadbekken, med fokus på intern fosforgjødsling. Resultatene viste at dybdeforholdene var avgjørende for stratifiseringsregimet i innsjøene. De grunne forholdene i Østensjøvannet gjør at innsjøen er mer utsatt for vindindusert blanding. Vannutveksling mellom øvre og nedre vannlag resulterte i homogen fordeling av kjemiske parametere i vannsøylen. Nedbørsfattigeværforhold i den senere delen av vekstsesongen forlenget oppholdstiden i innsjøen og medførte skarpere kjemiske gradienter. Den vertikale blandingen var avgjørende for fytoplanktonsammensetningen tidlig i vekstsesongen i begge innsjøene. Ved grunn sirkulasjon og lav lysintensitet ble fytoplanktongrupper med adaptasjoner for vertikal posisjonering fremmet. Næringsstoffer var begrensende vekstfaktor i Årungen sent i vekstsesongen. Fytoplanktonsamfunnet i Østensjøvannet ble mest sannsynlig begrenset av konkurranse fra autotrofe pikoplankton. Det var relativt store forskjeller i fytoplanktonsammensetning mellom innsjøene. Lav vannføring og lengre oppholdstid begrenset transport av alger til nedstrøms områder. Partikler og næringsstoffer ble holdt igjen i Østensjøvannet, og retensjonskapasiteten økte som følge av forlenget oppholdstid og lav vannføring. Intern fosforgjødsling i Østensjøvannet ble ikke påvist i Østensjøvannet under vekstsesongen 2013. Partikler og fosfor ble holdt igjen i Bølstadbekken under lav vannføring. Videre, økte fosfat nedstrøms, noe som kan få konsekvenser for algeveksten i Årungen.

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1. Introduction

Eutrophication has been recognized as one of the main environmental problems regarding fresh water lakes (Dillon & Rigler 1974; Schindler & Fee 1974). Nutrient loading into water bodies alters the ecosystem structure and limits the possible uses in terms of drinking water and recreational purposes. Eutrophication is a problem which affects many lowland lakes situated in arable landscapes, where human activity such as agriculture and urbanization has resulted in extensive external loading of nutrients to rivers and lakes (Granéli 1999; Jeppesen et al. 1999). Increased nutrient loading and especially phosphorus usually results in an increase of phytoplankton biomass in lakes (Dillon & Rigler 1974; Håkansson & Jansson 1983; Schindler 1977; Zohary et al. 2010). The lake response to nutrient loading often leads to shift in species composition, towards green algae and cyanobacteria dominance (Reynolds 1998; Tilman et al. 1986).

Eutrophication of freshwater lakes have previously been related to the direct discharge of sewage water (Reynolds & Davies 2001). Such point sources have during the last decades been reduced in many European countries due to improvement of wastewater management, however the diffuse sources such as agricultural runoff are more difficult to identify and regulate (Jeppesen et al. 1999). Although external loadings have been significantly reduced in many water bodies, it has not resulted in the expected improvement in water quality (Granéli 1999). The inertness of the system to adjust to a reduction of nutrients is mainly explained by a compensation of internal phosphorus loading (Håkansson & Jansson 1983; Jeppesen et al. 1999). The response to reduced nutrient loading may however, be significantly different in lakes with different morphology and hydrology. Lakes situated within the same watershed can differ fundamentally in their trophic structure and biotic and abiotic interactions (Cottenie et al. 2001). Lake recovery is not entirely dependent on the reduction of external nutrient sources, but must also take into account lake specific properties, internal processes and catchment interactions.

Nutrient retention is a fundamental property of every aquatic ecosystem (Kõiv et al. 2011). Shallow upstream lakes often function as a sink for particles and nutrients. However an upstream lake can under certain conditions become a nutrient source, especially during the summer stagnation (Granéli 1999; Hillbricht-Ilkowska 1999).

The understanding of lake responses is important in order to evaluate the function of individual lakes within in the catchment area.

Lake Østensjøvannet and Lake Årungen are two lowland lakes situated in SE Norway. The catchment is situated under the marine limit and the background nutrient loading is naturally high (Yri 2003). Up until the end of the 1900th century Lake Årungen is believed to have been rather unaffected by human activities, possessing good self-purification mechanism to buffer nutrients input from the catchment. The years following the SWW, anthropogenic sources such as untreated sewage water and agricultural runoff resulted in extensive nutrient loading to Lake Årungen and Lake Østensjøvannet (Grøterud & Rosland 1981) and both lakes became hypertrophic and dominated by cyanobacteria (Andersen 1982; Erlandsen & Grøterud 1980). During the last decades, efforts to improve sewage treatment and reducing surface runoff from agriculture has improved the water quality, although poor water quality are still prevailing with toxic blooms of cyanobacteria being the main challenge. (Romarheim et al. 2012; Yri 2003). Both lakes has been characterized with exceedingly poor ecological status, in accordance to guidelines given by the water framework directive (WFD) (pura.no 2009). The water quality in Lake Årungen has been given much public attention since the lake is a popular area for recreational activities and hosts the national rowing arena. Achieving good water quality for swimming is one of the main goals of the municipal water management organ PURA (pura.no 2009). Lake Østensjøvannet, which is situated upstream from Lake Årungen, receives substantial runoff from its surrounding agricultural fields (Bechmann & Deelstra 2013). An estimated 30% of the yearly catchment runoff to Lake Årungen originates from Lake Østensjøvannet (Grøterud & Haaland 2007), Lake Årungen can accordingly be significantly affected by the water quality in Lake Østensjøvannet. During summer stagnation, internal phosphorus loading may result in downstream transport of bioavailable phosphorus. There has also been a concern that toxic cyanobacteria blooms in Lake Østensjøvannet could be transported to Lake Årungen (Grøterud & Rosland 1981). Biomanipulation, by removing planktivorous fish has been under consideration to improve water quality in Lake Østensjøvannet and reduce the nutrient loading to Lake Årungen (Skovgaard et al. 2011) .

The aim of this study is to evaluate lake dynamic differences in two previously hypertrophic lakes situated within the same catchment. Processes in the upstream lake may significantly affect the downstream recipient, by either acting as an effective buffer for catchment derived inputs or

as a source of nutrients, mainly by the risk of internal loading of phosphorus. The function of upstream water bodies is therefore important to understand when considering different management strategies. To what extent is the water quality in the upstream lake determining for the downstream recipient? Are nutrients and particles being retained in the upstream lake, preventing further nutrient loading to the downstream recipient?

2. Objectives

- To compare thermal stratification and mixing regimes, optical properties and nutrient cycling in the two lakes with emphasize on morphometric and hydrological differences.
- Identify factors controlling phytoplankton growth and composition in the two lakes.
- Assess the role of the upstream lake and the level of interconnection, in terms of nutrient retention and phytoplankton migration.
- Evaluate nutrient retention processes in the interconnecting stream, by looking at concentration changes from an upstream to a downstream location in relation to different discharge intensities.

3. Theoretical Background

3.1 Main factors controlling phytoplankton growth

3.1.1 Light (optical properties)

The optical properties of lake water, both quality and quantity, is important for controlling phytoplankton growth (Zohary et al. 2010). The transparency of lake water and light attenuation in the mixing depth are largely dependent on light absorbing components (Diehl 2002). Such components are made up of humic substances and clay particles that can ultimately limit the primary production in lakes. Phytoplankton itself can also absorb light and shade for other species (Diehl 2002).

The optical quality affects thermal, hydrodynamic and chemical properties of water bodies (Wetzel 2001). In water, the incident light is attenuated with increasing depth by absorption and scattering mechanisms (Wetzel 1991). Incoming solar radiation are absorbed by the progressive removal of photons, and becomes more diffuse due to various scattering mechanisms (Kirk 1985). The removal of photons by absorption and upward scattering at any given depth, results in diminishing downward irradiance in an approximate exponential manner. The layer of water in which irradiance of photosynthetic active radiation (PAR) falls to 1 % of the surface value is commonly used (Kirk 1985) and denoted as the euphotic depth, (Z_{eu}). The wave lengths of PAR lies approximately between 390 and 710nm and the different wave lengths are selectively attenuated in the water column. The absorption in the red and infrared end of the spectra is strong in water, which governs the thermal behavior in lakes. Approximately 53% of total incident light is absorbed in the first meter and results in a rapid heating of the upper layer of the water column (Wetzel 2001). The strong absorption in the red end of the visible spectrum, results in deeper penetration of blue light. However, dissolved organic matter selectively absorb the shorter wavelengths (the blue and green end of the spectra) and shifts the apparent color to yellow and red (Wetzel 1991). Consequently; in humic stained water the wavelengths penetrating to significant depth are in the yellow and red portions of the spectra.

3.1.2 Temperature, stratification and mixing regimes

The thermal properties of the lake coupled with lake morphology controls stratification processes in lakes. In temperate lakes, the water column is usually stratifies during summer due to increased

temperature and insolation (Jonsson et al. 2001). Lake stratification is caused by the development of thermal density gradients in the water column, where less dense water masses are layered on top of denser water masses (Jonsson et al. 2001). The circulation of the water column will be controlled by the stratification regime. The upper layer (epilimnion) will be in circulation, while the deeper layer (hypolimnion) will become stagnant. The density gradient between mixed and stagnant layer (thermocline), can be either steep or extend over several of meters.

The mixing regime of the lake will be of high consequence for the phytoplankton community. Intense mixing is favoring towards non-motile species like green algae and diatoms (Harris & Baxter 1996; Huisman et al. 2004; Reynolds et al. 1983). These groups are dependent on intense mixing in order to counteract sinking velocities. During poor light and mixing conditions the best positioning for phytoplankton species is close to the surface, which will consequently favor motile species (Huisman et al. 2004; Reynolds et al. 1983), such as cyanobacteria.

The light availability for phytoplankton depends on the ratio between mixing depth (Z_{mix}) and euphotic depth (Z_{eu}). At low ratio the phytoplankton cells will spend more time in insufficient light conditions, compared to higher $Z_{\text{eu}}/Z_{\text{mix}}$ ratios (Nicklisch 1998). The time spent in circulation beneath the Z_{eu} will subject phytoplankton cells to light deviancies. At near surface stratification densities light becomes less limiting (Reynolds 1998) and nutrient limitation will become more determining (Padisak et al. 2010).

3.1.3 Grazing

Grazing control on phytoplankton has been recognized by e.g. Jeppesen et al. (1999). Large zooplankton, especially in the *Daphnia* taxonomic group, is able to filter large volumes of phytoplankton per day. Trophic compositions in lakes which favor the dominance of larger zooplankton can constitute an effective top down control on phytoplankton. This relationship will only briefly be mentioned, since zooplankton is not a part of the current study. It is however, important to mentioned this, as an important control mechanism, since many management strategies has focused on the top down control to mitigate phytoplankton by removal of planktivorous fish (Jeppesen et al. 1999).

3.1.4 Nutrients

The two nutrients most associated with eutrophication in lakes are Phosphorus (P) and Nitrogen (N). Efforts to reduce anthropogenic inputs of N and P, are central in order to mitigate eutrophication (Conley et al. 2009; Schindler 2006). Nutrient enrichment in lakes are usually accompanied by an alteration in phytoplankton community by replacing small flagellated taxa with increasing proportions of green algae and cyanobacteria (Reynolds 1998; Tilman et al. 1986).

Silicon (Si) is another major nutrient influencing algal production in many lakes due to the importance for the growth of diatom algae, which utilize Si to build their frustules. The major source of silicon is from the degradation of aluminosilicate minerals in the catchment, and are therefore determined mainly by natural background concentrations (Wetzel 2001). Si loadings to lakes derive largely from influent surface waters of rivers, but in certain lakes ground water can be the primary source of Si.

3.2 Phosphorus control and lake eutrophication

The relationship between mean total phosphorus (TP) concentration and chlorophyll *a* (*chl_a*) concentrations has been recognized by e.g. Dillon & Rigler (1974). High TP concentrations after spring mixing have a direct relationship with high summer mean concentrations of *chl_a*.

Schindler & Fee (1974) recognized phosphorus as the primary limiting nutrient for algal growth in freshwater lakes in their “Whole lake experiment” on lake 227 and double basin lake 226, within the lake experimental area (ELA) in Canada. By selectively fertilizing the lakes Schindler and Fee showed the importance of phosphorus for algae growth in contrast to carbon and nitrogen.

Nitrogen is an equally important nutrient for primary producers in freshwater and the strong focus on phosphorus as the number one limiting nutrient for phytoplankton communities has been criticized by e.g. Lewis & Wurtsbaugh (2008). In a management perspective, efforts to reduce phosphorus has proven to be the most effective measure to mitigate eutrophication (also recognized by Lewis & Wurtsbaugh) and by exerting phosphorus control, nitrogen inputs may also be reduced (Jeppesen, E et al. 2007)

Eutrophication is commonly associated with low N:P values (Downing & McCauley 1992). Studies on ELA has shown that lakes enriched in phosphorus compared to nitrogen (low N:P

ratio) are more likely to develop cyanobacteria blooms (Schindler 1977). Cyanobacteria can fix and utilize atmospheric nitrogen and will have a competitive advantage over other phytoplankton groups when dissolved nitrogen becomes exhausted in the water column. Different sources of phosphorus loading have different effects on the N:P relationship in lakes (Downing & McCauley 1992). Agricultural runoff and sewage inputs are especially known to contribute to a lower N:P ratio.

3.3 Phosphorus retention and release from lake sediments

In most lakes there is a net retention of phosphorus in the sediments (Boström et al. 1988). Lakes often function as a buffer system for external phosphorus loading from the catchment. Particle associated phosphorus (PP) will eventually settle and accumulate in the lake sediments and phosphate can adsorb to suspended particles in the water column or co-precipitate with metals and settle in the lake sediments. The total concentration of phosphorus in sediments are generally higher in the deeper parts of the lake (Granéli 1999; Håkansson & Jansson 1983) due to continuous transport and translocation from shallower areas to deeper accumulation areas. This phosphorus flux can be reversible; consequently sediments can become re-suspended by different physical, chemical and biological processes and thereby result in internal phosphorus loading (Boström et al. 1988; Granéli 1999; Håkansson & Jansson 1983).

Different fractions of phosphorus are more or less subjected to release. Some fractions are strong precipitates, with low solubility and less sensitive to changed redox conditions, pH and concentration gradients. Four main phosphorus fractions deposited in lake sediments are described by Håkansson & Jansson (1983). Including i) NAI-P non-apatite inorganic phosphorus (this fraction is mainly made up by phosphorus associated with iron and aluminum complex), ii) apatite-P iii) organic-P (such as structural elements of settling dead organisms and humic complexes) and iv), very loosely sorbed P. The relative importance of these fractions depends on the external supply of apatite minerals, organic matter and complex forming agents like iron and aluminium and also to a large extent the productivity of the system. In order for the lake to become self-loading with phosphorus, the sediments need to be saturated. Lakes which has been subjected to heavy external loading from e.g. sewage, are more predisposed to become dominated by an upward phosphorus flux rather than downward (Håkansson & Jansson 1983). The pre-history, trophic state and the sorption capacity of the sediments are essential factors

controlling phosphorus release from lake sediments. Changes in chemical factors like pH, redox potential, chelating agents and biochemical enzymatic reactions can further induce P release from the sediments and are often driven by biological activities in the lake. Productive lakes with higher metabolism are therefore, more subjected to develop conditions that increase the risk of internal phosphorus loading.

3.3.1 Phosphorus release under reducing conditions at the sediment-water interface

In oxidized sediments, phosphorus is co-precipitated with iron (III) (Fe^{3+}) mainly in the form of iron (III) hydroxide gels. Such aggregates may form in the oxidized surface sediments or precipitated in the overlying water (Håkansson & Jansson 1983). Sedimentation of iron oxides likely occurs by flocculation by self-association, or by association with other particulate matter. Phosphorus concentrations in the water column will then be partly regulated by the supply of iron (III), which mainly derives from groundwater input and catchment runoff.

Under anoxic (reducing) conditions, iron (III) is reduced to iron (II) (Fe^{2+}), by this process both iron and phosphate is returned to solution. Iron (III) is reduced at an oxygen level of approximately 0,1 mg/L, and a redox potential of 200mV, but the presence of nitrate and manganese, which are reduced before iron can delay the process (Håkansson & Jansson 1983). Phosphorus release from sediments can also occur from well aerated sediments (Bostrom & Pettersson 1982). Thus, there must not be anoxic conditions to release phosphorus from the sediments.

Sulfides in the sediments originate from the reduction of sulfate (SO_4^{2-}) to sulfide (S^{2-}), when the oxygen in sulfate is used in the oxidation of organic matter, in the absence of dissolved oxygen. One of the reaction products is H_2S and in the presence of iron (II) in the sediments, precipitation of FeS can occur through ligand exchange. FeS is highly insoluble and forms both amorphous and stable, crystalline phases that darken the color of anaerobic sediments. This process requires completely reduced sediments, especially common for productive lakes where the oxidation of organic material is so intense, that the redox potential drops below 100mV and completely anoxic conditions prevail.

3.3.2 Phosphorus release at high pH levels

Phosphorus mobilization from iron and aluminum hydroxide gels can be pH mediated when pH increase in the alkaline regime. Adsorbed phosphorus are replaced by hydroxide ions and thus returned to solution (Håkansson & Jansson 1983). Increased pH can also shift the adsorption equilibrium resulting in the release of P from the sediments to the water column. pH can be raised substantially as a result of increased photosynthetic activity. The consumption of CO₂ is higher than the replacement from air-water diffusion. When CO₂ becomes depleted the chemical equilibrium in the water is altered and the pH is raised. Productivity is usually higher in the surface water and phosphorus release due to increased pH, is more likely to happen in the sediment-water interface of shallow lakes, where the entire productive water column is in contact with the sediments. In stratified lakes, re-suspension of particles from the sediments into the productive layer can contribute to the overall phosphorus release (Koski-Vahala & Hartikainen 2001). A high pH in the oxidized surface water will in addition reduce the re-precipitation of iron (III) (Boström et al. 1988).

3.3.3 Equilibrium reactions

The stability of inorganic precipitates of phosphorus may also be a result of equilibrium reactions (Håkansson & Jansson 1983). The inorganic compounds are sensitive to concentration changes in the water column or in the sediment pore water. The equilibrium between adsorbed phosphorus and phosphorus in solution is important in the water column, since it will determine the availability of phosphorus. Particles may release phosphorus when the concentration in the water becomes less than the equilibrium concentration, and adsorb phosphorus when the concentration in the water is higher than the equilibrium concentration (Wang et al. 2008). Inorganic particles in the water column can accordingly act as a sink for phosphorus. Phosphorus which is adsorbed to particles will eventually be deposited in the lake sediments or flushed out of the lake. In the sediments, adsorbed phosphorus is regulated by the equilibrium between sediments and the pore water. It is from this relatively small phosphorus pool that transport back to the water column occurs (Håkansson & Jansson 1983).

3.3.4 Wind re-suspension of phosphorus

In shallow areas of the lake, particles can be re-suspended from the sediments by wind force. Re-suspension of particles by wind can act to buffer the concentration of phosphate in the overlying water, given that the phosphorus concentration adsorbed to the particles is less than the concentration in the water. Conversely, wind induced re-suspension of phosphorus can also impose a problem when loosely bound P may become available for algae growth, it may also reduce the retention capacity of the lake when particle associated P become suspended and flushed out of the lake (Jeppesen, E et al. 2007).

3.3.5 Bioturbation

Bioturbation is the mechanical mixing of sediments by bottom dwelling fishes, macro fauna and meiofauna. This is more common under aerobic conditions due to the natural limitations, since the benthic fauna can be very sensitive to changes in the abiotic environment, such as oxygen, pH and redox conditions (Håkansson & Jansson 1983). Benthic fauna can bring phosphorus into suspension by burrowing in, and feeding from the sediments. Bottom foraging fish such as bream, roach, perch, eel and whitefish may contribute to internal phosphorus loading, when present in sufficient numbers, by either re-suspend phosphorus in their search for food or translocate P to the water column by bottom feeding. The fish may digest lot of excessive phosphorus and later excrete it into the water masses (Schindler et al. 1993).

The effect of bioturbation is mainly physical mixing of the sediment surface layer with increased phosphorus outflow due to the mixing or decreased release due to improved redox potential in reduced sediments.

3.4 Phosphorus transport in rivers

Phosphorus can to some degree be retained in rivers by different binding mechanisms such as sedimentation, adsorption and biological assimilation (Jarvie et al. 2005; Mainstone & Parr 2002). Nutrient particles are either transported, settled to the sediments, or re-suspended in the stream depending on grain size and velocity (Hillbricht-Ilkowska 1999). The fine sized and organically rich particles are often accumulated in the impoundment sections of streams (Selig & Schlunbaum 2002). Jeppesen et al (1999) showed that P and N retention in streams were low in stream sequences dominated by coarse stream bed substrate and high within macrophyte patches

and vegetated riparian zones. The retention of P in streams is mainly a result of sedimentation of particle-associated P. Dissolved P can adsorb to sediments or be assimilated by macrophytes and benthic algae, which may also contribute to some retention (Jeppesen et al. 1999). The sediment trapping in rivers are more efficient at low flow conditions (Reddy et al. 1999). During periods of high discharge large part of the accumulated load will be flushed out from the river sediments. Riverbed sediments play an important role in buffering phosphate concentration. Phosphate release from the sediments is dependent on a diffusion gradient across the benthic interface, with higher concentration of particulate bound exchangeable phosphorus in the sediments compared to phosphate concentration in the water. At low flow conditions there is relatively long contact time between water column and bed sediments, and the surface area of the sediments is relatively large compared to the overlying water volume. High temperatures and low redox potential can also result in phosphorus release from stagnant parts of the river during low flow conditions (Mainstone & Parr 2002).

4 Methods and Materials

4.1 Study area

4.1.1 The Lake Årungen catchment

The catchment is situated below the marine limit and comprise of surficial deposits made up of mainly marine clay and glacial till on top of a Precambrian granite and gneiss bedrock. The catchment area is 51 km², including Lake Østensjøvannet and Lake Årungen which comprises of 1.6 km². The land use within the catchment is mainly agricultural (53%), forest (34 %) and urban settlements (10%) (figure 1).

4.1.2 Lake Årungen

Lake Årungen is a lowland lake situated in SE Norway (59°41'093 N, 10°44'844 E). The lake is 3 km long and has a mean width of 450m. The outlet is situated in the north (fig 1) and discharges into Bunnefjorden, a part of the Oslo fjord. The inlet from Lake Østensjøvannet is situated on the east side and the road E6 passes the lake on the NV side. The Lake has a surface area of 1.2 km², a maximum and mean depth of 13.2m and 8m respectively and the hydraulic retention time is 4.5 months (table 1). The lake substantial inputs of erosion material from the catchment and has a high sedimentation rate of approximately 0.7 cm a year (Riise et al. 2010).

Table 1 Morphometric and hydraulic characteristics for Lake Årungen. Showing catchment area (A_C), Lake area (A_L), maximum depth (Z_{max}), mean depth (Z_{mean}) annual inflow (Q_{in}) theoretical retention time (T_w).

A_C (km ²)	A_L (km ²)	Z_{max} (m)	Z_{mean} (m)	V (Mm ³)	T_w (months)
52	1.2	13	8	9.2	4.5

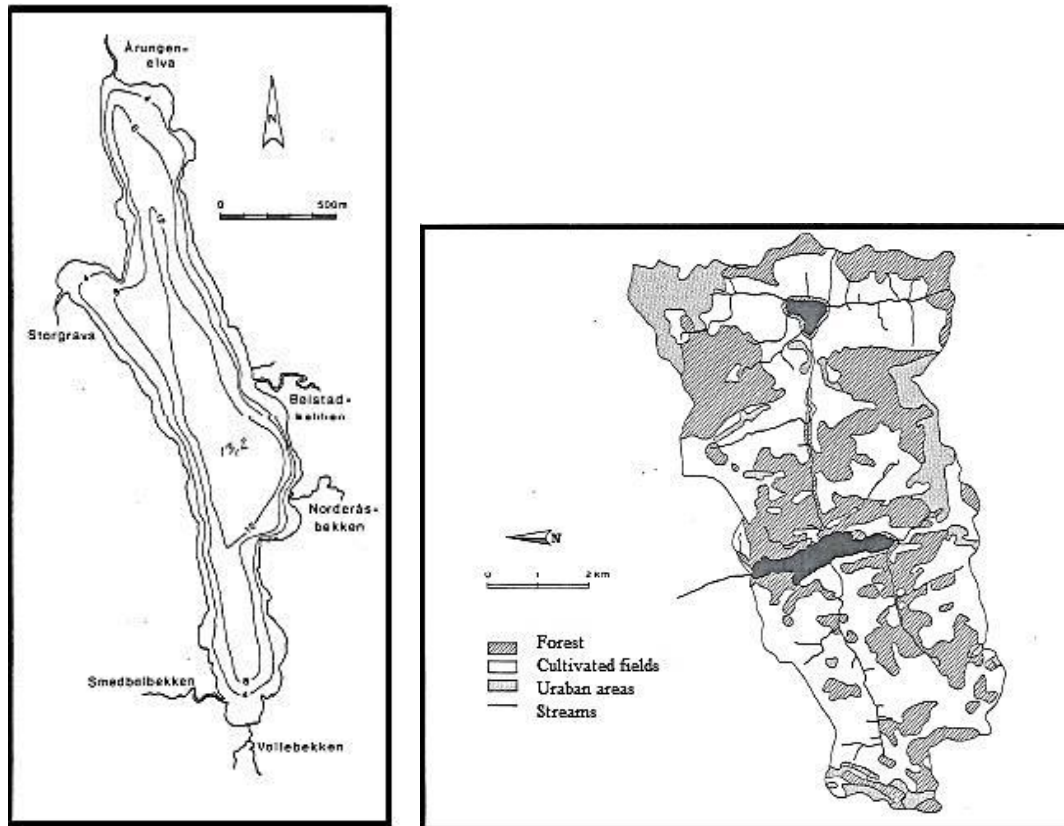


Figure 1 Morphometric map of Lake Årungen (l.h.s) and its catchment (r.h.s) including land use properties. Source (Borgstrøm et al. 1984)

4.1.3 Lake Østensjøvannet

Lake Østensjøvannet is a small and shallow, eutrophic lake situated 4 km East of Lake Årungen (59°41'188 N, 10°49'472 E). It has a catchment area of 14 km², which comprises of approximately 50 % cultivated land, 30% forest and 15% urban areas. It has a lake area of 0.34 km² and a lake volume of 1.31 Mm³. The maximum depth is 7.1 m and the mean depth is 3.9m (table 1 & figure 2). The hydraulic retention time is approximately 72 days. Lake Østensjøvannet has since the year of 1992 been a protected nature reserve, mainly because of its rich bird life (Lovdata.no 1992). Recreational activities on the lake are prohibited between the months of May until October. The road E18 borders to the lakes southern side, and on the North Slope there is the agricultural field, Skuterud, which also is an experimental field for surface runoff, managed by research institute Bioforsk.

Table 2 Morphometric and hydraulic characteristics for Lake Østensjøvannet. Showing catchment area (A_C), Lake area (A_L), Lake volume (V), maximum depth (Z_{max}), mean depth (Z_{mean}) annual inflow (Q_{in}) theoretical retention time (T_w).

A_C (km ²)	A_L (km ²)	V (Mm ³)	Z_{max} (m)	Z_{mean} (m)	Q_{in} (Mm ³)	T_w (days)
14	0.34	1.31	7.1	3.9	6.62	72

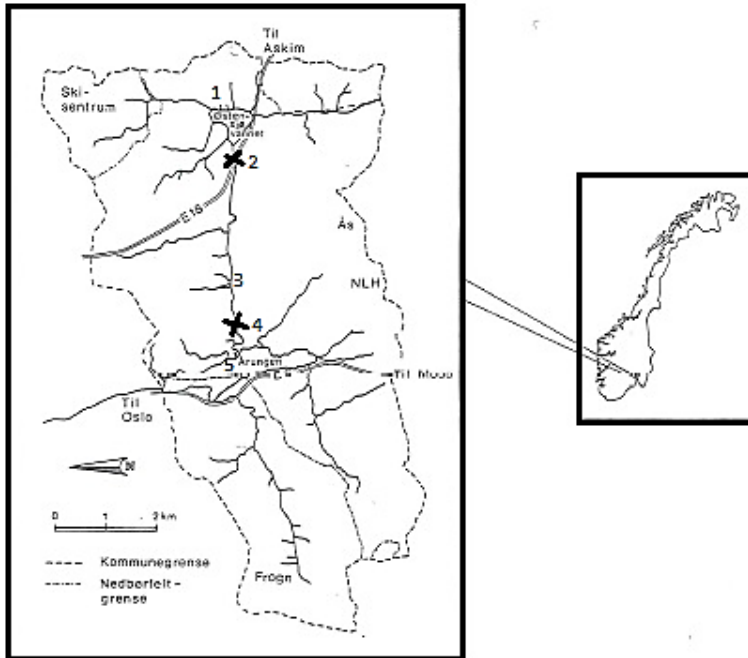


Figure 2 The catchment of Lake Årungen, including Lake Østensjøvannet and Bølstadbekken. The 5 locations are marked by numbers 1 to 5, where 1) Lake Østensjøvannet 2) Bølstadbekken upstream 3) location for discharge measurements 4) bølstadbekken downstream and 5) Lake Årungen.

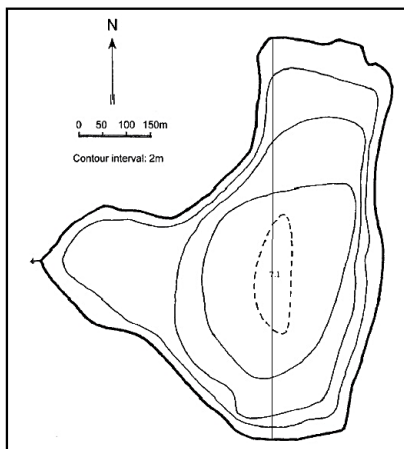


Figure 3 Morphometric map of Lake Østensjøvannet. Source (Grøterud & Haaland 2007)

4.1.4 Bølstadbekken

The outlet from Lake Østensjøvannet, Bølstadbekken, is located to the west (figure 2). The first part of the stream drains mainly through agricultural areas before entering a gully of ravines and at the lower parts, the stream is meandering through dense and lush vegetation before entering Lake Årungen. Water samples for physical and chemical analysis were taken at one upstream and one downstream location. The upstream location (59°41'183 N, 10°48'567 E) is characterized by slow flowing water through a cultivated field, located between the lake and the road E 18. The downstream location is also characterized by low water velocities, next to field of grazing livestock. The two locations are marked in fig 2 (location 2 & 4). The locations for discharge measurements are located in between the two sampling spots at (59°41'212' N, 10°45'826'E) in an area of high water velocities and turbulence (figure 2, location 4)

4.2 Fieldwork

The field work was carried out during the months of May to October in 2013, with a frequency varying between three to four weeks. A total of 8 water sample series was collected from the deepest location of the lakes, for chemical and biological analyses. Water samples were also collected from the connecting stream, Bølstadbekken at one upstream location (fig 2, location 2) and one downstream location (fig 2 location 4).

At Lake Østensjøvannet an inflatable boat was used for water sampling. A GPS (Garmin, Oregon 450) was used for navigation with an accuracy of ± 5 m.

4.2.1 Water sampling

Dissolved oxygen (DO) concentration (mg L^{-1}) and saturation (%), conductivity ($\mu\text{S cm}^{-1}$) and temperature ($^{\circ}\text{C}$) were measured in the field, over the deepest part of the lakes at 0.5 m, 1 m and continuously downwards with a 1m interval. The measurements were taken using a 3420 Multi optic sensor device (WTW, Weiheim Germany), with an oxygen sensor (FDO 925-p) and a conductivity sensor (TetraCon 925).

Water samples used for physical and chemical analysis were collected along vertical profiles at the deepest part of the lakes, using a modified Rüttner water sampler, at three depths in Lake Østensjøvannet (1, 3 and 6 m) and seven depths in Lake Årungen (0.5, 1 3 5 8 12 and 13 m).

In addition, water temperature, pH and electrical conductivity were logged hourly in Lake Årungen at 1 m depth with an YSI logger (6920) and V2 sond (YSI incorporated, Ohio, USA).

4.2.2 Light

Water transparency was measured as secchi depth (SD) by using a 13x18cm white secchi disk, and as photosynthetic active radiation (PAR) at different depths by using a Light meter and sensor (Model: Li -250 from Li-Cor, USA). Light measurements were not performed in Lake Årungen on the 8th of May, due to problems with the Light meter.

4.2.3 Discharge measurements

Discharge measurements in Bølstadbekken were carried out at a turbulent sequence of the stream (59°41'181N, 10°45'262), (fig 1 location 3) using the salt dilution method according to (Lundekvam 1982) . The discharge was measured at six different times during the summer and autumn 2013 (20/6, 30/7, 15/8, 4/9, 1/10 and 29/10). Sodium chloride (NaCl) was applied to the stream instantaneously, and the change in conductivity was measured over time. Calibration was carried out in the lab on stream water, by adding a known mass of NaCl into 2 liters of water and constructing a calibration curve. The calibration curve was used to convert conductivity to concentration of NaCl. The discharge (Q) was then measured as:

$$Q = M/A [Ls^{-1}]$$

Where: M is the total mass of applied NaCl and A is the area under the time/concentration curve. The accuracy of the salt dilution method at instantaneously NaCl application is in theory $\pm 2\%$ but the practical accuracy are likely in the interval $\pm 5-20\%$ (Lundekvam 1982).

4.3 Physical, chemical and biological analyses

Filtration, subsampling and conservation for chl_a, water color, DOC, Ion chromatography, inorganic nutrients and phytoplankton analyses were carried out on the day of sampling. The spectrophotometric measurement was carried out using a UV-VIS Spectrophotometer UV-1201 (Shimadzu, Kyoto, Japan).

pH

pH was determined potentiometric, according to Norwegian standard (NS 4720), on non-filtered sample using a pH meter (PHM210 Standard pH meter, Copenhagen). The pH meter was calibrated against 2 standard solutions with pH 4 and pH 7. The precision and accuracy are dependent on the composition of the sample but normally a precision of ± 0.02 and accuracy of ± 0.05 pH are obtainable. The measurements were carried out on the day of sampling.

Alkalinity

The total alkalinity was determined according to Norwegian Standard (NS-EN ISO 99631), by endpoint titration to pH 4,5, by adding Hydrochloric acid (HCL) until pH reaches the equivalent point for H^+ and HCO_3^- (4,5), using a pH meter (PHM210 Standard pH meter, Copenhagen) and Dosimat (665) (Metrohm, Switzerland). The pH meter was calibrated against 2 standard solutions with pH 4 and pH 7 (detection limit: 0.04 meq L^{-1}).

Turbidity

Turbidity was determined according to Norwegian Standard (NS-ISO 7027) on unfiltered sample, with a laboratory turbidimeter model 2100AN IS (Hath company, Loveland, CO, USA), and expressed as formazin nephelometric units (FNU). The turbidity is determined nephelometric, which is a measure of light scattering by suspended particles. The portion of light reflected for a given density of particles, is governed by the particle properties such as shape, color, and reflectivity. The turbidity measurements were carried out on the day of sampling.

Suspended solids

Suspended solids was determined according to Norwegian standard (NS-EN 872). Pre-known volumes of water were filtered through a glass fiber filter (GF/c, Whatman). The filters were then dried at 105°C for 1h to evaporate all water. Total suspended solids were quantified by weighing the mass of the residue retained on the filter and subtracting the weight of the empty filter. Total organic solids were quantified after burning the filters at 480°C for 45min and subtracting the weight of the residue retained after burning, from the weight of the residue on the dried filter. Results are expressed as total suspended solids and organic suspended solids (mg) per liter. The

results should be based on a dry residue of at least 2 mg. The uncertainty of the method lies between 4.29 - 4.72%.

Total organic carbon (TOC) and dissolved organic carbon (DOC)

Total organic carbon (TOC) and dissolved organic carbon (DOC) was analyzed by qualified personnel at the water and earth science laboratory at IMV, using a TOC analyzer (ASI-V and TOC-V CPN, Shimadzu corporation, Japan) (detection limit: 0.2 mg L⁻¹).

Water color and absorbance at 254 nm

The water color determination was carried out according to Norwegian standard (NS 4787) on water samples filtrated through a 0.45µm pore sized membrane filter. The absorbance was measured spectrophotometrically at 410 nm in 50 mm cuvettes. The results are expressed as true color units (TCU) in mg L⁻¹Pt, by comparing absorbance to a platina standard solution (platina cobolt chloride). The absorbance at 254 nm was also measured and expressed as the absorbance at 254 nm. Here 10 mm cuvettes were used.

The water color measurements were carried out on the day of sampling. Oxidation of iron and manganese complexes can interfere with the results.

Nitrate (NO₃-N), chloride (Cl⁻) and Sulfate (SO₄²⁻)

Cl⁻, NO₃-N and SO₄²⁻ were determined by ion chromatography (NS-EN ISO 10304-1) with an electrical conductometer as the detector (XYZ auto sampler, ASX-500 series, Zellweger analytics). The anions get separated in a column with low capacity anion-ionexchange as the stationary phase. The mobile phase, (the eluent) consist of a carbonate/bicarbonate buffer. After passing through a suppressor (cation exchange which reduces the conductivity of the eluent) the peaks are registered with a conductivity meter. Detection limit lies between 0.01 and 0.03 mg/l. The analyses were executed by qualified personnel at the water and soil laboratory (at IMV).

Total phosphorus

Total Phosphorus (TP) was determined according to Norwegian Standard (NS 4725). Samples were oxidized with peroxide sulfate and autoclaved for 30 min, in order to convert all phosphorus into orthophosphate. TP was quantified by a reaction with ascorbic acid and antimony-

molybdate. The yellow antimony-molybdate phosphorus complex in acid condition was reduced by ascorbic acid to form a blue colored complex, which was measured spectrophotometrically at 800 nm. The absorbance of the blue complex is proportional to the concentration of orthophosphate (detection limit: $1 \mu\text{g L}^{-1}$).

Phosphate

Phosphate (PO_4^{3-}) was determined according to Norwegian standard (NS 4724) on water samples filtrated through a $0.45\mu\text{m}$ pore sized membrane filter. Quantification of phosphate is obtained by adding ascorbic acid, and molybdate. The yellow antimony-molybdate phosphorus complex in acid condition was reduced by ascorbic acid to form a blue colored complex, which is measured spectrophotometrically at 800 nm. The absorbance of the blue complex is proportional to the concentration of phosphate (detection limit: $1 \mu\text{g L}^{-1}$).

Total Nitrogen

Total Nitrogen (TN) was determined using Flow Injection Analysis (FIA) (FIAstar 5023 Spectrophotometer with 5023 Detector Controller and 5010 analyzer from Tecator). Samples were added an oxidizing solution, (potassium persulfate) prior to analysis (which becomes alkaline, in neutral solution to form sulfuric acid) and autoclaved for 30 min at 121°C . The digest sample is mixed with a buffer solution and nitrate is reduced to nitrite in a cadmium redactor. On the addition of an acidic sulphanilamide solution, the nitrite will form a diazo compound. The compound is coupled with N-(1-naphtyl)-Ethylene Diamine Dihydrochloride (NED) to form a purple azo dye, which is measured at 540 nm.

Analysis was carried out by qualified personnel at the soil and water science laboratory at IMV.

Ammonium-N

Ammonium-N ($\text{NH}_4\text{-N}$) was determined according to modified Norwegian standard (NS 4746), on samples filtrated through a $0,45\mu\text{m}$ pore sized membrane filter. Ammonium reacts with hypochlorite in alkaline conditions to form mono-chloramine, which in the presence of salicylate forms a blue complex and is measured spectrophotometrically at 655 nm. The detection limit is $0.02 \text{ mg/l NH}_4\text{-N}$. (NS 4746 uses phenol instead of salicylic acid)

Silicon

Silicon (Si) is measured as molybdate reactive silicate according to the method described by Golterman et al.(1978), and was determined by adding molybdate to 2ml sample. After stirring the sample and letting it rest for 10 min, a reducing agent was added (a mix of menthol-sulfite solution, oxalic acid and dilute sulfuric acid). Molybdate reactive Si in acid solution reacted with ammonium-molybdate and formed a Silicon molybdate complex, with by adding menthol is reduced to an intense blue complex. Oxalic acid was added to prevent interference by phosphate. The absorbance was then measured spectrophotometrically after 20 hours at 815 nm (detection limit: 0.1 mg L⁻¹).

Cations

Cations were determined using inductively coupled plasma – optical emission spectroscopy (ICP-OES), an analytical method able to detect many elements simultaneously. Samples were added HNO₃ acid (5%) previous to the analysis. The analysis was carried out by lab personnel at the water and soil laboratory at IMV. Results are given as total concentrations of each element in mg L⁻¹, detection limits for; Na (0.001), Mg (0.0004), Al (0.0010), Si (0.001), P (0.0011), K (0.004), Ca (0.008), Mn (5E-06), Fe (0.0002). Cu concentration is given in µg L⁻¹, detection limit 0.2.

Chlorophyll *a*

Chlorophyll *a* (*chl_a*) was determine according to Norwegian Standard (NS 4767) by filtrating known volumes of water through glass fiber filters (GF/c, Whatman). The filters were dried under dark conditions for 2h and conserved by freezing. *Chl_a* extraction were carried out by adding methanol and letting the filters soak for 20h, then centrifuging the filters for 10 min at 4000 rpm. The absorbance was determined spectrophotometrically in 1 cm cuvettes, at 665 and 750 nm respectively. A specific absorption coefficient was used when calculating the concentration in the methanol extract ($g_{chl-a} = 77 \text{ l g}^{-1} \text{ cm}^{-1}$). *Chl_a* was the calculated according to:

$$C_{chl-a} = (mAbs_{665} - mAbs_{750}) / (g_{chl-a} * l_{cm}) \text{ (g/L)}$$

$$C_{chl-a} * V_{methanol} = V_{water} * C_{chl-a-water}$$

Phytoplankton biomass and composition

Water samples for determination of phytoplankton biomass and composition was retrieved from a 1 meter depth in Lake Østensjøvannet and Lake Årungen and preserved with Lugol's solution.

The biomass count was carried out on an inverted phase contrast microscope (Leitz. Fluovert FS, Leica, Wetzlar, Germany). Analyzes were carried out by Pål Brettum at NMBU

Calculation of euphotic depth

The euphotic depth was estimated as the depth in which irradiance of photosynthetic active radiation (PAR) falls to 1 % of the surface value.

$$100(I_0 - I_z)/I_0$$

Where: I_0 is the irradiance at the surface and I_z is the irradiance at depth z

5 Results

5.1 Weather conditions

The results are based on data collected in the period between 8th of May until 29th of October.

The weather conditions during this period were characterized by high air temperatures and sparse precipitation, in late summer and autumn, compared to the normal period (1961 – 1990) (figure 5 A). Figure 4 A & B shows the average daily temperatures and precipitation values for Ås, in the period of May until October 2013. There was an early peak in daily temperatures in the end of May followed by a period of cooler air temperatures in June. The next temperature peak occurred in the middle of July which was followed by a period of stable period of high temperatures that lasted until the end of the month. August through September were characterized by daily average temperatures around 12 – 15°C.

The precipitation during the months of April to June was on average above the values for the normal period (fig 5 A). July to September were dry months compared to the normal period, while October showed precipitation values close to the normal average of the month (figure 5A). May and June were characterized by quite large variations in daily precipitation values, followed by a long period in July without precipitation (figure 4 B). There was some rainfall in the middle of September while towards the end of the month, the precipitation was sparse.

The discharge in Bølstadbekken, presented in figure 5 B, was 108 L s⁻¹ at the 20th of June. The discharge in July to October was characterized by the dry weather conditions during the summer, while the discharges for the last date (29th of October) was characterized by an increase in precipitation, especially prior to the measuring date, resulting in relatively high discharge (570 L s⁻¹). Dry soil holds back less precipitation, due to overland flow and increased preferential flow, which explains the high increase in discharge. The discharge patterns in Bølstadbekken corresponded well to the daily average discharge in Skuterudbekken (upstream) and Årungen (downstream) in terms of periods with low flow contra periods of high flow (figure 6 A & B).

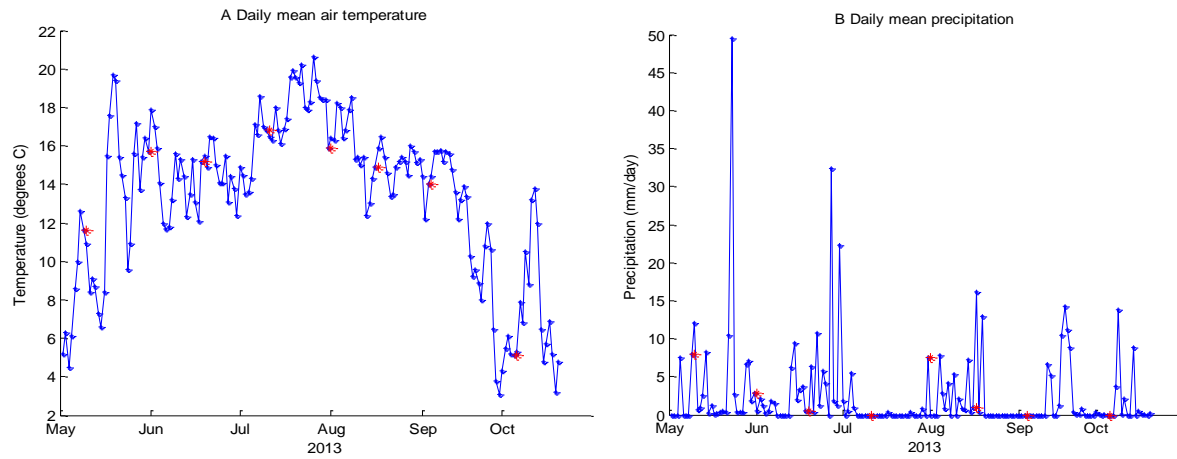


Figure 4 (A) Average daily temperatures ($^{\circ}\text{C}$) and (B) average daily precipitation (mm/d). Red markers represent the conditions for the days of sampling. The data is collected at Sørås meteorological station at Ås (Thue-Hans en & Grimenes 2012)

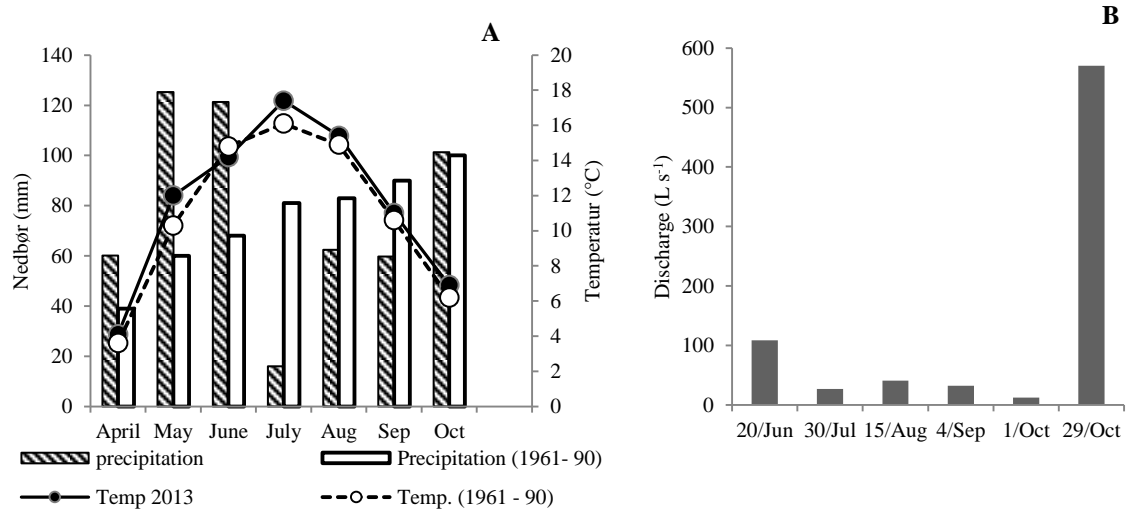


Figure 5 A. Monthly average values of precipitation from April to October 2013 (blue striped bar) compared to the normal values (1961 -1990) (open bars) and monthly average temperatures for the same period (black markers) compared to the normal values (open markers) B) Discharge in Bølstadbekken (L s^{-1}) measured at 6 different occasions between 20th of June until 29th of October.

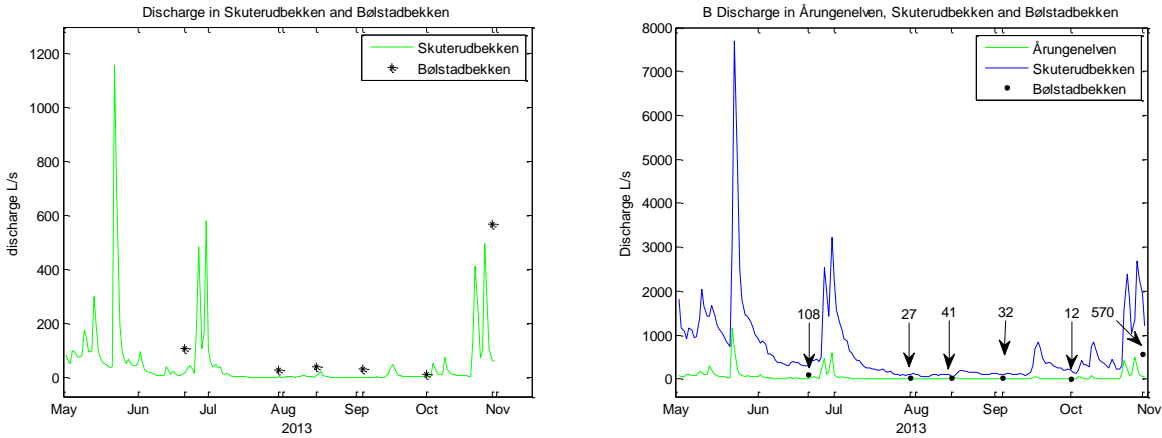


Figure 6 A Discharge in Skuterudbekken and Bølstadbekken and B discharge in Bølstadbekken in relationship to Skuterudbekken (upstream) and Årungenelven (downstream). Discharge data for Skuterudbekken and Årungenelven has been provided by Johannes Deelstra at Bioforsk.

5.2 Optical properties and suspended solids

Figure 7 and 8 presents the secchi depth (A) and suspended solids (B) in the epilimnion of Lake Østensjøvannet and Årungen respectively. A low secchi depth corresponded to high concentrations of total suspended solids in both lakes. In Lake Østensjøvannet the maximum transparency was reached in the beginning of July and in the beginning of September, In Lake Årungen there was gradual improvement of the transparency with time.

In Lake Østensjøvannet wave lengths of PAR was absorbed in the first meter of the water column at an early stage of the season (figure 9). The transparency increased gradually throughout the season (between 2 – 2.5 meters on average). Wavelengths in the blue end of the spectra were absorbed strongly; green light was generally absorbed stronger than red light, which penetrated furthest down in the water column. An exception is registered for the 17th of June when green light penetrated furthest. On the 15th of August and 2nd of September green and red light were nigh equally absorbed in the water column.

In Lake Årungen, total incident light reached a depth between 2 meters in the beginning of the season to over 6 meters in the end of the season (figure 10). Similar to Lake Østensjøvannet the blue end of the spectra was absorbed first, followed by green light. Red light penetrated deepest in the water column; with the exception for the 15th of August, when green and red wave lengths were more equally impaired.

Table 3 show light attenuating parameters in the epilimnion of the two lakes, and table 4 show determining parameters for water color. The Pearson product moment correlation coefficient (R^2) is given as a measure of correlation between the parameters. The regression lines are presented in appendix 7.

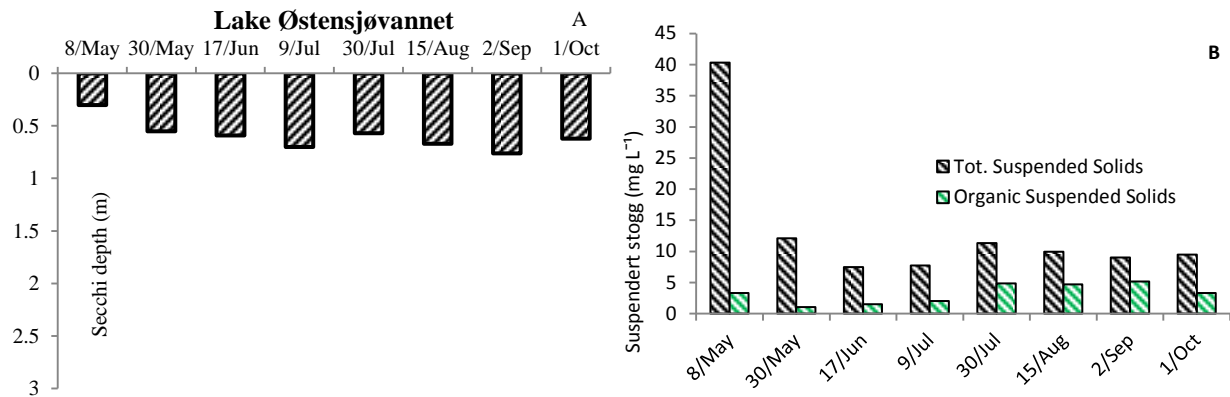


Figure 7 A) Secchi depth and B) suspended solids based on average concentration 1-3 meters in Lake Østensjøvannet, in the period of May until October 2013

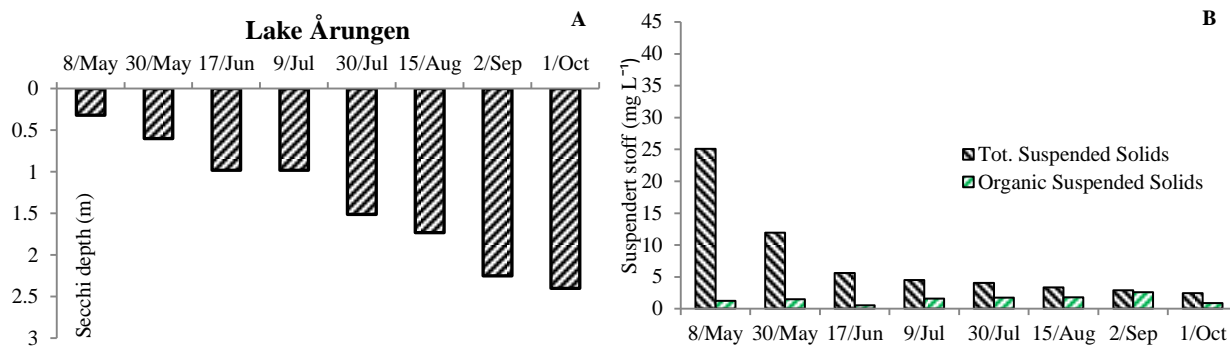


Figure 8 A) Secchi depth and B) suspended solids based on average concentration 1-5 meters in Lake Årungen in the period of May until October 2013.

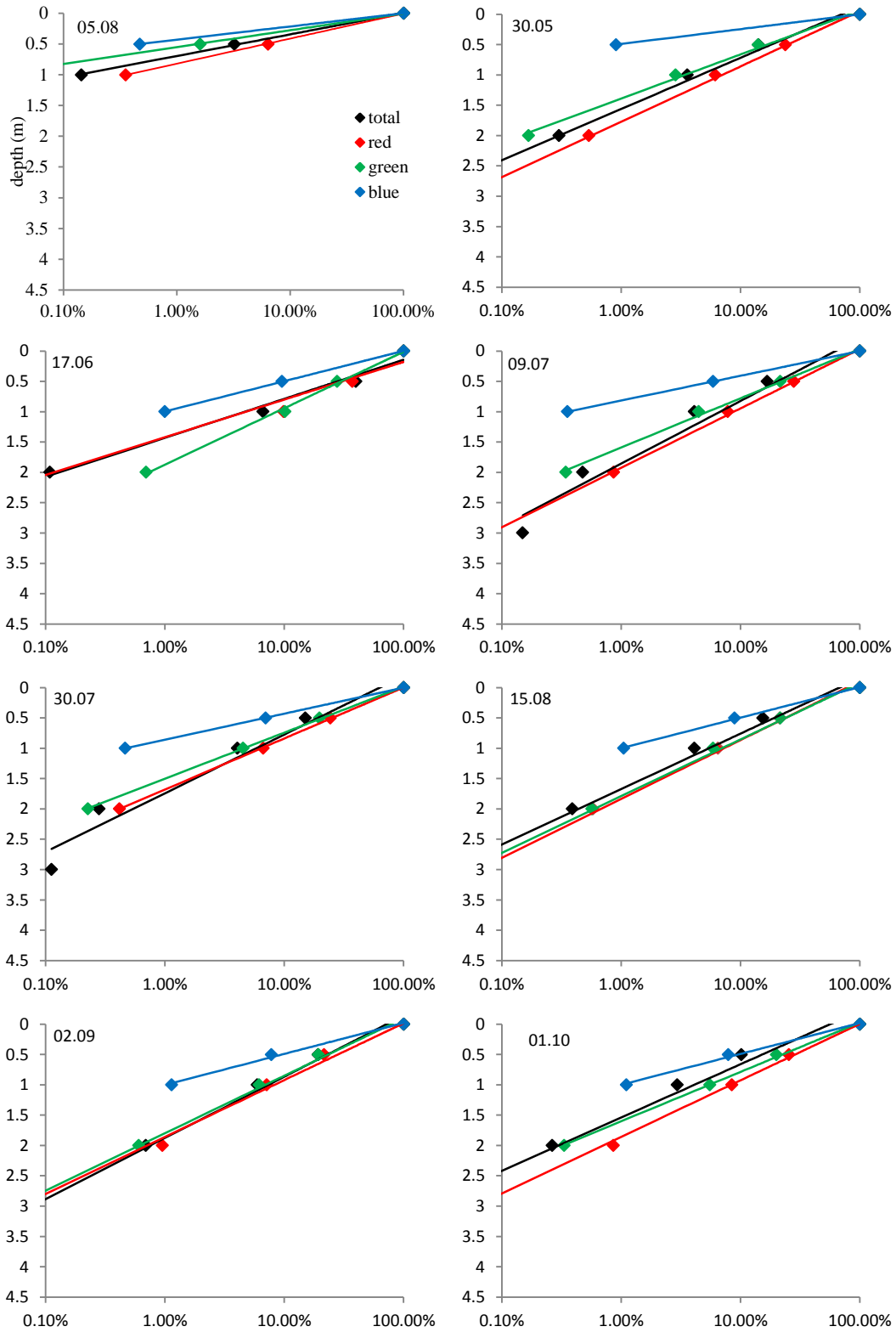


Figure 9 Relative irradiance in Lake Østensjøvannet (x axis), with depth (y axis) for PAR, and red, green and blue light presented by the logarithmic regression line for each series. PAR is represented by black line, the other colors indicates light irradiance in the red, green and blue spectra.

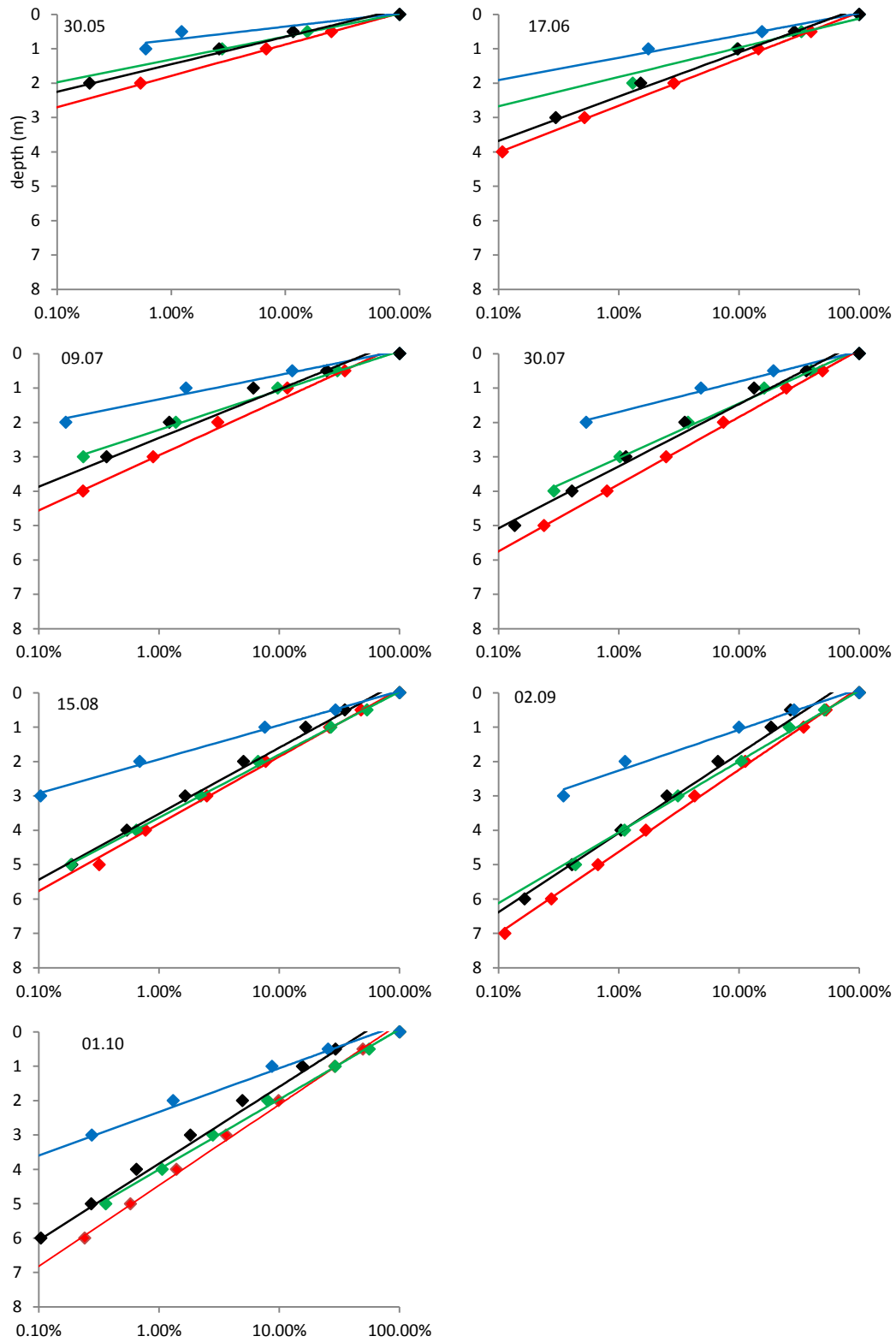


Figure 10 Relative irradiance in Lake Årungen (x axis), with depth (y axis) for PAR, and red, green and blue light presented by the logarithmic regression line for each series. PAR is represented by black line, the other colors indicates light irradiance in the red, green and blue spectra.

Table 3 Light attenuating parameters in the epilimnion of Lake Østensjøvannet and Lake Årungen

	Z _{eu} Østensjøvannet	Z _{eu} Årungen	
Color	0.1	0.91	R ²
Turbidity	0.85	0.63	R ²
Susp. Solids	0.82	0.77	R ²
DOC	0.47	0.43	R ²

Table 4 Parameters determining water color in the surface water (1 m) in Lake Østensjøvannet and Årungen

	Water color		
	Østensjøvannet (1m)	Årungen (1m)	
Abs 254 nm	0.75	0.96	R ²
DOC	0.33	0.37	R ²
Susp. Solids	0.03	0.67	R ²

5.3 Temperature and oxygen variations with depth

5.3.1 Lake Østensjøvannet

At the first date of sampling there was no thermal stratification in Lake Østensjøvannet, and the water column was in full circulation (figure 11). The 30th of May, a weak stratification had developed; the thermocline was distinguished by a temperature shift between 2 and 3 meters depth, followed by a shift in oxygen concentration due to reduced mixing between the epilimnion and hypolimnion. The thermocline on the 17th of June was located between 2 and 4 meters depth and the oxygen level beneath 4 meters depth was less than 0.1 mg L⁻¹. At the 9th of July, the temperature gradient was less distinct as a result of heating of the upper water column (from 17.5 to 19.5°C) and wind mixing of water masses. There was a gradual warming of the hypolimnion, down to a depth of 5 meters throughout the season, while the water at 6 meters is kept at a stable temperature of approximately 10°C. July was characterized by near-surface stratifications and long temperature gradients. In August the temperature gradient were sharper, and oxygen deficits were registered below a depth of 4 meters. In September the surface water temperatures were reduced from its summer maximum and the thermocline was located deeper in the water column.

The 1st of October the water column was in full circulation as seen from the extinguished temperature and oxygen gradients.

5.3.2 Lake Årungen

In Lake Årungen, the water column was initially in full circulation (as seen in figure 12). In June the water column had become stratified, as seen from the shift in temperature and oxygen concentration between 4 and 5 meters depth. Oxygen concentrations lower than 0.1 mg L^{-1} was not occurring over the sediments until the 9th of July. In July the temperature gradient became less steep as a result increased air temperatures and wind distribution of water masses with different temperatures. In the end of July the temperature gradient was sharper and the thermocline was located between 3 and 5 meters. Oxygen levels under 0.1 mg L^{-1} was seen from a depth of 8 meters and the oxygen levels continued to be low under a depth of 8 meters until the end of the sampling period. From the 15th of August the thermocline was located between 5 to 8 meters. Unlike the conditions in Lake Østensjøvannet, the autumn turnover had not yet occurred in Lake Årungen at the last day of sampling and there were still anoxic conditions in the hypolimnion.

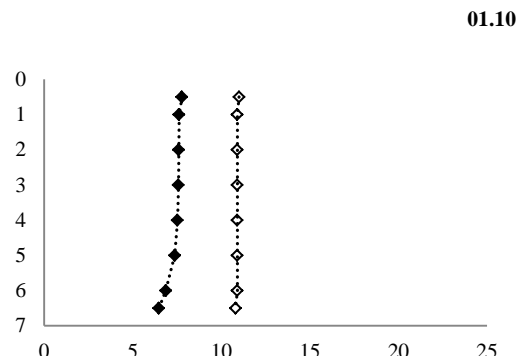
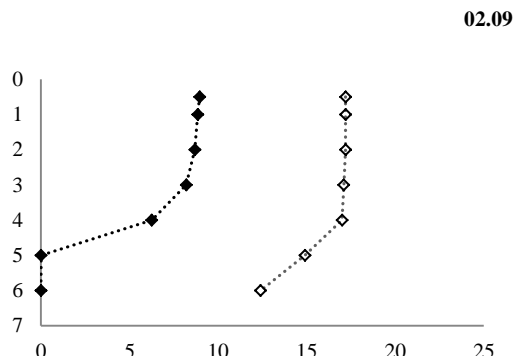
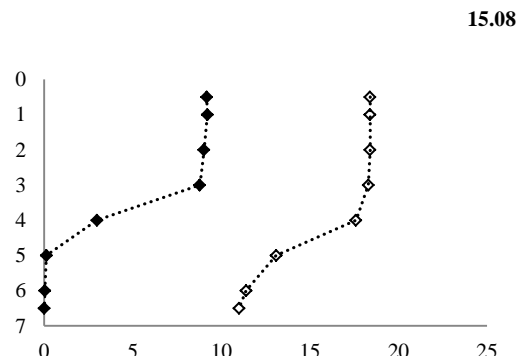
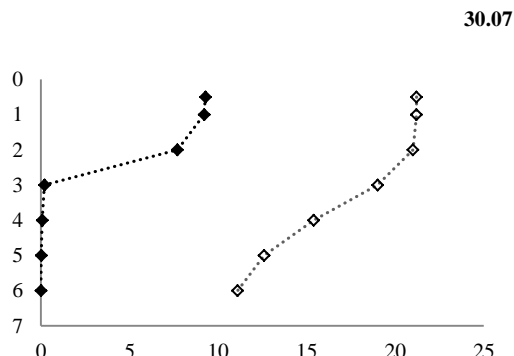
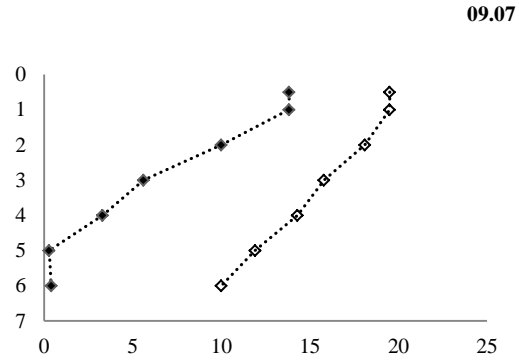
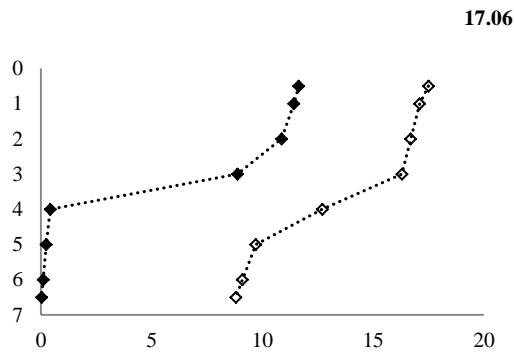
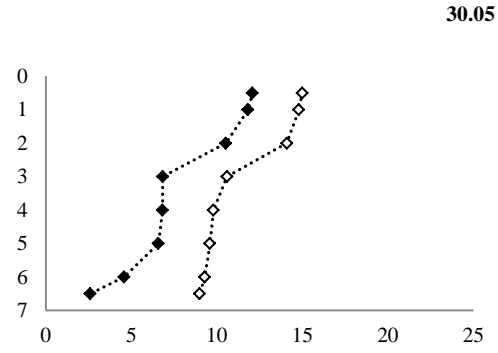
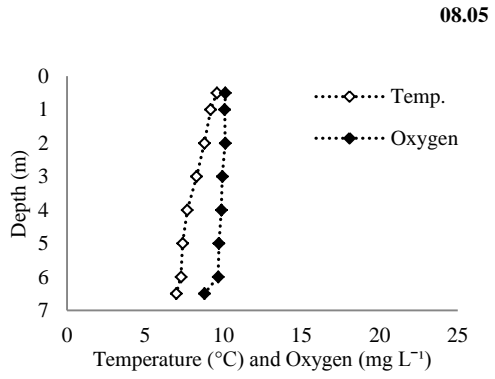


Figure 11 Vertical series of; temperature (°C) (open markers) and dissolved oxygen (mg L⁻¹) (black marker) in Lake Østensjøvannet, between 8th of May until 1st of October 2013.

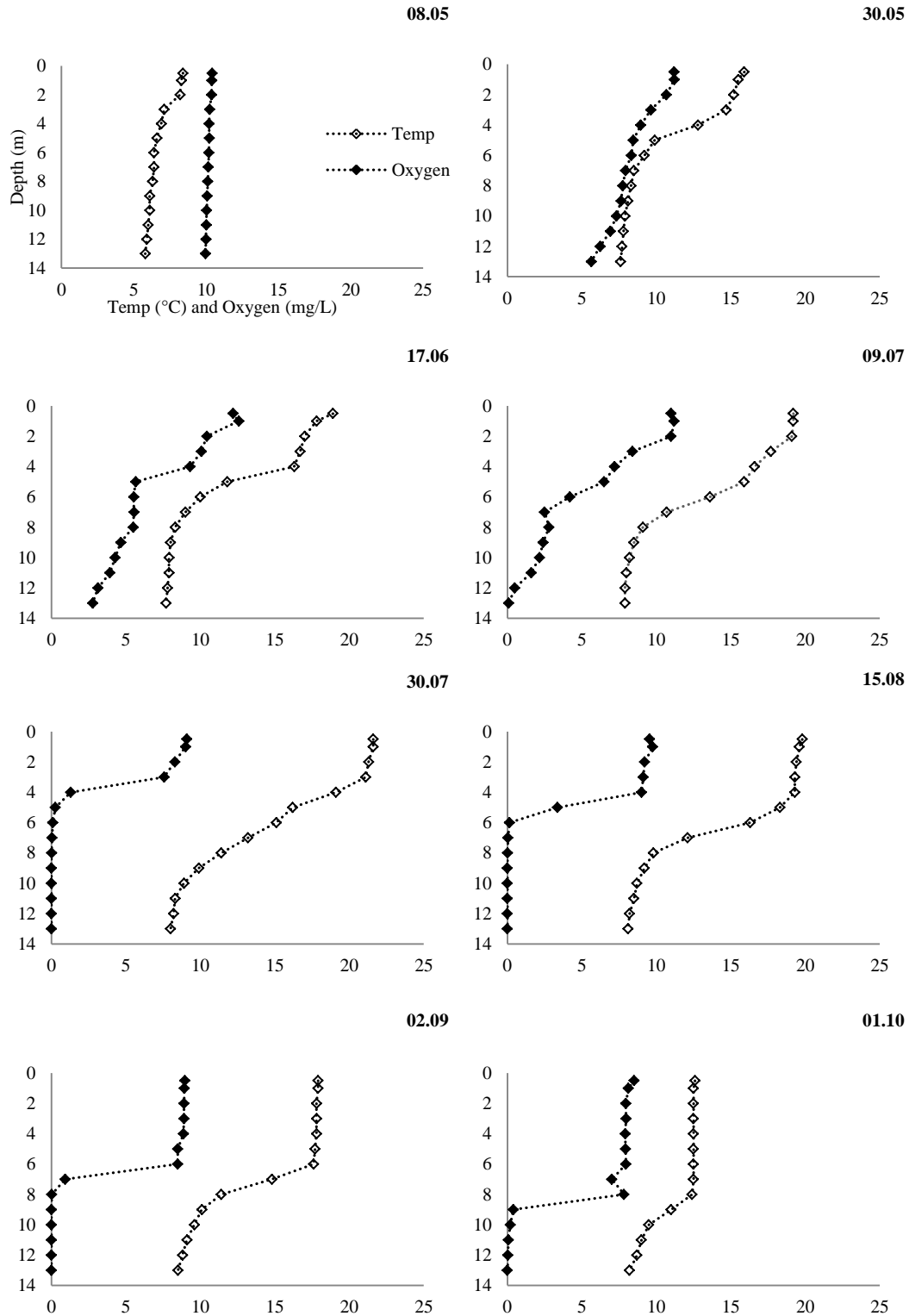


Figure 12 Vertical series of; temperature (°C) (open markers) and dissolved oxygen (mg L⁻¹) (black marker) in Lake Årungen, between 8th of May until 1st of October 2013.

5.4 Isopleth diagrams of water quality parameters in Lake Østensjøvannet

The following diagrams show concentrations isopleths over a time - depth scale. Vertical series with absolute values of *chl_a*, PO₄-P, NO₃-N and SO₄²⁻ is given in appendix 2, 3, 4 and 5.

5.4.1 Temperature, oxygen saturation and pH

Temperature and oxygen saturation (%) over time is presented in figure 13. The maximum temperature in the surface water occurred in the end of July (21°C) and the highest saturation occurred in the beginning of July at 140%, indicating a period of high primary production. pH is another parameter that indicate phytoplankton growth in the lake, in figure 14, pH values was elevated simultaneously as the maximum saturation was reached. pH also varied with depth in the same manner as temperature and oxygen saturation.

5.4.2 Chl_a and nutrients

Variations of *chl_a* and silicon concentrations are displayed in figure 15. The *chl_a* concentration peaked in the 30th of May and 15th of August (> 60 µg L⁻¹), and showed large spatial and temporal variations in the water column. The highest concentrations were found in the epilimnion, but elevated values were also found down to 6 meters depth (28 µg L⁻¹ at the most). Silicon was reduced in the epilimnion in June (2 mg L⁻¹) and increased progressively throughout the remaining part of the season. The concentration was subsequently increased in the hypolimnion.

Initial TP concentration after spring turnover was between 140 and 150 µg L⁻¹ in the water column and reduced to between 74 and 94 µg L⁻¹, after autumn turnover.

Total phosphorus and PO₄-P (figure 16) generally followed the same spatial and temporal pattern, although TP showed more variations in the epilimnion during the season than PO₄-P, which was mainly kept at a concentration of < 2 µg L⁻¹ between the month from June to October. PO₄-P was reduced all the way down in the water column between the month of June until July, and increased in the hypolimnion between the months of July to September. The relative amount of PO₄-P is given in percent in figure 17B. Iron concentration at 1 and 6 meters depth is given in figure 17 A.

Total nitrogen and nitrate (figure 18 A and B), displayed a similar spatial and temporal pattern during the season. The vertical concentration distribution was homogenous and there was a temporal concentration reduction during the season. The concentration peak of both parameters occurred in the 30th of May with values higher than 7 $\mu\text{g L}^{-1}$.

Ammonium (figure 20 A) exhibited only small variations in the water column initially. Elevated concentration in the hypolimnion occurred in the middle of June and became gradually increased towards the end of the season. The concentration peak in the deeper water was registered at the 2nd of September (2.3 mg L^{-1}), while concentrations in the epilimnion remained low. The $\text{NO}_3\text{-N/TN}$ relationship (fig 19), were rather uniform at 1 and 6 meters initially. Towards the end of the season nitrate was reduced in proportion to TN, especially at 6 meters depth (11 %).

Particulate organic nitrogen (PON) concentration (fig 20 B) was higher in the epilimnion, especially in the surface water in the end of May and in July. Reduced concentration in the hypolimnion, was seen at the 2nd of September.

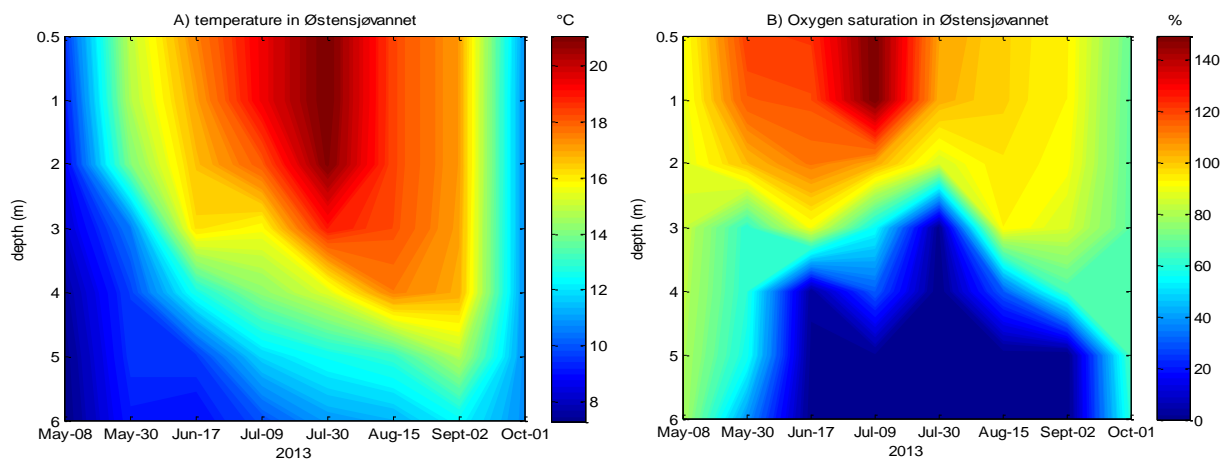


Figure 13 Time – depth Isopleth diagrams with A) temperature and B) oxygen saturation, in Lake Østensjøvannet from 8th of May until 1st of Oct 2013. Temperature is displayed in °C and oxygen saturation in %.

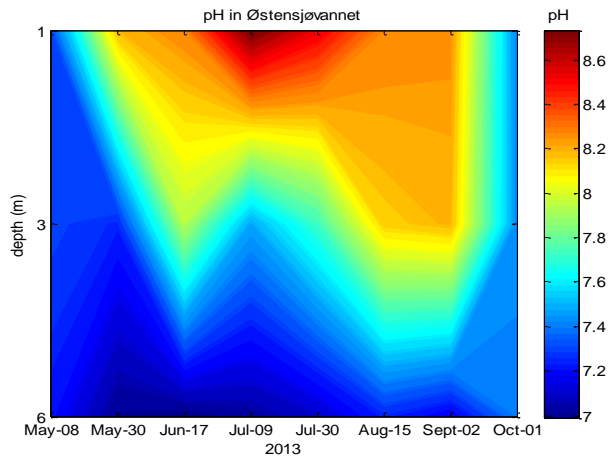


Figure 14 Time – depth isopleth diagram of pH in Lake Østensjøvannet from 8th of May until 1st of Oct 2013.

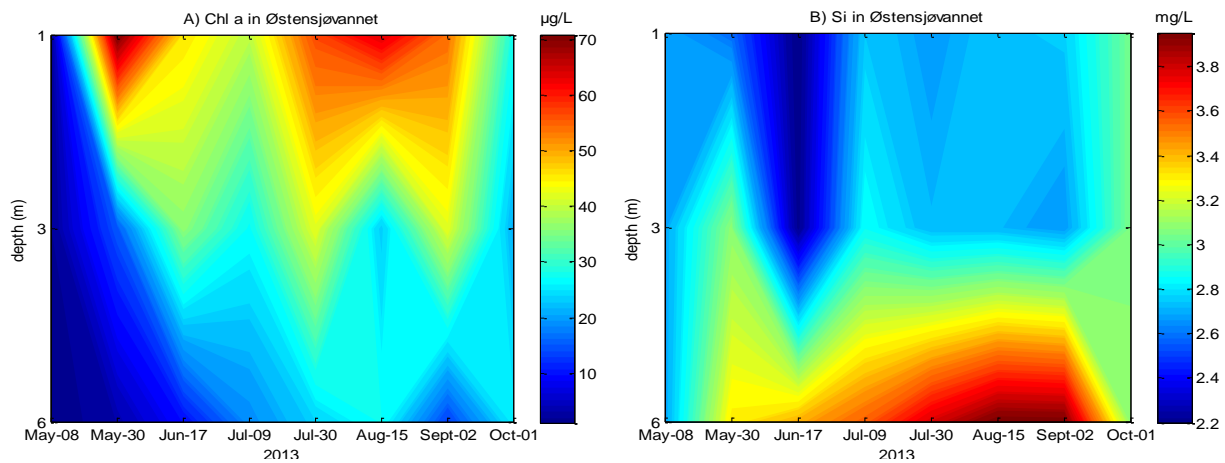


Figure 15 Time – depth Isopleth diagrams with A) chlorophyll a (chl_a) ($\mu\text{g L}^{-1}$) and B) silicon (Si) (mg L^{-1}), in Lake Østensjøvannet from 8th of May until 1st of Oct 2013.

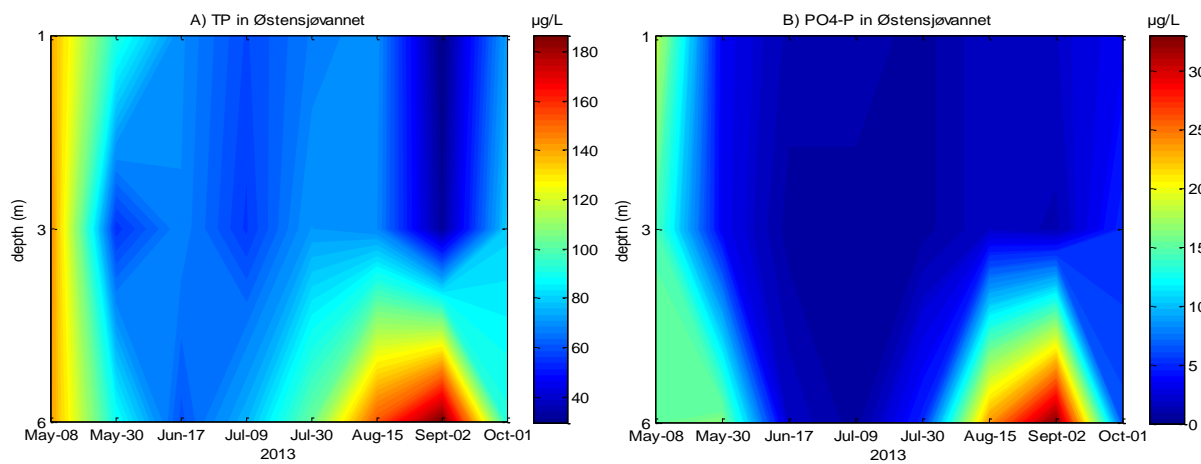


Figure 16 Isopleth diagrams with A) total phosphorus (TP) ($\mu\text{g L}^{-1}$) and B) phosphate ($\text{PO}_4\text{-P}$) ($\mu\text{g L}^{-1}$), in Lake Østensjøvannet from 8th of May until 1st of Oct 2013.

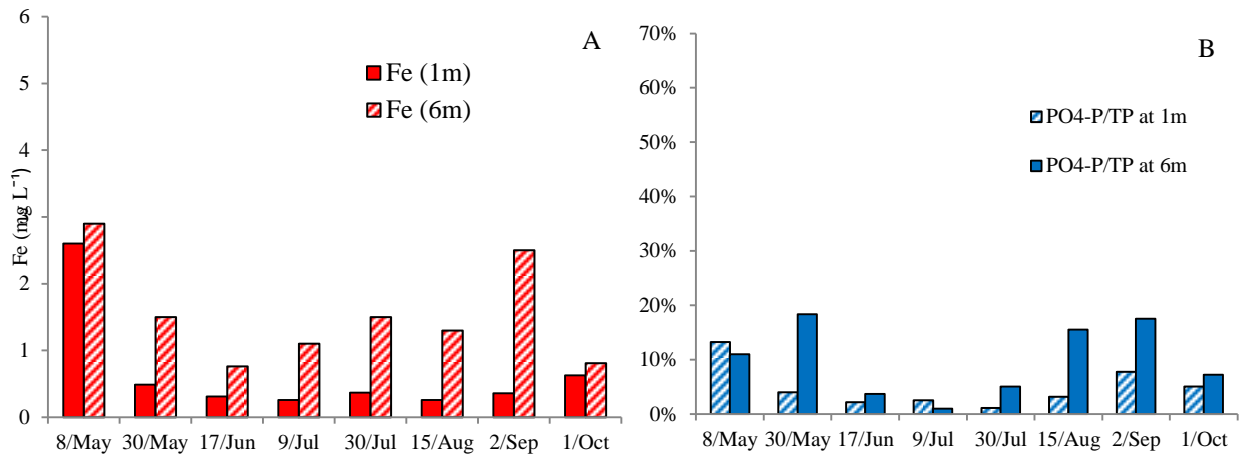


Figure 17 A) Iron (Fe) concentration at 1 and 6 meters depth and, B) Percentage of PO4-P/TP at 1 and 6 meters depth in Lake Østensjøvannet from 8th of May until 1st of Oct 2013.

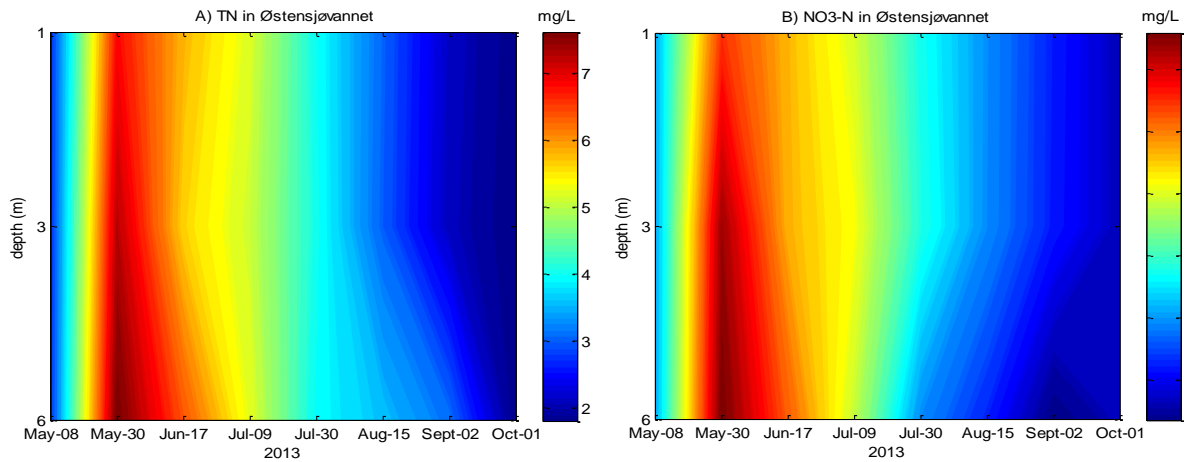


Figure 18 Time –depth isopleth diagrams with A) total nitrogen (TN) (mg L⁻¹) and B) nitrate (NO₃-N) (mg L⁻¹), in Lake Østensjøvannet from 8th of May until 1st of Oct 2013.

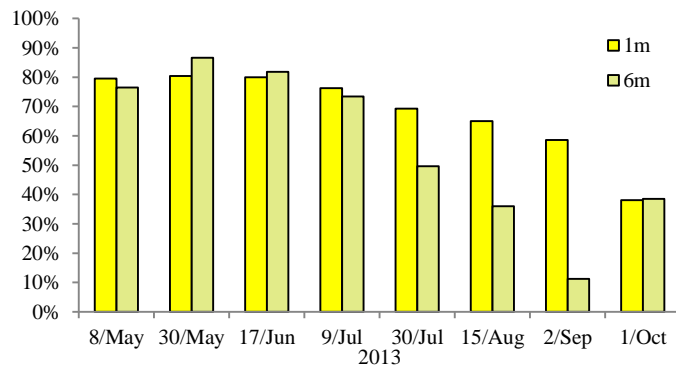


Figure 19 NO₃-N/ TN (%) in Lake Østensjøvannet at 1 and 6 meters depth, in the period between 8th of May until 1st of Oct 2013.

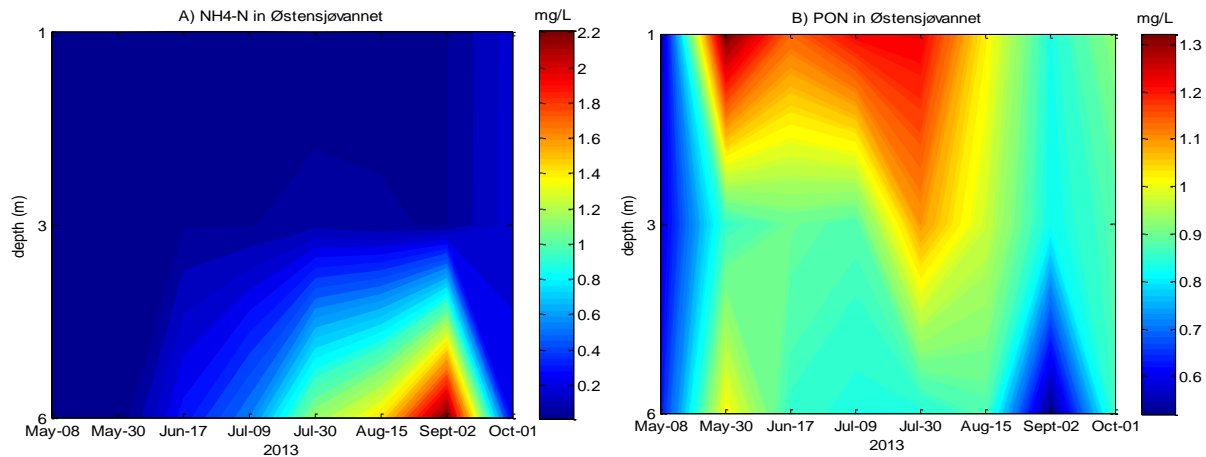


Figure 20 Time – depth isopleth diagrams with A) ammonium (NH₄-N) (mg L⁻¹) and B) particulate organic nitrogen (PON) in mg L⁻¹, in Lake Østensjøvannet from 8th of May until 1st of Oct 2013.

5.4.3 Total organic carbon (TOC), dissolved organic carbon (DOC) and particulate organic carbon (POC)

The concentration distribution of TOC and DOC were similar over the depth and time scale (figure 21). There was a concentration increase over the season and elevated concentrations were registered in the hypolimnion in late season. The concentration of TOC ranged from 6 to 8.5 mg L⁻¹ and DOC ranged from 5.7 to 8.4 mg L⁻¹.

The water color (figure 22B) was more variable with depth than TOC and DOC, and was reduced in the upper water column towards the end of the season while TOC and DOC increased in the entire water column. The peak at 6m depth for TOC and DOC in the 2nd of September was however followed by a similar peak in water color; reducing conditions in the hypolimnion could have interfered with the color results, when iron is oxidized during the filtration. Particulate organic carbon (PON) in figure 22 A, expressed as the difference between TOC and DOC was increased in the water column in August, and reduced over the sediments at the 2nd of September.

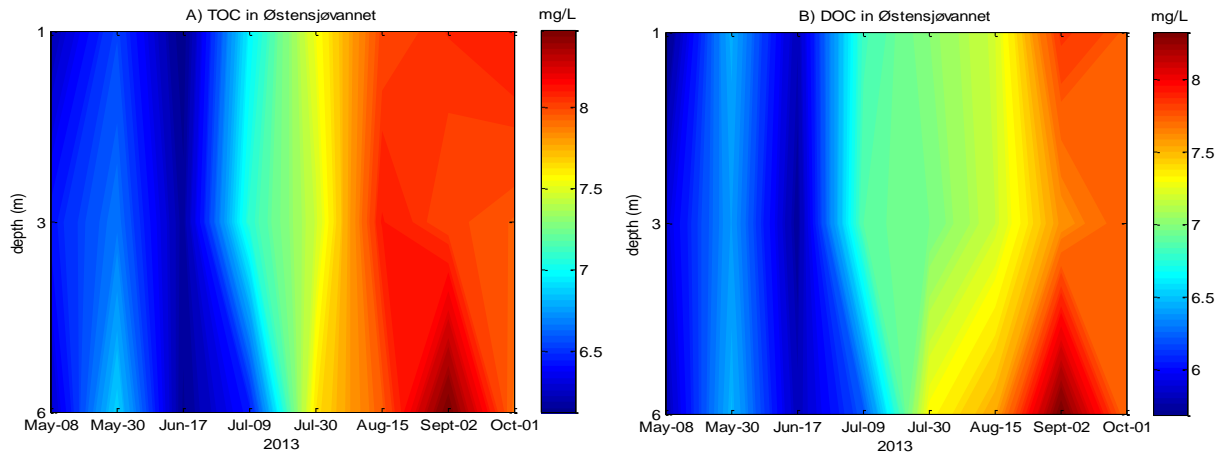


Figure 21 Time – depth isopleth diagrams with A) total organic carbon (TOC) (mg L^{-1}) and B) dissolved organic carbon (DOC) (mg L^{-1}), in Lake Østensjøvannet from 8th of May until 1st of Oct 2013.

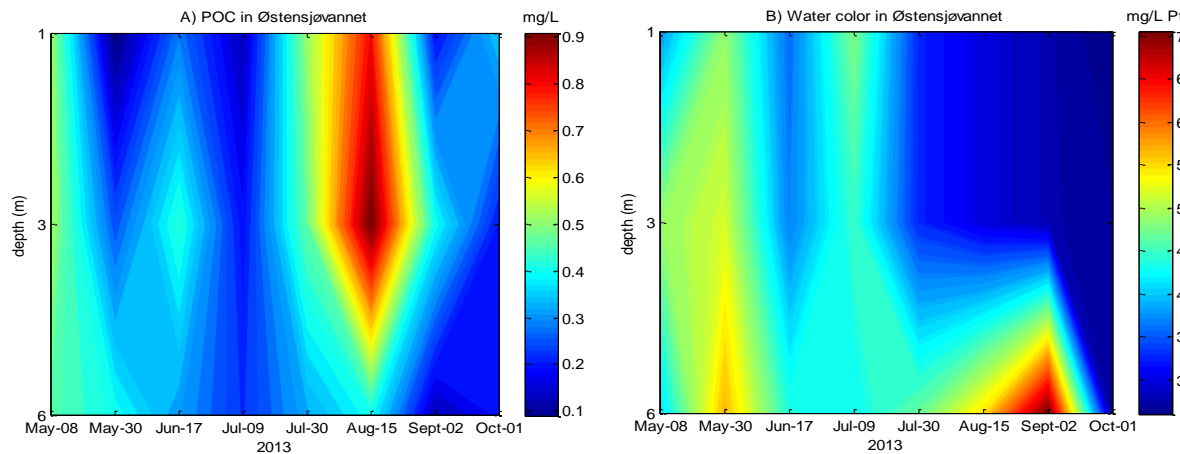


Figure 22 Time depth isopleth diagrams with A) particulate organic carbon (POC) (mg L^{-1}) and B) water color (mg L^{-1} Pt), in Lake Østensjøvannet from 8th of May until 1st of Oct 2013.

5.4.4 Other water quality parameters

Sulfate and chloride figure 23, are rather conservative ions in biological terms, sulfate is however sensitive to changes in redox conditions. Sulfate (fig 23 A) accumulated in the epilimnion towards the later part of the season, due to catchment inputs and reduced mixing. The concentration in the hypolimnion was reduced in August and September from concentration of 12 mg L^{-1} down to 9 mg L^{-1} . Chloride concentration (figure 23 B) was generally decreased from the beginning of the period until the last sampling date, with 38 to 25 mg L^{-1} respectively. Minor spatial variations occurred towards the end with decreasing concentration in the upper water column.

The conductivity (figure 24 A), showed a general increase during the season while turbidity (figure 24 B) decreased with time. Both parameters became elevated at 6m depth on the 2nd of September. Alkalinity (figure 25) was gradually increased in the entire water column. The peak concentration of 1.6 meq L⁻¹ occurred over the sediments at the 2nd of September.

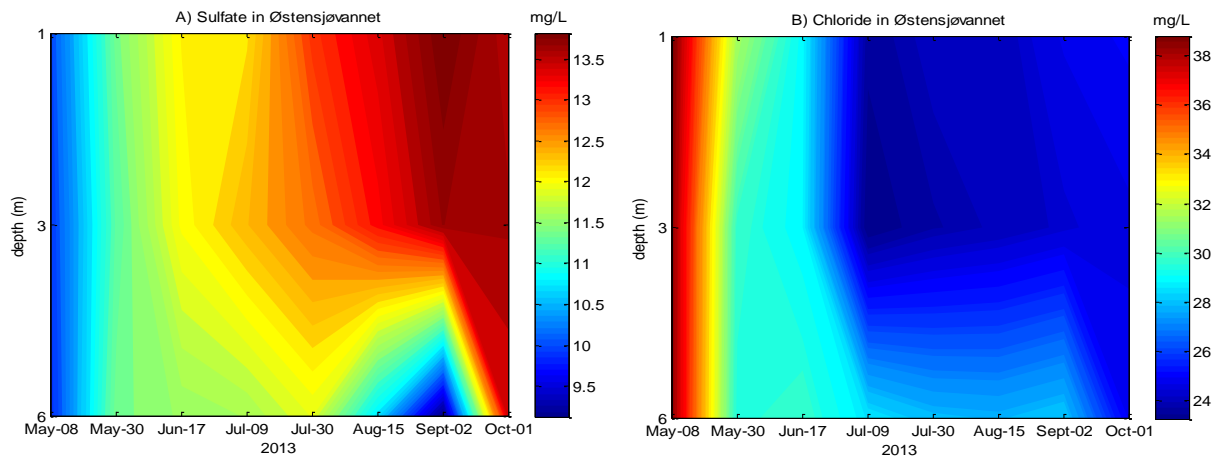


Figure 23 Time – depth isopleth diagrams with A) sulfate (SO_4^{2-}) and B) chloride (mg L^{-1}), in Lake Østensjøvannet from 8th of May until 1st of Oct 2013.

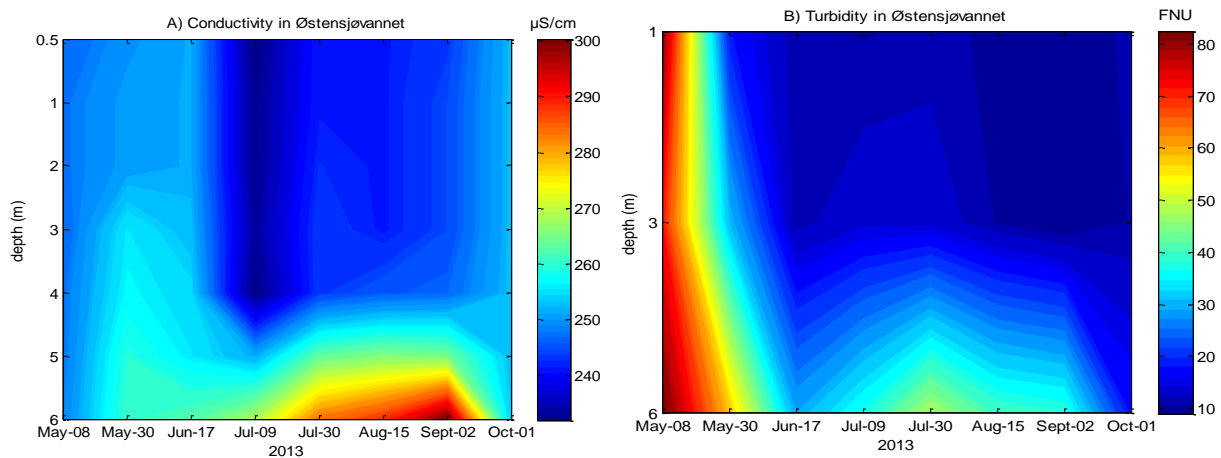


Figure 24 Time – depth isopleth diagrams with A) conductivity ($\mu\text{S cm}^{-1}$) and B) turbidity (FNU), In Lake Østensjøvannet from 8th of May until 1st of Oct 2013.

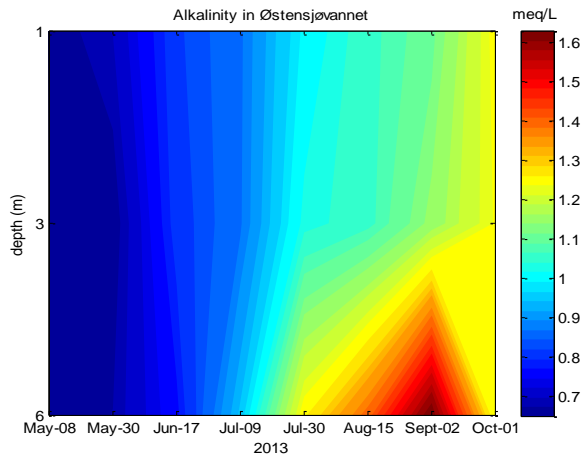


Figure 25 Time – depth isopleth diagrams with alkalinity (meq L^{-1}), in Lake Østensjøvannet from 8th of May until 1st of Oct 2013.

5.5 Isopleth diagrams of water quality parameters in Lake Årungen

The following diagrams show concentrations isopleths over a time - depth scale. Vertical series with absolute values of *chl**a*, $\text{PO}_4\text{-P}$, $\text{NO}_3\text{-N}$ and SO_4^{2-} is given in appendix 2, 3, 4 and 5.

5.5.1 Temperature, oxygen saturation and pH

In Lake Årungen the highest water temperature was obtained in the surface water in the end of July and the maximal oxygen saturation occurred in the middle of June (figure 26). The temperature in the hypolimnion varied between 6 to 8 C^o, under a depth of 6 to 8 meters. From the 9th of July, oxygen saturation was >0.1 % under a depth, varying from 12 to 4 meters, throughout the season. In the end of May until the 15th of August the saturation was higher than 100 % in the depth of 1 to 3 meters.

pH values (figure 27) had a similar pattern as oxygen saturation although the highest pH was registered in the 15th of August. The pH was unchanged over the sediments from 30th of May until 2nd of September and decreased at the last sampling date.

5.5.2 Chl*a* and nutrients

Figure 28 shows variations of *chl**a* and silicon concentrations, during the growth season. *Chl**a* exhibited a large variation in the epilimnion and low concentration was seen in the hypolimnion (< 5 $\mu\text{g L}^{-1}$). The peaks in *chl**a* concentrations occurred in the 17th of June and 30th of July with

concentrations $> 40 \mu\text{g L}^{-1}$. Silicon became reduced below 1 mg L^{-1} in the epilimnion towards the end of the period and also showed a smaller concentration reduction, in the 17th of July. The concentration in the lower parts of the column gradually increased to a maximum of $3,5 \text{ mg L}^{-1}$ at the last date of sampling (1st of Oct).

Due to large variations in total phosphorus and $\text{PO}_4\text{-P}$ in Lake Årungen the isopleths are shown as the natural logarithm of the concentration values in order to see the variations in the water column (figure 29). Total phosphorus was initially at $120 \mu\text{g L}^{-1}$, and was gradually reduced in the water column until the end of July when the concentration was increased in the deeper part. The highest concentration of TP was registered at the last sampling date, at 13 m depth ($680 \mu\text{g L}^{-1}$).

$\text{PO}_4\text{-P}$ was initially between $26 \mu\text{g L}^{-1}$ and $30 \mu\text{g L}^{-1}$ at the surface and bottom depths, respectively and became reduced in the epilimnion ($< 2 \mu\text{g L}^{-1}$), later on in the season, and kept at $30 - 32 \mu\text{g L}^{-1}$ in the hypolimnion. From the end of July concentrations became elevated over the sediments, peaking at a concentration of $430 \mu\text{g L}^{-1}$ at the last day of sampling. The increase in TP were mostly due to the increase of $\text{PO}_4\text{-P}$ over the sediments, as seen in figure 30 B. Iron concentration was increased at 13m depth and reduced in the surface water as seen in figure 30A. Total nitrogen and nitrate (figure 31) varied in a similar manner throughout the season. Both TN and $\text{NO}_3\text{-N}$ were homogenous at the first sampling date and gradually reduced both spatial and temporal. The concentration peak occurred in the epilimnion at the 30th of May for both parameters (4.6 and 3.7 mg L^{-1} respectively). TN was to a large extent composed by nitrate in the start of the season at both 1 and 13 meters depth as seen in figure 32. The nitrate fraction was reduced towards the end of the season at 13 meters depth (6 %).

Ammonium concentration (figure 33 A) was homogenous both spatial and temporal until the end of July, when elevated concentrations was registered in the hypolimnion. The increase in the hypolimnetic water was initially low until the 2nd of September. The 1st of October the ammonia concentration became substantially higher from a depth between 12 and 13 meters (1.4 mg L^{-1}). Particulate organic nitrogen (PON) expressed as $\text{TN} - (\text{NO}_3\text{-N} + \text{NH}_4\text{-N})$ was higher in the surface water, especially in June 17th and July 30th which correspond to the concentrations peaks of *chl a*. Concentrations were relatively low in the hypolimnion with a small increase in October 1st (figure 33 B).

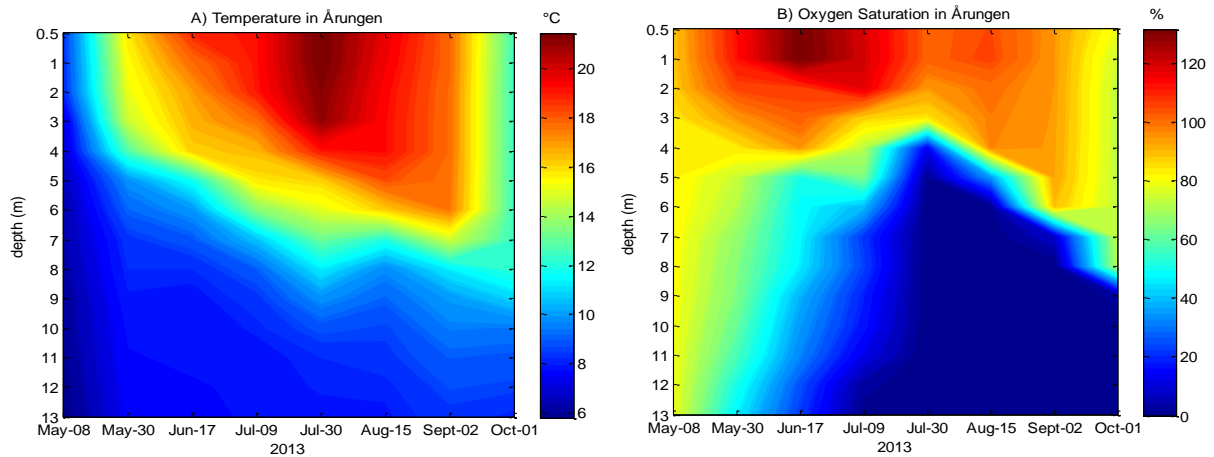


Figure 26 Time – depth isopleth diagrams with A) temperature and B) oxygen saturation, in Lake Årungen from 8th of May until 1st of Oct 2013. Temperature is displayed in °C and oxygen saturation in %

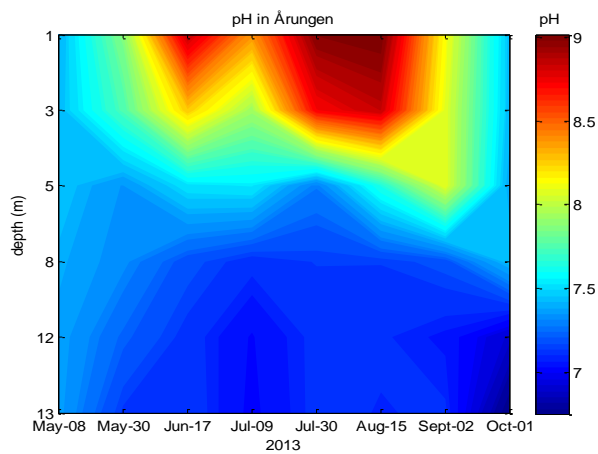


Figure 27 Time – depth isopleth diagram with pH, in Lake Årungen from 8th of May until 1st of Oct 2013.

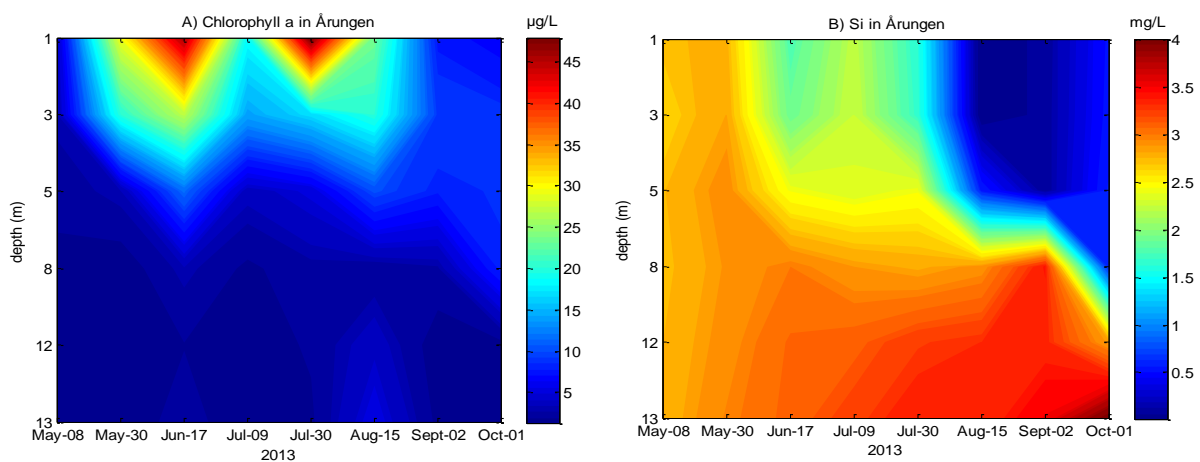


Figure 28 Time – depth isopleth diagrams of A) chlorophyll *a* (*chl a*) ($\mu\text{g L}^{-1}$) and B) silicon (Si) (mg L^{-1}), in Lake Årungen from 8th of May until 1st of Oct 2013.

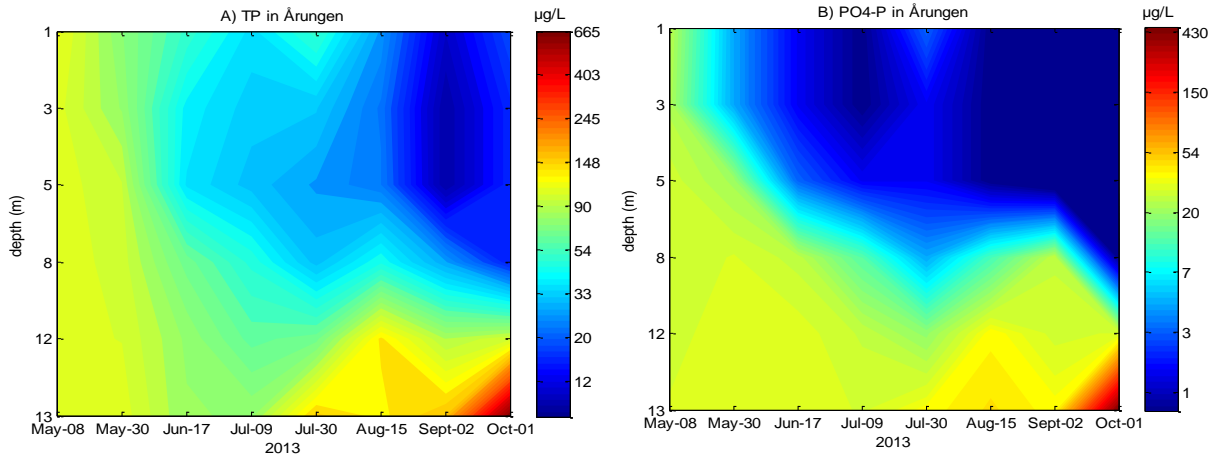


Figure 29 Time – depth isopleth diagrams of A) total phosphorus (TP) ($\mu\text{g L}^{-1}$) and B) phosphate ($\text{PO}_4\text{-P}$), in $\mu\text{g L}^{-1}$ presented as the natural logarithm, in Lake Årungen from 8th of May until 1st of Oct 2013.

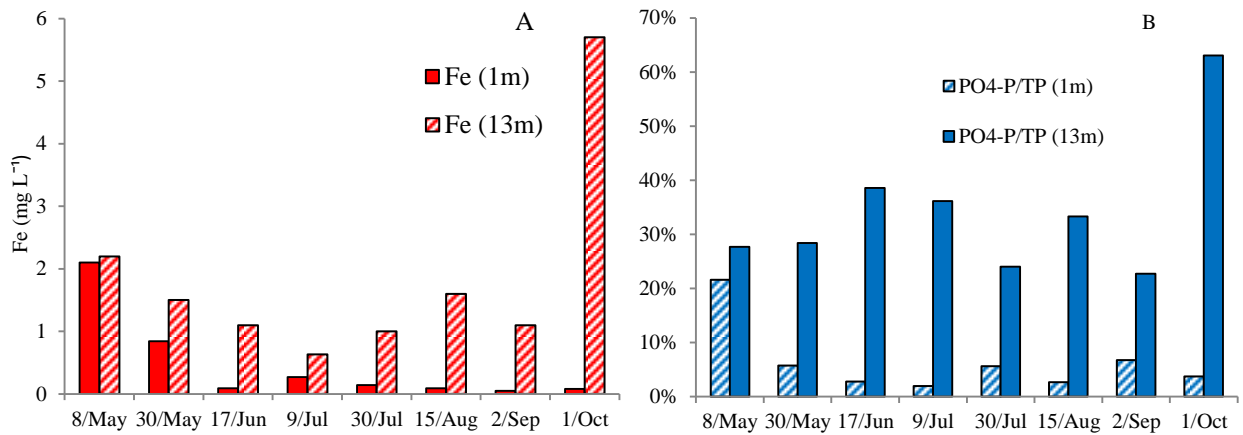


Figure 30 A) Iron concentration (mg L^{-1}) at 1 and 13 m depth in Lake Årungen and B) percentage of PO_4/TP , from 8th of May until 1st of Oct 2013.

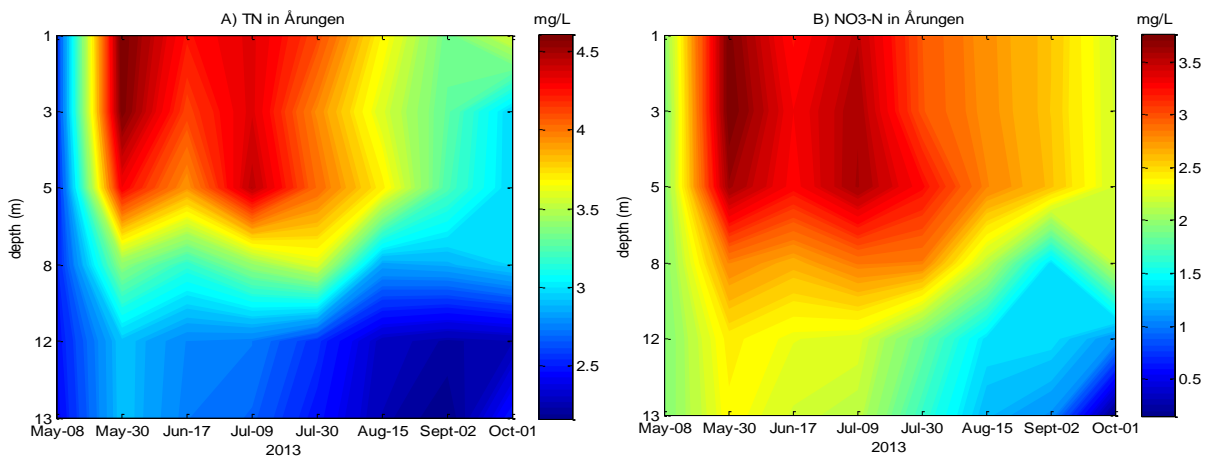


Figure 31 Time – depth isopleth diagrams of A) total nitrogen (mg L^{-1}) and B) nitrate (mg L^{-1}) In Lake Årungen from 8th of May until 1st of Oct 2013.

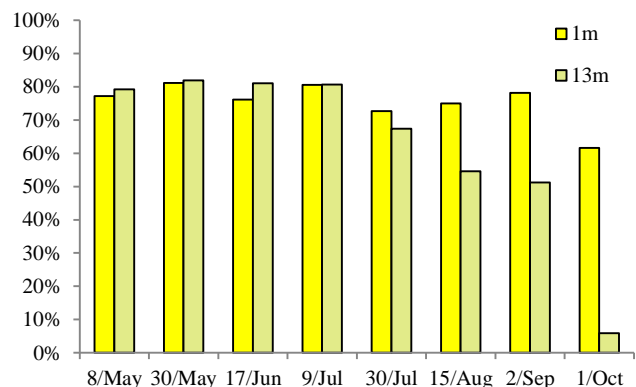


Figure 32 Percentage of NO₃-N/TN in Lake Årungen at 1 and 13 meters depth respectively, from 8th of May until 1st of Oct 2013.

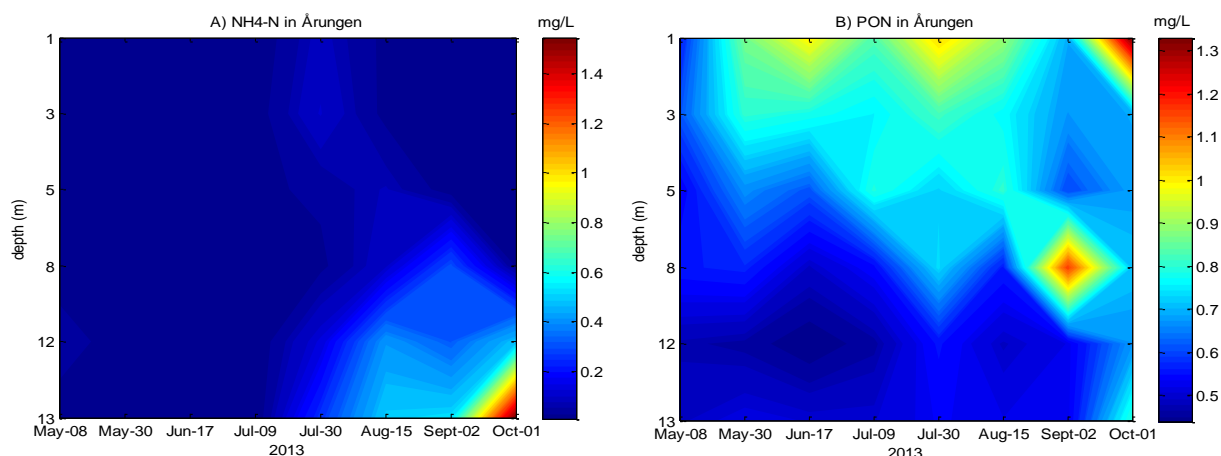


Figure 33 Time – depth isopleth diagrams of A) ammonium (NH₄-N) (mg L⁻¹) and B) particulate organic nitrogen (PON) (mg L⁻¹), in Lake Årungen from 8th of May until 1st of Oct 2013.

5.5.3 Total organic carbon (TOC), dissolved organic carbon (DOC) and particulate organic carbon (POC)

Total organic carbon and dissolved organic carbon (figure 34) exhibited a similar pattern with depth and time, with a concentration accumulation in the upper part of the water column. There was an increase in concentration over the sediments at the 1st of October. The concentrations of TOC ranged from 6 to 7.6 mg L⁻¹ and the concentration of DOC ranged from < 5.8 to 7.4 mg L⁻¹. Particulate organic carbon (figure 35A) is expressed as TOC-DOC, and was higher at an initial stage of the season, and progressively reduced towards the end of the season. The negative values of PON, is most likely a result of instrumental precision error, considering the detection

limit of 0.1 mg L^{-1} . Water color (figure 35 B) was reduced in the epilimnion towards the later part of the season and also increased over the sediments at the 1st of October. The color units ranged from < 30 to $120 \text{ mg L}^{-1}\text{Pt}$ at the most.

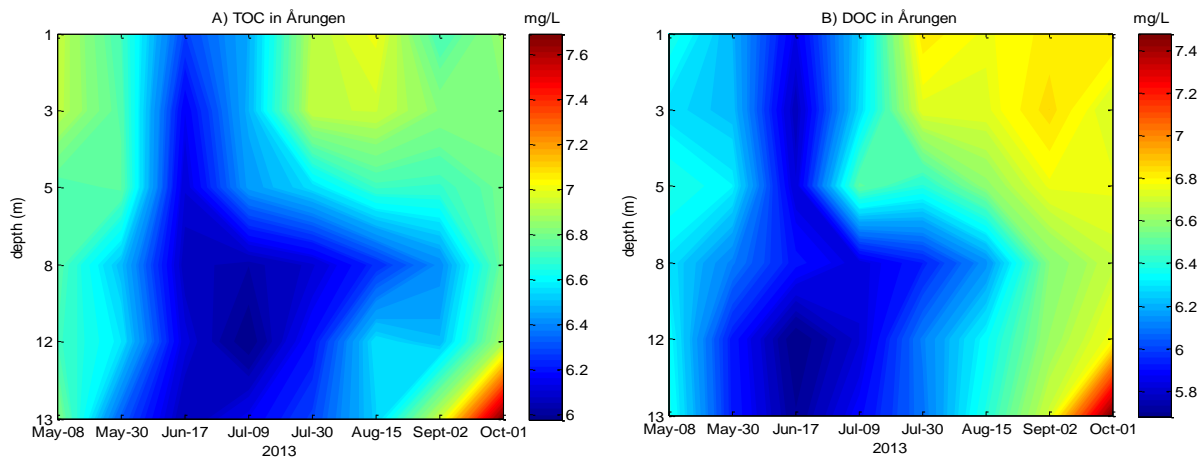


Figure 34 Time – depth isopleth diagrams of (A) total organic carbon (TOC) (mg L^{-1}) and B) dissolved organic carbon (DOC)(mg L^{-1}), in Lake Årungen from 8th of May until 1st of Oct 2013.

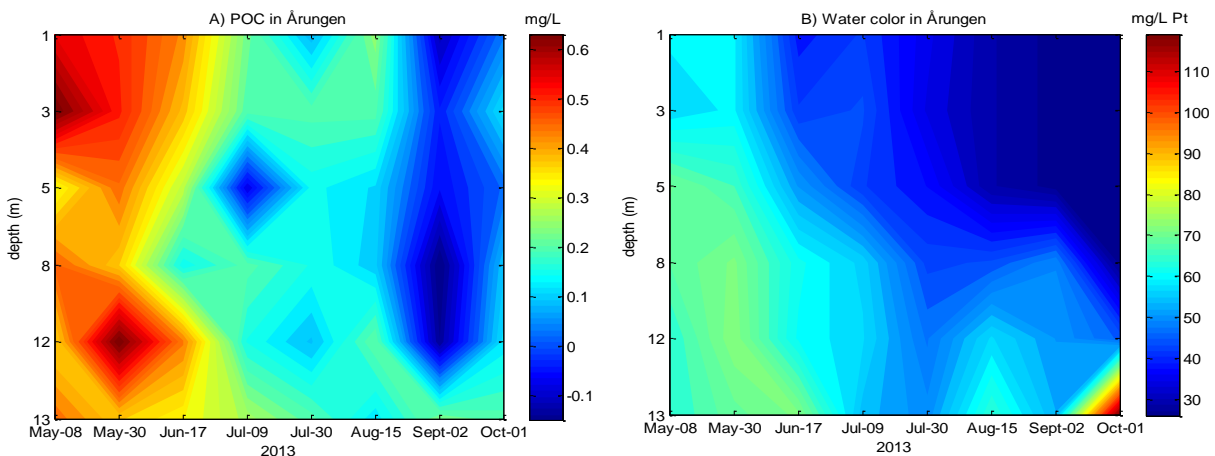


Figure 35 Time – depth isopleth diagrams of A) particulate organic carbon (PON) (mg L^{-1}) and B) water color ($\text{mg L}^{-1} \text{Pt}$) in Lake Årungen from 8th of May until 1st of Oct 2013.

5.5.4 Other in-lake parameters

Sulfur concentration (figure 36 A) was gradually increased in the upper part of the water column from 11 mg L^{-1} to 13 mg L^{-1} in the end of the season. Concentrations in the hypolimnion showed little variation until the end of July when the concentration was reduced. At the last date of sampling (Oct 1st) the concentration at 13 m depth were reduced to 7.5 mg L^{-1} .

The chloride concentration (figure 36 B) showed frequent variations in the entire column over the season. Generally there was a higher concentration in the lower part of the column compared to the upper part.

Conductivity (figure 37 A) was gradually increased with time and show only minor spatial variations over the water column, with the exception for the 1st Oct, when the values were substantially

elevated at the depth of 13m. The concentration range from $< 250 \mu\text{S cm}^{-1}$ to $320 \mu\text{S cm}^{-1}$.

Turbidity (figure 37 B) was higher in the initial stage and gradually reduced over the entire column with an increase over the sediments in September. Alkalinity (figure 38) ranges from 0.8 to 1.4 meq L^{-1} . The highest values were registered at 13 meters depth at the 1st of October.

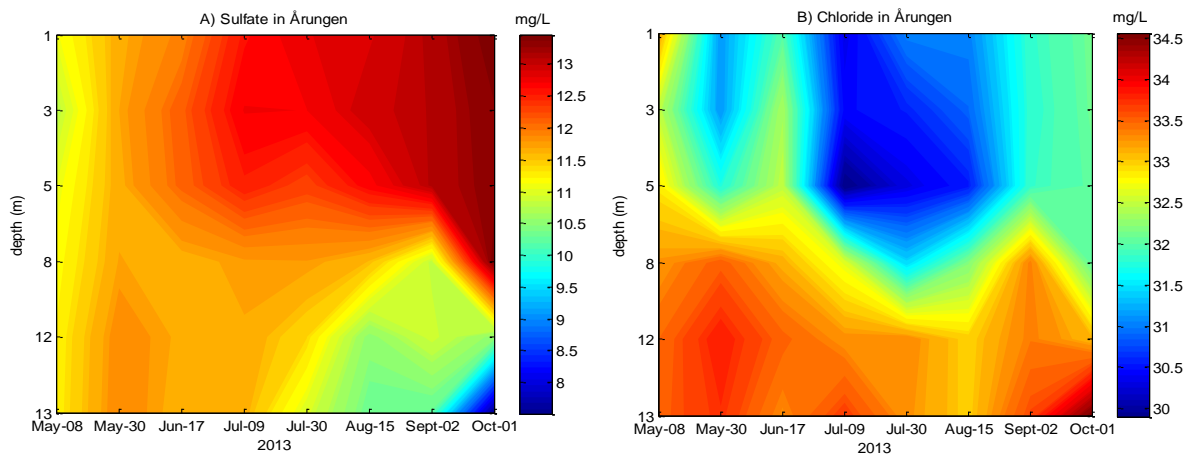


Figure 36 Time – depth isopleth diagrams of A) sulfate (mg L^{-1}) and B) chloride (mg L^{-1}), in Lake Årungen from 8th of May until 1st of Oct 2013.

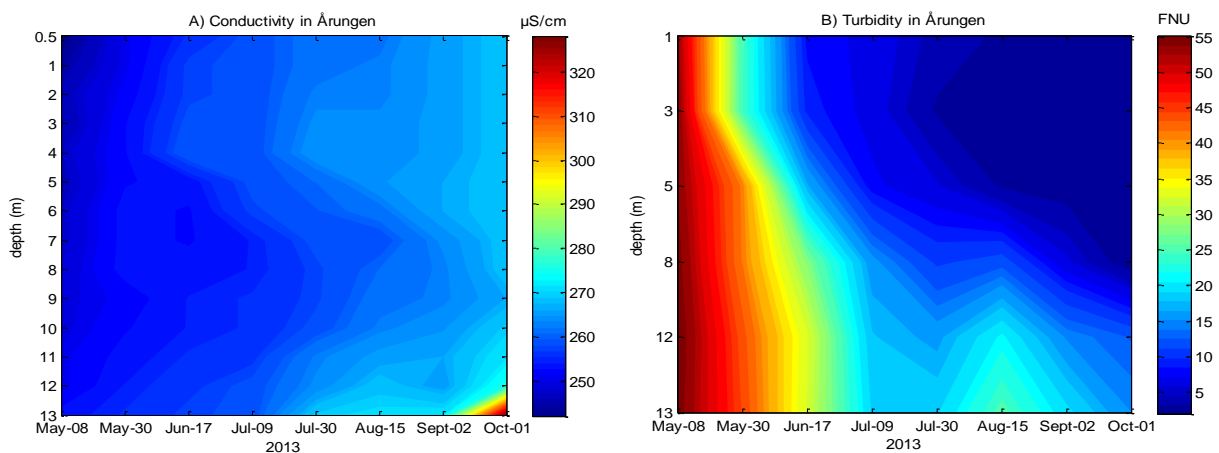


Figure 37 Time-depth isopleths diagrams of A) conductivity ($\mu\text{S cm}^{-1}$) and B) turbidity (FNU), in Lake Årungen from 8th of May until 1st of Oct 2013.

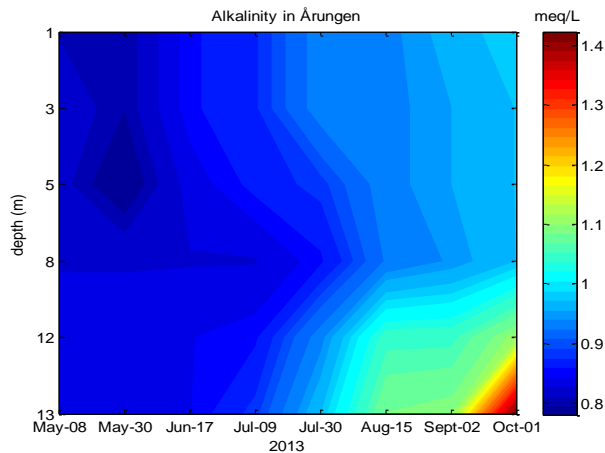


Figure 38 Time – depth isopleths diagrams of alkalinity (meq L⁻¹) in Lake Årungen from 8th of May until 1st of Oct 2013.

5.6 Epilimnetic average concentrations of key parameters

The concentrations presented in figure 39 to 40 are based on mean values in the depth of 1-3 meters in Lake Østensjøvannet and 1-5 meters in Lake Årungen, which is an estimate of the epilimnetic values in the two lakes respectively.

5.6.1 Lake Østensjøvannet

The TP concentration was 140 µg L⁻¹ at the 8th of May and became significantly reduced by the 30th of May (74 µg L⁻¹). By the 2nd of September the concentration was reduced to 31 µg L⁻¹ and increased to 80 µg L⁻¹ at the 1st of Oct. The TN concentration was initially 3 mg L⁻¹ and increased in the 30th of May to 7 mg L⁻¹. Throughout the rest of the season TN concentration became gradually reduced to 2 mg L⁻¹. Si concentration only showed minor fluctuations over time in the epilimnion between 2 and 3 mg L⁻¹. The chl_a concentration peaks was registered on 30th of May, 30th of July and 2nd of Sep at 43, 50 and 49 µg L⁻¹ respectively. The fluctuations of nutrient concentrations were not reflected by the chl_a levels.

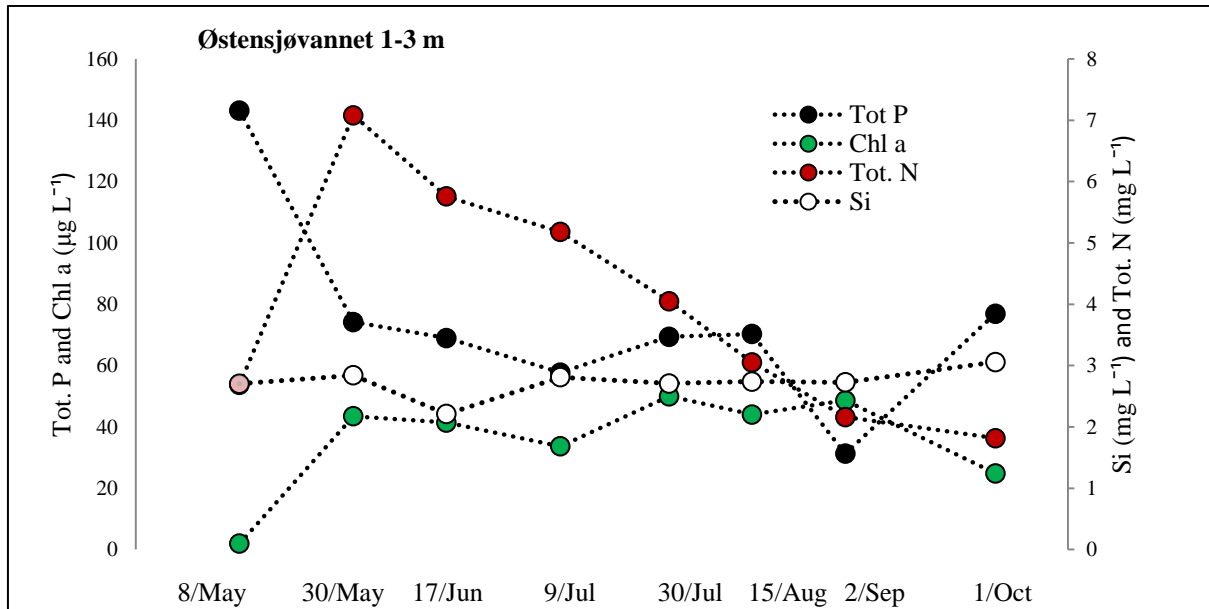


Figure 39 Mean values of total phosphorus (TP), total nitrogen (TN), chlorophyll *a* (*chl a*) and silicon (Si), in the epilimnion (1-3) of Lake Østensjøvannet from 8th of May until 1st of Oct 2013.

5.6.2 Lake Årungen

In Lake Årungen the initial values for TP in the epilimnion was $130 \mu\text{g L}^{-1}$ (figure 40) and the concentration was gradually reduced with time to $19 \mu\text{g L}^{-1}$ by the end of the season. The highest TN concentrations were registered in 30th of May with a concentration of 4.54 mg L^{-1} and reduced to 3.13 mg L^{-1} by the end of the season. *Chl a* concentration were initially $3.13 \mu\text{g L}^{-1}$ and had two peaks during the season at 17th of June and 30th of July with a concentration of 31.27 and $28.32 \mu\text{g L}^{-1}$ respectively. Inbetween the peaks there was a reduction in concentration that correspond to reduction in TP. Silicon was reduced to less than 1 mg L^{-1} towards the end of the season, which also correspond to a general reduction of *chl a* concentration.

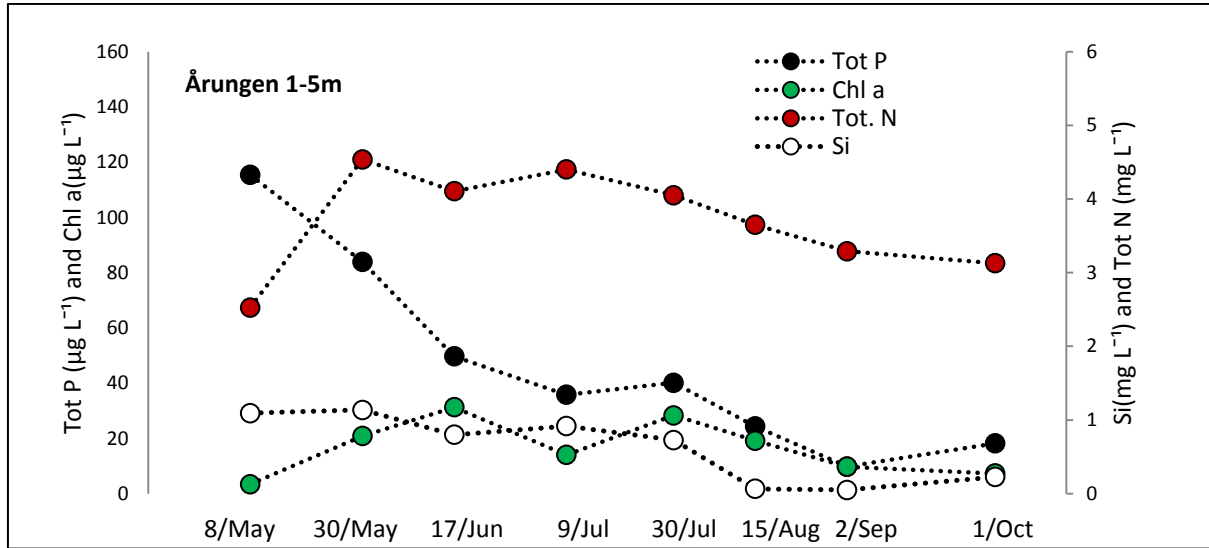


Figure 40 Mean values of total phosphorus (TP), total nitrogen (TN), chlorophyll *a* (*chl a*) and silicon (Si), in the epilimnion (1-5m) in Lake Årungen from 8th of May until 1st of Oct 2013.

5.7 Phytoplankton biomass

5.7.1 Lake Østensjøvannet

In Lake Østensjøvannet, Cryptophytes were the dominating group in early summer and peaked in the end of May (figure 41). The Cryptophytes continued to dominate all through the season (except between 30 July – 15 Aug when green algae dominated). The diatom biomass peaked in May and from the beginning of July the group was low in abundance. Golden algae had a relative large biomass in Lake Østensjøvannet until 30th July, when the biomass concentration was reduced. By the end of July the combined phytoplankton biomass had become heavily reduced and Cyanobacteria only occurred in small concentrations on an average throughout the season.

5.7.2 Lake Årungen

For Lake Årungen, the peak in total phytoplankton biomass (figure 45) occurred in the end of July and middle of August with levels above 7000 mg WW m⁻³. The dominating group in Lake Årungen in early summer was the Cryptophytes (2700 mg WW m⁻³). By the 15th of August the concentration was below 1000 mg WW m⁻³.

In July the phytoplankton community was dominated by diatom algae (which peaked in abundance at the 15th August at 5500 WW mg m⁻³). Between 15th of Aug and 2nd of September

there was a reduction of phytoplankton biomass in general due to a reduction in diatom biomass ($< 200 \text{ mg WW L}^{-1}$), Diatoms then increased in abundance once more from the 2nd of Sept until 1st October ($1200 \text{ mg WW m}^{-3}$).

Other groups in Lake Årungen were golden algae, green alga and cyanobacteria, each of these groups occurred only at a low concentration.

5.7.3 Phytoplankton subgroups

Figure 42 to 44 shows the composition within each phytoplankton group in Lake Østensjøvannet and figures 46 to 48 shows the composition of each phytoplankton group in Lake Årungen. The dominating cryptophytes in Lake Årungen was *Cryptomonas* sp, the same group also dominated in Lake Østensjøvannet early in the summer season. By the 17th of June the group had ceased being the dominating one and *Chroomonas* sp had taken over. Other similarities between the lakes, in composition of cryptophytes were *Rhodomonas lacustris*, quite abundant in both lakes (more so at Lake Østensjøvannet). The second most dominating group in Lake Årungen was *Cryptomonas erosa*, which was not present at any significant quantity in Lake Østensjøvannet. The dominating diatom in Lake Årungen was *Fragelaria crotonensis*. The group which was found in both lakes was *Aulacoseira* sp. It was the most dominating in Lake Østensjøvannet in late summer, while in Lake Årungen it was only present in small quantity in the beginning of the season. Table 5 shows the chl a /biomass ratio in the lakes at the different sampling dates. The ratio increased significantly in Lake Østensjøvannet in the latter half of the season, while Lake Årungen had decreasing ratios.

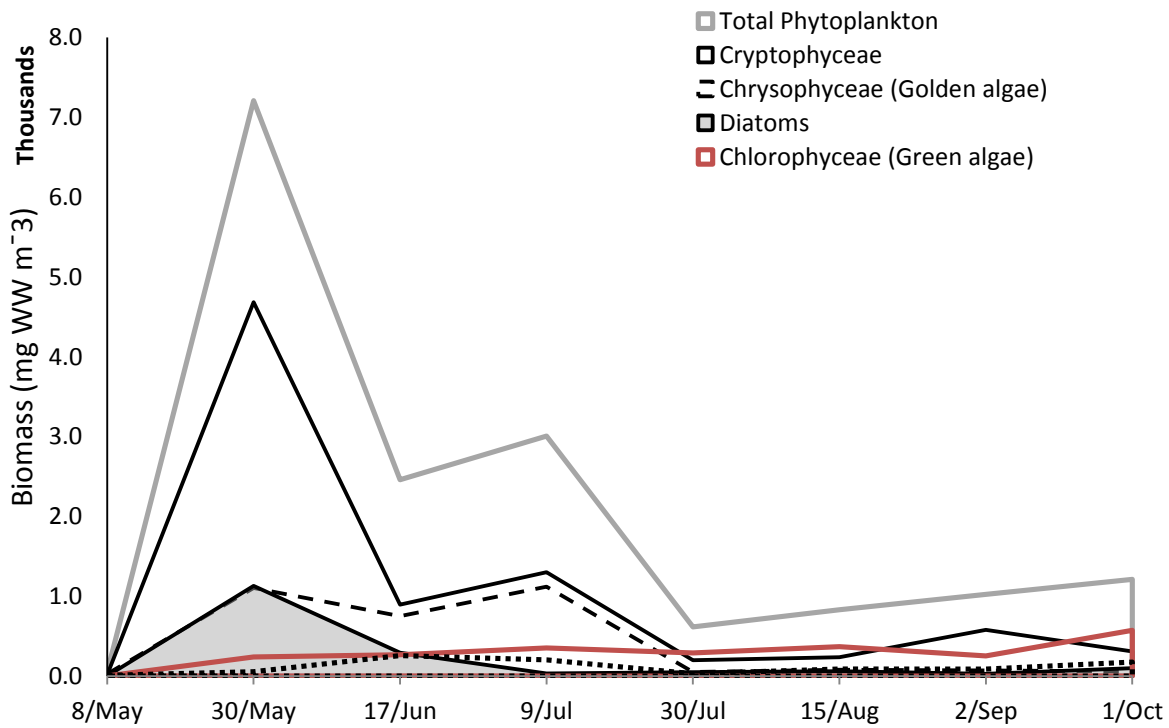


Figure 41 Phytoplankton biomass (mg WW m^{-3}) and composition in Lake Østensjøvannet from 8th of May until 1st of Oct 2013.

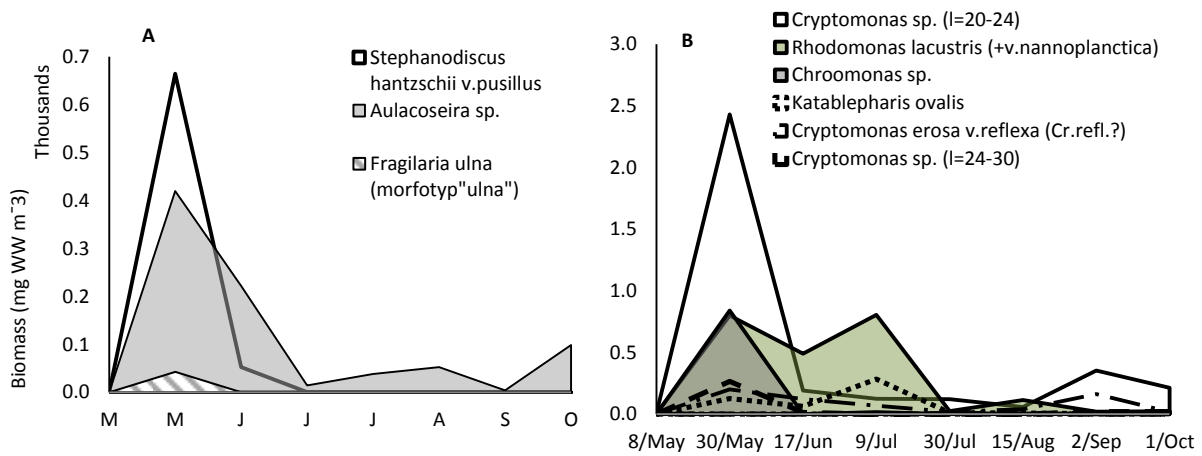


Figure 42 A) Dominating diatom groups and B) Cryptophyte groups in Lake Østensjøvannet from 8th of May until 1st of Oct 2013.

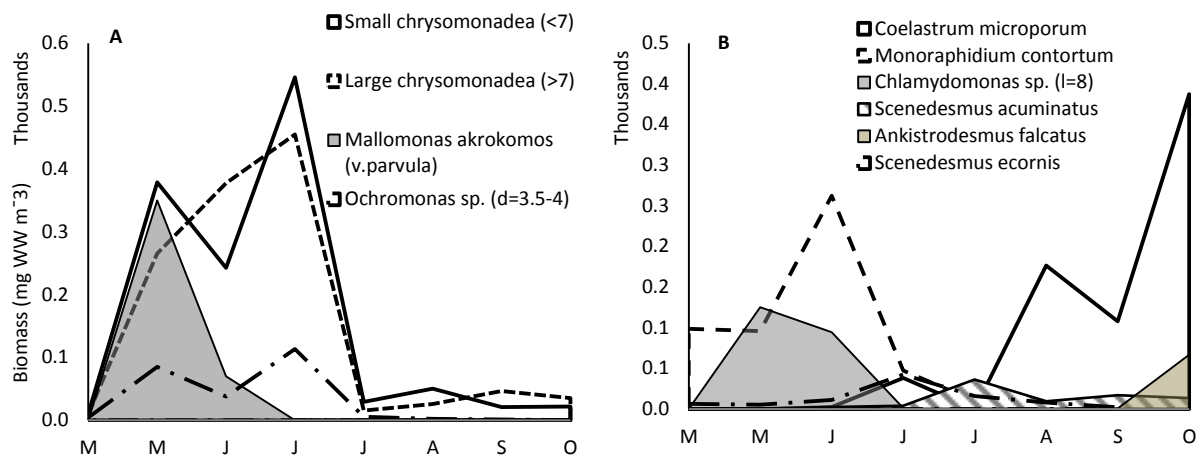


Figure 43 A) Dominating crysophyceae groups and B) dominating groups of green algae in Lake Østensjøvannet from 8th of May until 1st of Oct 2013.

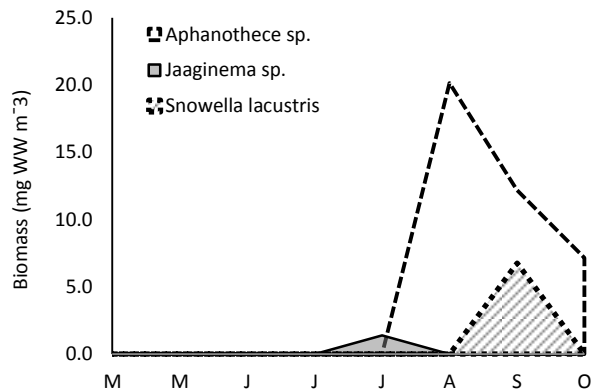


Figure 44 Dominating cyanobacteria groups in Lake Østensjøvannet from 8th of May until 1st of Oct 2013.

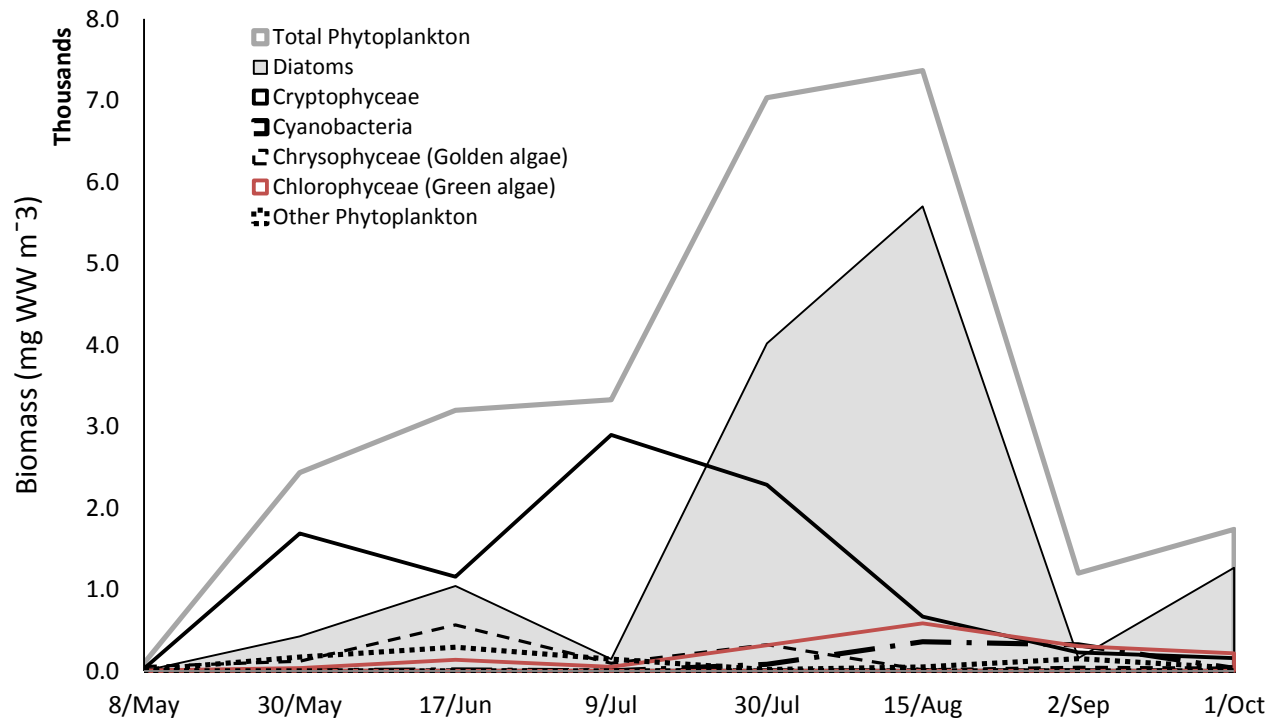


Figure 45 .Phytoplankton biomass (mg WW m⁻³) and composition in Lake Årungen from 8th of May until 1st of Oct 2013.

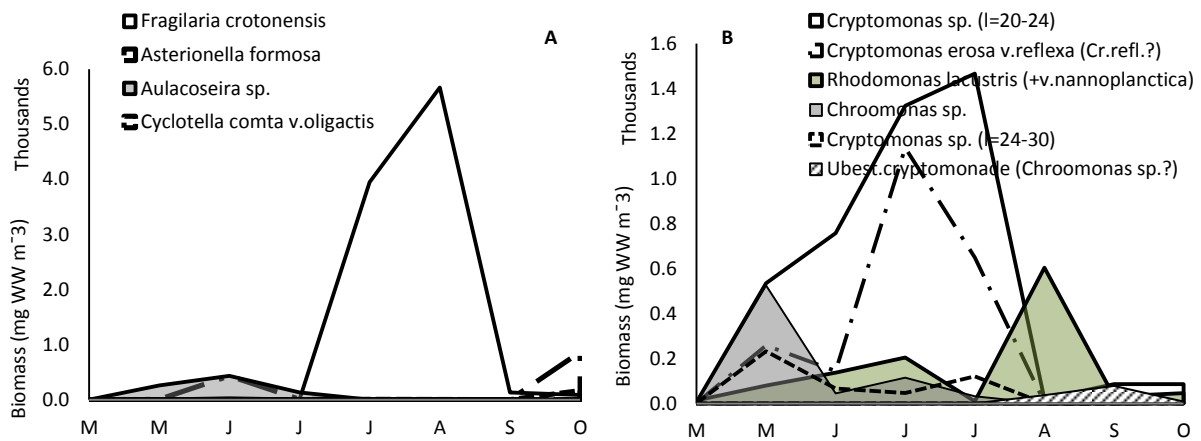


Figure 46 A) Dominating Diatom groups and dominating Cryptophyte groups (B) in Lake Årungen

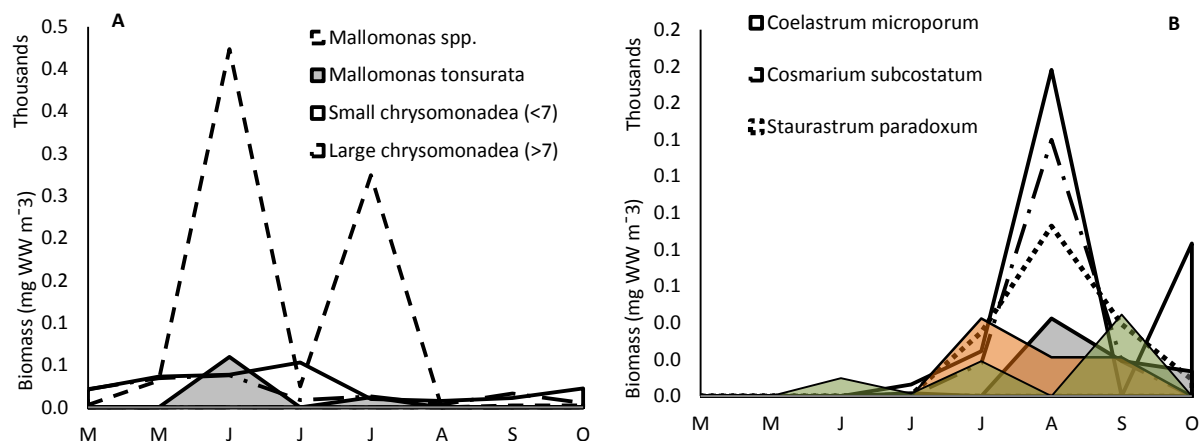
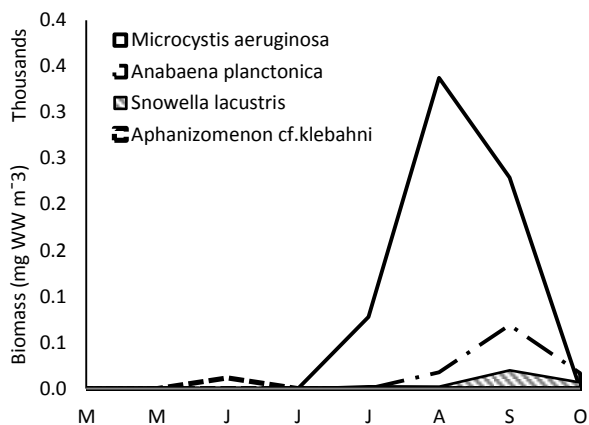


Figure 47 A) Dominating Crisophyceae groups and B) dominating green algae groups in Lake Årungen from 8th of May until 1st of Oct 2013.

Table 5 Chla/BMcount for Lake Østensjøvannet and Årungen.



Date	Østensjøvannet	Årungen
	Chla/BM _{count}	Chla/BM _{count}
08-May	3.4%	4.4%
30-May	1.0%	1.2%
17-Jun	1.9%	1.4%
09-Jul	1.3%	0.6%
30-Jul	9.7%	0.7%
15-Aug	7.9%	0.3%
02-Sep	5.4%	0.7%
01-Oct	2.3%	0.4%

Figure 48 Dominating cyanobacteria groups in Lake Årungen from 8th of May until 1st of Oct 2013.

5.8 Water quality parameters in the stream, Bølstadbekken

Suspended solids in Bølstadbekken (figure 49 A) were initially at a concentration of 30 mg L⁻¹, in the upstream and downstream location respectively. For all the sampling dates except 30th of May, the results showed a lower concentration at the downstream location.

The fraction of organic suspended solids (figure 49 B) was relatively low, early in the period, and increased with time. The organic fraction was frequently higher in the upstream location, which shows that both inorganic and organic solids are retained, although the retention mechanisms are stronger for the inorganic fraction.

TP concentrations (figure 50 A) were consistently higher at the upstream location. At the end of July the concentration differences between the upstream and downstream location were $\sim 11 \mu\text{g L}^{-1}$. By the 1st of October the difference had increased to $44 \mu\text{g L}^{-1}$. On the last day of sampling (29th of October) the overall concentration in the stream had increased but the difference between upstream and downstream location was reduced. $\text{PO}_4\text{-P}$ (figure 50 B) was initially unchanged between the upstream and downstream location until the 30th of July when the downstream location exhibit higher concentration than the upstream. This pattern persisted for the rest of the season with the largest differences in the range of $19 \mu\text{g L}^{-1}$ in the end of July to a difference of 10 and $11 \mu\text{g L}^{-1}$ in August and September respectively. In October the concentration differences were reduced.

Total Nitrogen (figure 51 A) was generally higher at the upstream location, except for the last day of the season, when the downstream concentration exceeded the upstream concentration, as a result of increased runoff. The variations were larger early on in the time period, which indicate high outputs from Lake Østensjøvannet. The concentration of TN increased quite drastically from the first day in field (2.6 mg L^{-1}) until the second, when the concentrations were 6.5 and 5.7 mg L^{-1} at the upstream and downstream location respectively. The lowest concentration was registered in the beginning of October with 1.7 mg L^{-1} . TN was almost totally dominated by the nitrate fraction (figure 51 B), while ammonia was relatively low (figure 52). Nitrate increased at the downstream location, and ammonia was reduced at the downstream location in the later part of the season. The relative differences in concentration of ammonia between the upstream and downstream location were of low order and only varied between 0.01 to 0.2 mg L^{-1} .

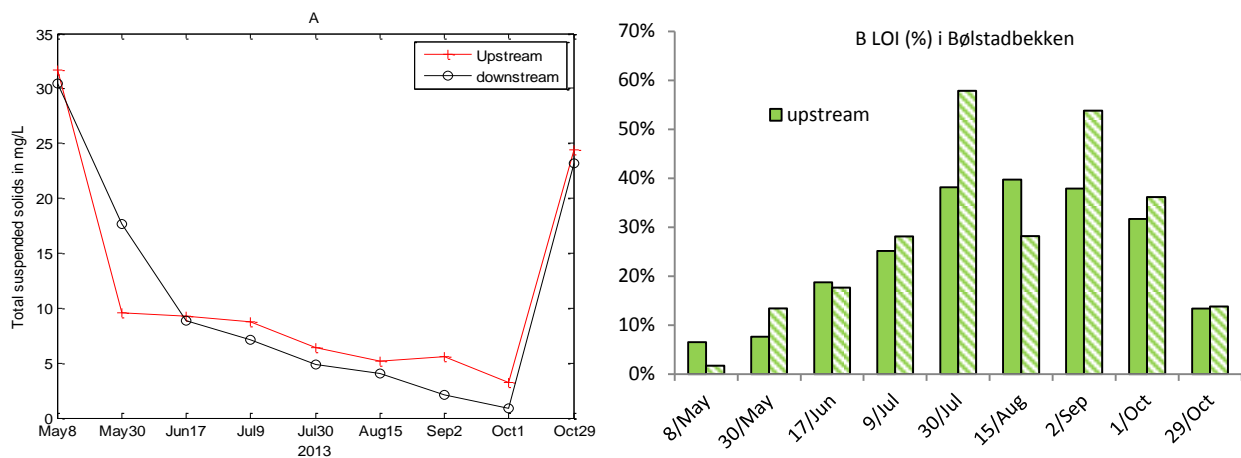


Figure 49 A) Total solids (mg L^{-1}) and B) relative fractions of organic suspended solids (loss of ignition) at the upstream and downstream location in Bølstadbekken, from 8th of May until 1st of Oct 2013.

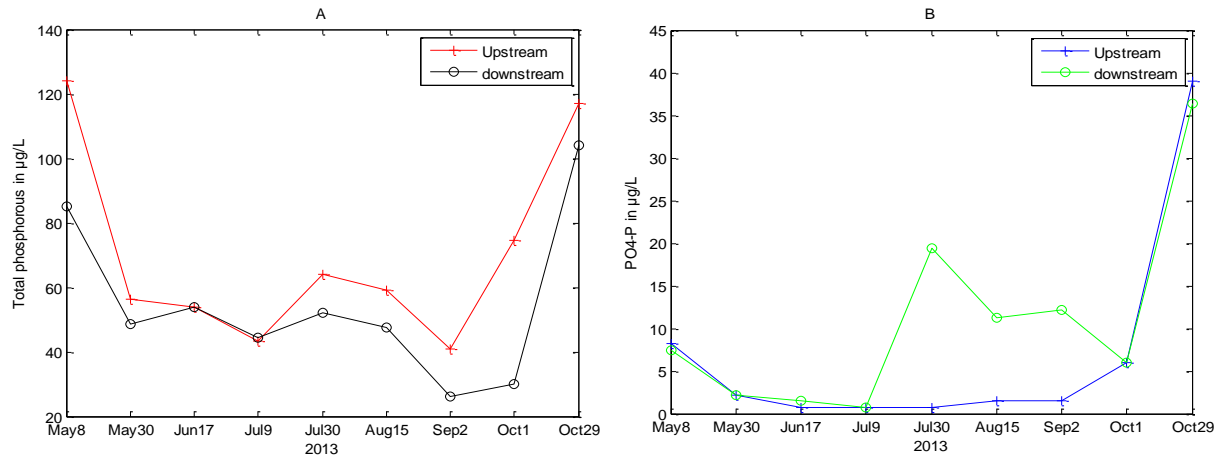


Figure 50 A) Total phosphorus ($\mu\text{g L}^{-1}$) and B) $\text{PO}_4\text{-P}$ ($\mu\text{g L}^{-1}$), at the upstream and downstream location in Bølstadbekken, from 8th of May until 1st of Oct 2013.

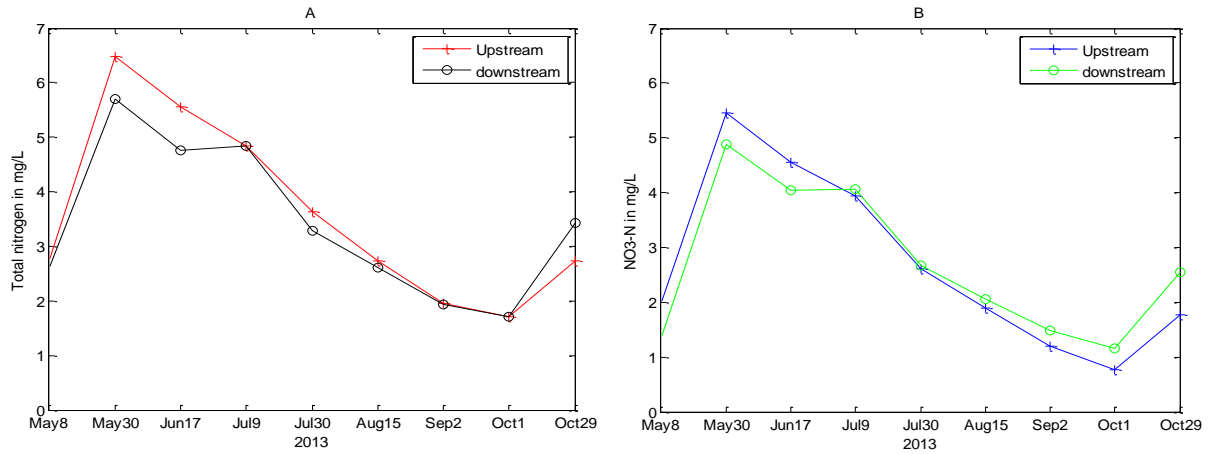


Figure 51 A) Total nitrogen (mg L^{-1}) and B) nitrate (mg L^{-1}) at the upstream and downstream location in Bølstadbekken, from 8th of May until 1st of Oct 2013.

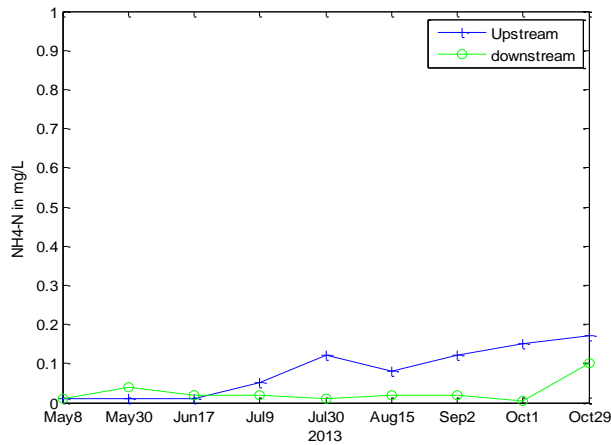


Figure 52 Concentrations of ammonia (mg L^{-1}) in the upstream and downstream location at Bølstadbekken, from 8th of May until 1st of Oct 2013.

5.8.1 Chla and silicon

Chla and silicon is plotted in figure 53 (A and B respectively). Chla concentrations were consistently lower at the downstream location. The maximum was registered at the 30th of May, for the upstream and downstream locations (46.5 and 30.4 $\mu\text{g L}^{-1}$ respectively). The concentration peaks followed the peaks in Lake Østensjøvannet both for the upstream and downstream location, but they were not consistent with the concentration maximums in Lake Årungen.

Silicon concentration became increased at the downstream location, with the exception of 15th of August. There was a temporal increase in silicon at the upstream location with start values of 2.69 mg L^{-1} , and at the end the concentration is 2.99 mg L^{-1} , which indicate less utilization in Lake Østensjøvannet in the latter half of the season.

5.8.2 Total organic carbon, dissolved organic carbon and water color

TOC and DOC concentrations are presented in figure 54 A and B respectively. The concentration curves for the two parameters were similar throughout the season. The concentrations were higher at the downstream location during the first half of the season. During the months of low precipitation, July – Sep, the upstream location presented higher concentrations. The minimum concentration was registered on the 17th of June with 6.17 and 6.86 mg L^{-1} respectively. The maximum concentrations were registered on the first of October, 8.06 mg L^{-1} and 7.55 mg L^{-1} , at the upstream and downstream location respectively. This coincides with the gradual increase in Lake Østensjøvannet.

At low flow conditions, TOC and DOC only differed by 0.08 mg L^{-1} at the upstream location, as seen for the 30th of July. The greatest difference between TOC and DOC was found at the first sampling occasion for both the upstream and downstream location with 0.7 and 0.54 mg L^{-1} respectively.

Water color in figure 55 was highest (53 and 59 mg L^{-1} upstream and downstream respectively) at an early stage of the season and was gradually reduced until the 1st of October (28 mg Pt^{-} at both locations). The greatest difference between the two locations was registered from the 8th of May until 17th of June, with increasing color downwards in the stream. Later on in the season only small differences were registered.

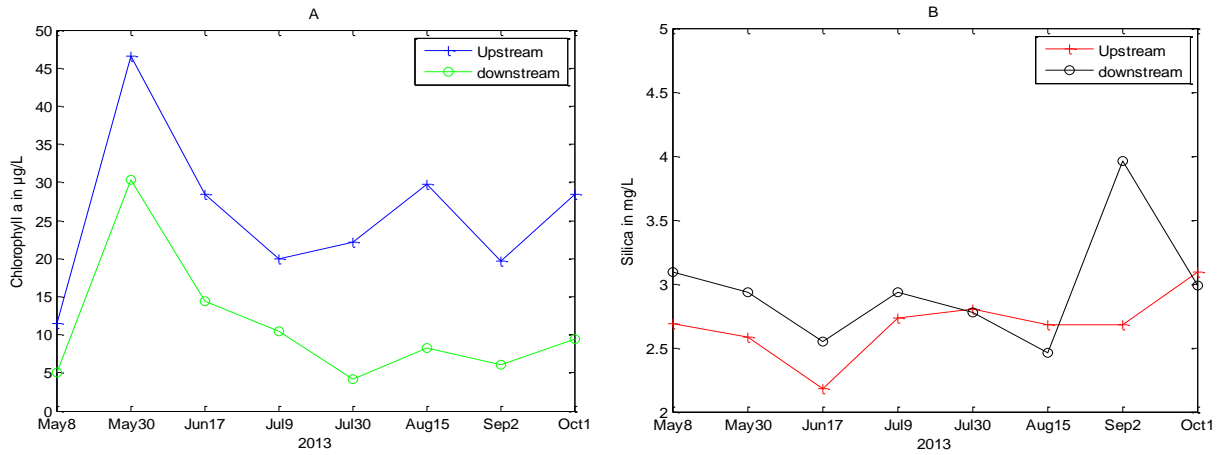


Figure 53 A) Concentrations of chlorophyll *a* ($\mu\text{g L}^{-1}$) and B) silica (Si) (mg L^{-1}) at the upstream and downstream location in Bølstadbekken, from 8th of May until 1st of Oct 2013.

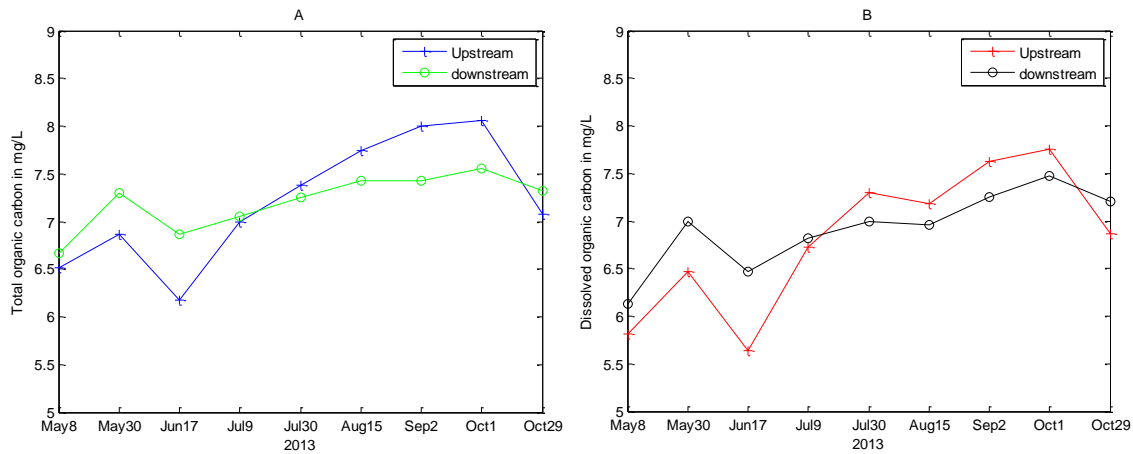


Figure 54 A) Concentrations (mg L^{-1}) of total organic carbon and B) dissolved organic carbon (mg L^{-1}) at the upstream and downstream location in Bølstadbekken, from 8th of May until 1st of Oct 2013.

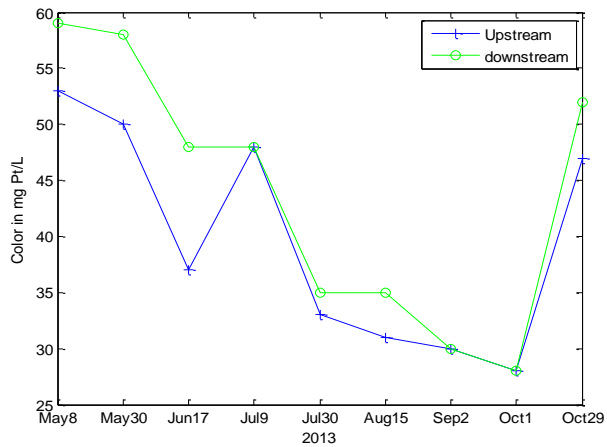


Figure 55 Water color ($\text{mg L}^{-1} \text{Pt}$) at the upstream and downstream location in Bølstadbekken, from 8th of May until 1st of Oct 2013.

5.8.3 Other in-stream parameters

Figure 56, show the percentual change from the upstream location to the downstream location, for the cations; Na^+ , Mg^{2+} , K^+ , Ca^+ and Fe^{2+} during the months of low discharge. The general increase from upstream to downstream was 10 – 20% during these months, except for Fe^{2+} , which was decreased at the downstream location at the two last occasions.

The change in conductivity from the surface water in Lake Østensjøvannet, through the stream and to the surface water of Lake Årungen was presented in figure 57. During the months of low discharge the conductivity was progressively increased downstream in Bølstadbekken.

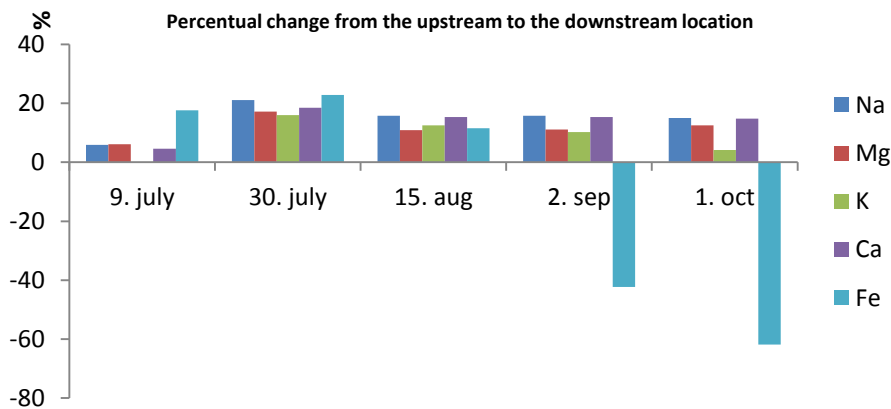


Figure 56 Concentration change (%) from the upstream location to the downstream location, for the cations; Na^+ , Mg^{2+} , K^+ , Ca^+ and Fe^{2+} , from 9th of July until 1st of Oct 2013.

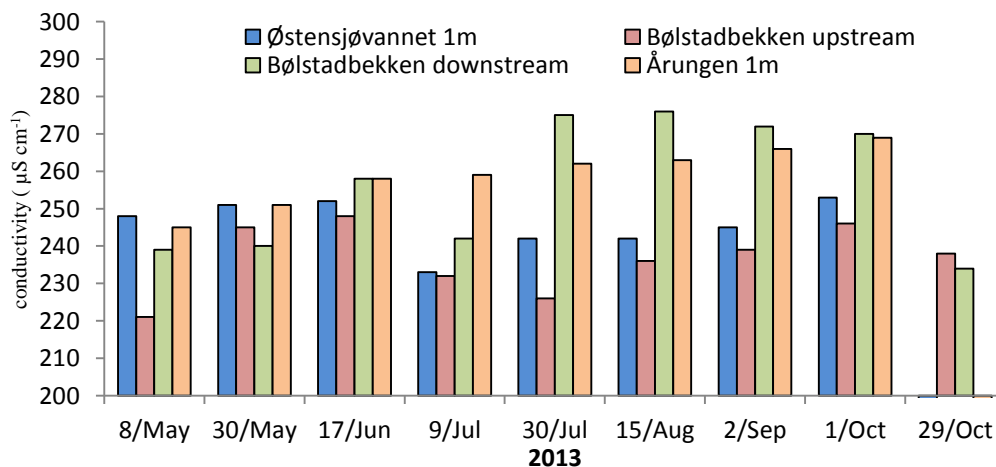


Figure 57 Conductivity in Lake Østensjøvannet at 1 meters depth (blue bar), Bølstadbekken upstream (pink bar), Bølstadbekken downstream (green bar) and Lake Årungen at 1meters depth (orange bar), from 8th of May until 29th of Oct 2013.

6 Discussion

6.1 Thermal stratification and mixing depth

Weather conditions are of great importance for the stability of the thermocline. The growth season of 2013 was both warmer and dryer compared to the normal period (1961 -1990) (figure 5 A). High summer temperatures and low precipitation promoted good conditions for long and stable stratification periods. As seen in figure 11 and 12 the thermocline was established at 30th of May in both lakes, due to warming of the surface water. The thermocline was located at a shallower depth initially and moved deeper in the water column towards the end of the season. The first part of the season were therefore, characterized by shallow mixing regimes while deeper mixing regimes developed in late summer. The temperature in the hypolimnion of Lake Årungen was stable at 7 degrees from 30th of May until the end of the sampling period. In Lake Østensjøvannet however there is a gradual heating of the hypolimnion until the end of July, suggesting that the two layers is not totally enclosed from each other. Temporary breakdowns of the stratification gradient are common for shallow lakes, where wind force can cause mixing the hypolimnetic water with warmer water from the epilimnion (Jones 2001; Kjensmo 1994; Krivtsov & Sigeo 2005). Further the temperature gradient in Lake Østensjøvannet is less steep and the concentration distribution of chemical parameters in the water column was more homogenous in the vertical profile than in Lake Årungen. As seen especially for total phosphorus, phosphate, total nitrogen, nitrate, and chloride (16 A & B, 18 A & B and 23B). Towards the later part of the season, the concentrations of chemical substances became more differentiated in the water column, which suggest less vertical mixing. The decrease in water inflow from e.g. Skuterudbekken (figure 6 A) resulted in less water entering from the littoral zone, which can destabilize the thermocline (Romarheim et al. 2012). The retention time increased due to less precipitation and discharge in the tributaries which also results in more stable chemical gradients in the lake. As both lakes became less turbid during the season, deeper penetration of insolation affects the thermal properties of the water column as seen by the correspondence between deeper thermoclines and light penetration in both lakes in August to September (figure 9, 10, 11 & 12).

6.2 Optical properties

The transparency in the lakes showed a good compliance with the total suspended load, especially in the early growing season, when inorganic solids dominated (figure 7 and 8). The decrease in total suspended solids and an increase of organic solids, compared to inorganic solids favored a deeper secchi depth in the lakes. This suggests that the inorganic fraction is an important controlling factor for light availability in the lakes.

The results from the light meter (figure 9 and 10) show that the wave lengths of photosynthetic active radiation, PAR are consumed within the first meter in Lake Østensjøvannet in the beginning of May. This coincides with the high turbidity measurements (figure 24 B). The variation in secchi depth corresponds to variations in suspended load throughout the season as seen in figure 7. In Lake Årungen, the increased depth of PAR coincides well with the measured secchi depth and reduction of suspended particles. The higher suspended load (both inorganic and organic) seen in Lake Østensjøvannet was determining for the poorer light conditions in the lake. Considering the low precipitation during the season, the secchi depth was most likely deeper compared to a growing seasons with richer precipitation, due to less incoming particle load and longer retention time. Since blue light is absorbed first in both lakes (figure 9 & 10), the water column contains relatively large amounts of humic substances (Wetzel 1991). When green light penetrated deeper in Lake Østensjøvannet than red light, the water column was most likely dominated by phytoplankton, which is known to absorbs light in the red end of the spectra (Bowling 1989). The episode of deep green light penetration, also corresponds with increasing *chl a* concentration and organic suspended solids.

The negative correlation between the euphotic depth and suspended solids is strong in both Lake Østensjøvannet and Lake Årungen with R^2 of 0.82 and 0.77 respectively (table 3), however the results for Lake Østensjøvannet becomes misleading by the high suspended load of the first sampling date (appendix 7). Water color show a high degree of explanation in Lake Årungen (R^2 0.91), but is poorly correlated with the euphotic depth in Lake Østensjøvannet (table 3). The DOC concentration correlates positively with euphotic depth in both lakes, which reflects a poor connection between color and DOC (table 3). The water color and absorbance at 254nm has a positive correlation which further supports relatively large amounts of humic substances (table 4). The negative relationship between color and DOC suggests that humic substances become photo-

bleached in the epilimnion during the summer. The increasing color is more coupled with increasing DOC in the hypolimnion than in the epilimnion (figure 22 and 35).

Water color correlates positively with suspended solids in Lake Årungen (table 4) which indicates that humic substances are related to the particle load. In Lake Østensjøvannet DOC showed a low degree of explanation for water color. The increase in TOC and DOC in Lake Østensjøvannet is likely more governed by high lake metabolism, which also explain why the two parameters increase progressively, despite low inputs from the catchment, since most organic carbon in eutrophic lakes originates from the autochthonous pool (Jonsson et al. 2001).

The factor which has the overall strongest impact on light attenuation in Lake Årungen is water color and suspended particles, similar results was found by Bowling (1989). In Lake Østensjøvannet, light attenuation was best explained by the suspended load.

6.3 Oxygen saturation, pH and chl_a

Dissolved oxygen and pH values were strongly influenced by biological activity in the lakes. High primary production in the surface water is usually correlated with an increase in pH and oxygen saturation, by the photosynthetic removal of CO₂, and production of oxygen molecules to the water column. High pH therefor corresponds to DO super saturation in the surface water as seen in figure 13 and 14. The chl_a concentration peaks in Lake Østensjøvannet do not correspond with the pH and DO peaks, as well as it do in Lake Årungen. Elevated pH values in Lake Årungen coincide with periods of oversaturation and high chl_a concentrations. In Furthermore pH values are generally higher in Lake Årungen for the period between 17th of June until 2nd September and the chl_a concentrations are consistently higher in Lake Østensjøvannet.

Oxygen became depleted in the hypolimnion of both Lake Østensjøvannet and Lake Årungen (figure 13 and 26). In Lake Østensjøvannet, oxygen was reduced to less than 0.1 mg L⁻¹ by the 17th of June (figure 11). In contrast, oxygen levels lower than 0.1mg L⁻¹ occurred first in the middle of July in Lake Årungen (figure 12). Due to the higher production in Lake Østensjøvannet, as indicated by higher chl_a concentrations, DO saturation and organic suspended solids (figure 15A, 13B and 7B), the oxygen demand became higher in the hypolimnion, as sinking organic matter accumulated over the sediments. Since the lake is shallower than Lake Årungen the ratio between the epilimnion and hypolimnion is higher, which results in more accumulation of organic matter over the sediments.

6.4 Nutrients

6.4.1 Phosphorus

The relationship between *chl_a* and total phosphorus is well documented by e.g. Dillon & Rigler (1974). High mean summer *chl_a* concentrations are often strongly correlated with high TP values after spring turnover. This relationship corresponded well to the observations of this study. The higher TP concentration in Lake Østensjøvannet, by the time of first sampling, was followed by higher *chl_a* concentrations throughout the season compared to Lake Årungen. In the middle of June phosphate concentration had been reduced to $< 2 \mu\text{g L}^{-1}$ in the entire water column of the lake, and there seemed to be algae growth throughout the vertical profile (figure 15A & 16B). In contrast; elevated *chl_a* values were restricted to the surface water in Lake Årungen (figure 28A). Phytoplankton growth at 6m depth is unlikely considering that incident light is consumed within the first 3 – 4 meter. Water exchange is most likely occurring between the epilimnion and hypolimnion, which explains why phosphate decreased in the entire profile. Conversely, the phosphate concentration in Lake Årungen was stable at the initial concentrations in the hypolimnion due to a stable mixing barrier.

An increase in total phosphorus and phosphate over the sediments occurred in both lakes by the end of July (figure 16 & 29). Considering that the increase of phosphate concentration in Lake Østensjøvannet is relatively low ($33 \mu\text{g L}^{-1}$), the increase is most likely derived from mineralization of the organic matter pool. The increase of phosphate over the sediments in Lake Årungen at the 1st of October was however much higher ($430 \mu\text{g L}^{-1}$) in comparison, which indicate phosphorus release from the sediments.

6.4.2 Total nitrogen, nitrate and ammonium

In Lake Østensjøvannet there was large temporal concentration variations of TN in the water column and limited vertical differences (figure 17 A). Conversely, Lake Årungen show large differences with depth. As seen in figure 17 A and B, there are no vertical concentration gradients which means that there are more vertical mixing in Lake Østensjøvannet which resulted in continuous nitrate supply to the hypolimnion. In Lake Årungen the inputs of nitrate was foremost accumulated in the epilimnion. Consequently, nitrate concentrations under the thermocline were not affected by nitrogen inputs after the temperature gradient was established. High nitrogen inputs in the end of May, was registered in both lakes and was coupled with

fertilizing application on the surrounding agricultural fields. The results are consistent with annual nitrogen loss rates from the agricultural field, Skuterud (Bechmann & Deelstra 2013). Nitrate made up a large part of TN early in the season, approximately 80 % in both lakes. Towards the end of the season it only represents 6 and 11 % of TN at the deepest point in Lake Årungen and Lake Østensjøvannet respectively (figure 19 and 32). The subsequent reduction of nitrate in the hypolimnion of both lakes (figure 18B and 31B) is the result of denitrification under progressively lower oxygen concentrations. Denitrification coupled with internal phosphorus loading may adjust the TN:TP relationship towards lower ratios which might affect the phytoplankton community (Downing & McCauley 1992) although the phosphate increase and denitrification intensified in a later stage of the season and had little impact on the current growing season. The concentration of ammonium in Lake Østensjøvannet increased in the deeper parts in June (figure 20 A), which is usually associated with degradation products of organic material deposited throughout the season (Satoh et al. 2002) . Due to the anoxic conditions ammonium is prevented from being oxidized to nitrate (Satoh). Similar conditions are observed in Lake Årungen towards the end of the season, with a relatively low increase in comparison. This further reflects the temporal differences in productivity and mineralization intensity between the two lakes.

6.4.3 Silicon

In Lake Østensjøvannet there was a reduction of silicon between 1-3 meters depth at June 17th, (figure 15 B). This is a result of diatom utilization in the end of May as seen by a relatively high diatom biomass (figure 41). The silicon concentration was constant in the surface water after June 17th, at concentrations between 2.6 and 2.8 mg L⁻¹. The threshold concentration of silicon for diatom utilization in freshwater is considered to be 0.5 mg L⁻¹ (Wetzel 2001). In Lake Årungen silicon was reduced at the 17th of June and 30th of July, which corresponds to high chl a concentrations (figure 28 A & B). Towards the end of the season the concentration was reduced to less than 0.5 mg L⁻¹ in the epilimnion and became limiting for phytoplankton growth. Silicon is increased in concentration towards the end of the season in the deeper water of both lakes. An increase of silicon concentration in the hypolimnion is associated with the degradation of organic matter (Satoh et al. 2002). The increase over the sediments occurs earlier in Lake

Østensjøvannet, and is related to the earlier bloom and subsequent decrease of diatom biomass (figure 41) and an overall higher lake metabolism.

6.5 Comparison of phytoplankton biomass and composition, and assessment of limiting factors

The results show that the same group of phytoplankton (Cryptophyceae) dominated in both lakes at an initial stage of the growing season, when the stratification gradient was located close to the surface. This group has adaptations that allow vertical positioning. Mechanisms to overcome sinking velocities are important since it enables phytoplankton groups to position themselves in the water column where conditions are optimal (Padisak et al. 2010; Reynolds et al. 1983). The beginning of the season was characterized by very turbid water and low transparency.

Consequently; such adaptations would make it possible to gain optimal light conditions. Reynolds et al (1983) found that the most dramatic changes in population were noted among the species that occurred early in spring (e.g. diatoms and Chrysophyceae). This corresponds well to the observations made both for Lake Årungen and Lake Østensjøvannet, where Cryptophyceae, Chrysophyceae (golden algae) and diatoms were abundant initially (figure 41 & 45).

The Cryptophyte group displayed a bimodal pattern with a new biomass maximum in the beginning of July for both lakes, although the second peak in Lake Østensjøvannet was much lower than the first. It seems like the initial conditions in the lakes were of such character that this group is favored, conversely conditions in the later part of the growing season did not favor this group in Lake Østensjøvannet.

Diatom algae had a strong presence in Lake Østensjøvannet in the end of May and were reduced heavily in June. The shallow mixing regimes in June and July were non favorable towards diatoms since phytoplankton species with high sinking velocities may fall out of suspension at shallow mixing regimes (Reynolds et al. 1983). Padisak et al. (2010) emphasized that diatoms are especially sensitive to changes in the thermal density gradients. Early spring blooms and rapid decline in diatom populations has been observed by e.g. Huisman et al. (2004), Johnk et al.(2008) and Reynolds et al. (1983). The diatoms decline in Lake Årungen also corresponded to shallow mixing, however, the diatom population recovered later in the growing season, in contrast to Lake Østensjøvannet. The study made by Znachor et al. (2013) showed that growth rates of dominating diatoms were positively correlated with light availability, but not with nutrient

concentrations, which suggest that the deep water mixing and improving light conditions in Lake Årungen favored this group. The only subgroup of diatoms, present in both lakes is *Aulacoseira* sp, which had a strong presence early in the season in Lake Østensjøvannet (figure 42A).

McCausland et al. (2001) showed that the population growth of *Aulacoseria* sp. is highly sensitive to light limitation caused by sinking from the well-lit surface. Further thermal stratification and low flow conditions will give this species a competitive disadvantage, as seen from the absence of a second bloom (figure 42A & 46A)

The *chl a* concentration in Lake Årungen was reduced alongside reductions in silicon concentration (figure 40). The second decline in diatoms in Lake Årungen is likely caused by silicon limitation since insufficient silicon concentrations often limits the duration of the diatom bloom (Poister & Armstrong 2003). Further the decline in silicon coincides with the diatom population collapse (figure 45) at a time when the mixed layer is deep (figure 12). Accordingly; at sufficient light and deep water mixing the limiting factor in Lake Årungen at the latter part of the season was nutrients, which are in line with predictions made by (Romarheim 2012).

Conversely, nutrient limitation in Lake Østensjøvannet is unlikely as seen by the lack of compliance between nutrients and *chl a* in figure 39. The control of phytoplankton biomass and composition can therefore not be attributed to nutrient availability alone as pointed out by e.g. Nydick et al.(2003) Peretyatko et al.(2007) Reynolds (1998), but must take into consideration physical constraints such as low insolation, temperature and deep water mixing and biological control such as competition between phytoplankton groups and grazing by zooplankton.

6.5.1 Cyanobacteria

Both lakes was previously hypertrophic in the 1970s and 80s, dominated by cyanobacteria (Andersen 1982; Erlandsen & Grøterud 1980; Grøterud & Rosland 1981).

As recognized by e.g. Huisman et al.(2004) cyanobacteria tend to be favored during periods of warm weather and weak wind mixing, accordingly the conditions during late summer and autumn 2013 should have been optimal in both lakes. Cyanobacteria biomass (figure 44 & 48) are nevertheless low in both lakes. The shift in algae composition in Lake Østensjøvannet, from being dominated by cyanobacteria, to a dominating phytoplankton community of diatoms and green algae have been recognized by Skovgaard et al.(2011). Cyanobacteria dominated in Lake Årungen by late summer of 2011 (Romarheim 2012). In contrast, diatoms were the dominating

phytoplankton group in Lake Årungen in late summer of 2013, which is more associated with oligotrophic lakes (Reynolds 1998). The possibility of phosphorus limitation in the epilimnion in Lake Årungen (figure 40) could give a competitive advantage for diatom algae since some diatom species are superior competitors for phosphorus (Tilman et al. 1986). Schindler et al. (2008) recognized that N additions to hypertrophic lakes can actually suppress cyanobacteria, by allowing other phytoplankton species to thrive. The high concentrations of nitrogen, especially in Lake Østensjøvannet can be beneficial for other species and suppressing for the cyanobacteria taxa. Whether the shift in dominating phytoplankton taxa is a result of nutrient control in the catchment, climate change or changes in lake dynamics, the absence of cyanobacteria appears to be caused by an overarching factor that influences both lake systems. Changes in retention time can produce large variations in annual phytoplankton biomass as recognized by Elliott et al. (2009). As shown by Padisak et al. (2010) phytoplankton communities undergo significant changes from one year to another, and also the phytoplankton composition are determined by the specific competitive arena of each lake. No certain conclusion can be made on the basis of this study, but further investigations into these observations could be of great value in order to unravel the controlling factors for cyanobacteria growth in the lakes.

6.5.2 Phytoplankton migration

The low abundance of cyanobacteria in Lake Østensjøvannet during the season of 2013, and the differences in the phytoplankton community between the lakes (as seen by figure 41 – 48), overthrows previous concerns of cell transport to Lake Årungen (Grøterud & Rosland 1981). Cell import of phytoplankton can however shape the community of another lake provided short retention times (Bergstrom et al. 2008). Accordingly; the weather conditions in 2013 made the lakes more enclosed than they would be during a season with high precipitation. Phytoplankton is also sensitive to physical damage (Bergstrom et al. 2008), which will further limit the phytoplankton transport. As seen in figure 53 A, the *chl a* concentration in Bølstadbekken is reduced downstream. The survival rate of phytoplankton cells are often determined by river characteristics (Kohler et al. 2002), in shallow streams losses are often higher due to sinking velocities, and since Bølstadbekken have reaches of high turbulence, the physical damage to the cells may be significant.

6.5.3 Evaluation of biomass and chl_a discrepancies

The result of the phytoplankton biomass estimation, were inconclusive for Lake Østensjøvannet. Even though the overall phytoplankton biomass was heavily reduced by the end of July, there was not any correspondent reduction in chl_a concentration, which was still relatively high. The DO and pH results (figure 13 B & 14) coupled with increasing organic solids further support continued growth in the lake after the decrease in algal biomass. When filtering the water for chl_a analysis, the filters were stained with green color, which is not consistent with reduced algal growth. Results from lake monitoring by Norwegian institute for water research (NIVA) during the summer of 2013 show a strong presence of autotrophic picoplankton in the lake (Haande 2014). Kasprzak et al.(2008) showed that such picoplankton can contribute significantly to the overall phytoplankton biomass, but not always be detected by microscopic examination. The observations of large amounts of picoplankton in the lake could give miss guided chl_a to biomass ratios. Conversely chl_a per unit of biomass can differ largely between taxonomic groups (Reynolds 1984). Highest proportions have been found in green algae and lowest in cyanobacteria. A high amount of green algae could explain why the chl_a/biomass ration increased, however Lake Årungen had approximately the same green algae abundance, with lower chl_a concentrations. The result is therefore better explained by autotrophic picoplankton, which can subject other phytoplankton taxa to competition of light. The organisms may act light scattering, as seen from the deep penetration of green light and reduced transparency in August (figure 9). Light limitation by picoplankton density at deep water mixing and competition for nutrients in the surface water especially phosphate, are possible theories, which can explain why other taxa became less dominant. These findings may have other implications for the lake since, protozoa are the predominant organism feeding directly on autotrophic picoplankton (Callieri & Stockner 2002). Accordingly; reducing the predation pressure on zooplankton by manipulating the planktivorous fish population in Lake Østensjøvannet, may not have the desired effect on water quality. However further studies can contribute to a better understanding of picoplankton interactions in the lake ecosystem.

6.6 Nutrient retention in Lake Østensjøvannet

The upstream location in Bølstadebekken is strongly influenced by the surface water in Lake Østensjøvannet as seen by the TP (figure 50), TN (figure 51), chl_a (figure 53) and TOC (figure

54) concentrations. Nitrogen that enters lakes by catchment runoff is mostly flushed out as nitrate (Faafeng et al. 1997), which is reflected by the immediate response to higher nitrogen loading in the end of May, at the upstream location in Bølstadbekken (figure 51). Retention processes within the lake are therefore not very strong for nitrate.

The water in early summer was very turbid and a large part of the suspended load was made up by inorganic particles as seen in figure 7. These particles can adsorb phosphate from the water column before settling to the sediments. Suspended solids can act as a buffer for dissolved nutrients by adsorbing especially P elements (Abell & Hamilton 2013). This retention process could contribute to the overall phosphorus retention in the Lake, provided that the particles are not already saturated in respect to P. A large portion of the incoming particle load is however transported out of the lake due to the short retention time, as seen by the corresponding high load of suspended solids in Bølstadbekken in the beginning of the season (49A). The retention of particles increased later in the season as the retention time became longer. Consequently; more particle bound phosphorus was retained. The net phosphorus flux in many eutrophic waters on a yearly basis is towards the sediments (Håkansson & Jansson 1983). However during the summer stagnation the nutrient retention capacity may be significantly reduced, especially in lakes that has received large phosphorus loads from sewage (Bostrom & Pettersson 1982; Granéli 1999; Jeppesen, E. et al. 2007; Schindler 2006). By comparing the results of this study with the previous study by Vekterli (1989) it shows that TP concentrations has been significantly reduced in the lake. The TP concentrations were in the range of 150 -200 $\mu\text{g L}^{-1}$ in late growing season in 1989. The TP concentration in 2013 ranged from 40 to 100 $\mu\text{g L}^{-1}$ and the concentration peaks were associated with the spring turnover and not with the later part of the season.

Lake Østensjøvannet has by its location functioned as accumulator of pollution from the catchment (Grøterud & Rosland 1981). On a yearly basis, the lake will normally act as a sink for phosphorus as suggested by Vekterli (1989). The extent of nutrient retention in Lake Østensjøvannet is however highly dependent on the prevailing hydrological conditions of the season. In larger lakes with longer residence time (approximately 1 year) far more nutrients will be lost from the productive zone due to sedimentation of dead algae (Schindler & Fee 1974). Kõiv et al. (2011) showed that lakes with a longer retention time, retained more of the external phosphorus. The increased hydraulic retention time consequently resulted in less nutrient transport downstream.

6.6.1 Internal phosphorus loading due to low redox conditions

The increase in parameters such as silicon, phosphate, ammonium, TOC and DOC are associated with mineralization processes (Sato et al. 2002) which is observed in Lake Østensjøvannet towards the end of the growing season (figure 15B, 16B, 20A and 21A & B). Increasing conductivity and alkalinity over the sediments further indicate dissolved nutrient salts associated with mineralization of organic matter (figure 24A & 25).

Some dissolution of phosphate from the redox sensitive fraction is however likely occurring, based on the elevated iron concentration and sulfate reduction (figure 17A & 23A). Highly elevated phosphate concentration over the sediments, coupled with an increase of dissolved divalent iron and a reduction of nitrate and sulfur concentration are signs associated phosphorus release from the sediments (Amirbahman et al. 2003; Holmer & Storkholm 2001).

Phosphate constituted approximately 20 % of the increase in total phosphorus load over the sediments for the 2nd of September (figure 17 B). If the increase in phosphorus concentration is largely made up by phosphate, as seen in Lake Årungen (< 60%) (figure 30B), it is more likely that the observed increase originates from Fe-P dissolution due to low redox potential. High concentration of nitrate (>6 mg L⁻¹) in the hypolimnion of Lake Østensjøvannet early in the season, functioned as a buffer against iron reduction. The importance of nitrogen in preventing phosphorus release from sediments, have been recognized by e.g. Bostrom & Pettersson (1982) Boström et al. (1988) & Jeppesen, E et al. (2007). There is some uncertainty to whether internal P loading took place between the 2nd of September and autumn turnover, although the redox potential did most likely not become low enough considering that Lake Østensjøvannet is susceptible to mixing.

Mineralization processes are present in Lake Årungen as well, by increasing mineralization products such as increased silicon, phosphate, ammonium, TOC and DOC concentration (figure 28A, 28B, 33A and 34a & B). The conductivity and alkalinity (figure 37A & 38) became elevated as well. The concentration of total phosphorus is however increased to over 600 µg L⁻¹. The TP concentrations corresponds to results seen in e.g. Norwegian Lake Kolbotnvannet, which have had returning episodes of internal loading (Haande et al. 2013). Nitrate concentrations becomes exceedingly low (0.15 mg L⁻¹) at the last sampling date (1st of October), which suggest that the buffering capacity of the nitrate supply has been reached. Boström et al (1988), showed that nitrate concentration less than 0.5 mg L⁻¹ constitutes a risk for iron reduction. This is further

reflected by the progressively lower ratio between $\text{NO}_3\text{-N}$ and TN in the bottom samples throughout the season (fig 32). The increase of iron and reduction of sulfur over the sediments, give additional indications of Fe-P dissolution (figure 30B & 36A). Since indications of internal loading were seen late in the season, the impact on primary production was minor. Some of the released phosphate will most likely co-precipitate with ironhydroxides again during autumn turnover, provided that there has been no substantial ligand exchange with sulfide (Driscoll et al. 1993; Holmer & Storkholm 2001; Kleeberg & Schubert 2000; Phillips et al. 1994).

The sediments of lakes can vary in their sensitivity to redox conditions (Kisand & Noges 2003) due to differences in phosphorus fractions. Accordingly the sediments in Lake Østensjøvannet could be more resistant to P release under anoxic conditions. Since the Non-apatite Inorganic P (NAIP) is assumed to be the fraction that can be released under anoxic conditions and high pH levels

(Håkansson & Jansson 1983), sediment cores from Lake Østensjøvannet could provide more information about the phosphorus composition and redox sensitivity of the sediments. It is difficult to determine whether or not internal P loading occurred from water samples alone; however the results give some good indications to the dominating processes in the two lakes.

6.6.2 pH mediated phosphorus release

The mean pH values is assumed to have been under 8.4 on a daily basis, with an exception for July 9th, based on the 8 vertical series that was sampled during the growing season (figure 14). In order for the phosphorus release at high pH would be significant, based on previous studies, the mean pH in the water on a daily basis should be higher than 8,4 (Erlandsen & Grøterud 1980). Bostrom & Pettersson (1982) showed that anaerobic P release were high in the Swedish lake Erken at a pH interval between 8 -10, indicating that pH over 8 may constitute a risk for P release.

The stratification regime further limits water mixing; accordingly, the high pH values of the surface layer are unlikely to affect the sediments at the deepest parts in the lake, where P accumulation is considered to be highest. However, Koski-Vahala & Hartikainen (2001), showed from lab experiments, that high pH in the surface water linked with intense resuspension could increase the risk of P loading. Accordingly the impact on high pH at the surface water cannot be disregarded.

The results from this study give a general picture of the pH conditions of the growing season, however there is still the possibility of much higher pH values occurring in-between the sampling dates. As seen in Appendix 1, high resolution measurements of pH from Lake Årungen shows that pH can have large daily variations. Although, assuming the results are representative it would seem that the release of P as a result of high pH is not the main concern in Lake Østensjøvannet given 2013 conditions.

6.6.3 Wind re-suspension of particles

Internal loading by wind suspension may also occur in Lake Østensjøvannet, since it is a shallow and wind exposed lake. However the sediments in shallower areas may be less loaded with phosphorus. The pattern of sedimentation of particles in lakes are; coarser material being deposited in the littoral zone, while the sediments below the deeper part of the lake is rich in fine clay material (Riise et al. 2010). Since particulate P is more associated with clay material, the latter part will be more P loaded, which limits the effect of the wind. Further, the study of 3 finish shallow lakes in an agricultural setting done by (Krogerus & Ekholm 2003) showed that most of the wind resuspended P were unavailable to algae.

6.6.4 Bioturbation

Bioturbation has not been the focus of this study, however it is worth mentioning that the low oxygen supply in the hypolimnion of Lake Østensjøvannet coupled with high ammonium concentration would enclose large volumes of the lake for fish to dwell in. This observation is consistent with the observations made by Haugen et al (2012), which found that reduced oxygen levels were occurring under a depth of 5.5m in 2012. The summer of 2012 was quite wet and cold compared to 2013, which suggest that thermal stratification in the lake are more common than previous assumptions made by Grøterud & Haaland (2007). Since the study has not included the littoral zones, bioturbation from those areas could contribute to the overall re-suspension, although the highest phosphorus reserves are assumed to be located in the sediments of the profundal zone.

6.7 Phosphorus transport and retention processes in Bølstadbekken

The results from the upstream and downstream water samples indicate retention of particles and nutrients within the stream (figure 49 – 55). Studies done on phosphorus retention in rivers (Jarvie et al. 2005; Mainstone & Parr 2002) has shown that phosphorus are retained in rivers during summer months, however it seems to be some variation between rivers. Banazuk & Wysocka-Czubaszek (2005) showed the opposite relationship in the study on the Narew River, in NE Poland, where P was retained during spring flood and released during the summer months. The observed reduction in TP concentration, between the upstream and downstream location seems to be highly connected with the suspended solid load. Since the sediment trapping is more efficient during low flow (Reddy et al. 1999), the particulate P fraction will be retained, explaining the reduction in TP (figure 50 A). Phosphate is however increased downstream during low flow as seen in figure 50 B. Selig & Schlunbaum (2002) recognized that anaerobic and aerobic release of P can occur in impoundment sections of rivers and processes at the sediment-water interface are similar to those in shallow, non-stratified lakes. Consequently; anaerobic release of P could occur during low velocities in the river. If the sediments are saturated there could be a release of phosphate due to equilibrium reactions, which could further explain the increase in phosphate at the downstream location. The ability of a river to retain phosphorus depends on the characteristics of the river and the river sediments. Sequences of high velocities tend to have coarser bed characteristics which is unfavorable to plant growth, thus, phosphate goes unutilized down that river reach. Sequences of slow flowing water, however, especially sluggish river reaches are ideal for the accumulation and potential biota uptake of P. These sluggish river reaches have the ability to accumulate P and therefor effectively retain nutrients, but it can also serve as a source. Banazuk & Wysocka-Czubaszek (2005) showed that high levels of particulate P occurred during sudden (even small) increase of flow, which lead to scouring and flushing of P rich sediments. Elevated TP concentrations were observed in the beginning of May and on the onset of autumn elevated flows (figure 50 A), which is consistent with observation by Banazuk & Wysocka-Czubaszek (2005). Their results also showed a net gain of phosphate to the water column during low flow, which is consistent with the observations from Bølstadbekken. It is however pointed out that this relationship may be highly variable from stream to stream. Erlandsen & Grøterud (1980) emphasize that there are some self-purifying processes in Bølstadbekken before it reaches Lake Årungen, but concludes that these effects are of little

importance on a yearly basis. The retention in Bølstadbekken can however be crucial for the water quality in Lake Årungen if they are effective in the growing season. Results however indicates a net gain of phosphate to the water masses in the stream before entering Lake Årungen, during low flow, which can catalyze phytoplankton growth during sensitive time periods.

When monitoring rivers there are problems related to regular temporal sampling of such kind often seen in lakes. River systems are much more governed by “events”(Jones 2001), because much of the total transport in a river can happen during storm events. Therefore the sampling frequency in lakes does not transfer well to rivers. As seen by figure 6, the discharge can vary substantially, from day to day. More frequent sampling and high resolution discharge data from the stream can provide more accurate estimates of transport and retention of nutrients.

The groundwater input could also have had an effect on the changes in concentration along the river reach. As seen from the results many parameters become reduced from the upstream location to the downstream location. The input of groundwater may dilute the stream water with water of a different chemical character. Appendix 6 lists cations found at the upstream and downstream locations. As seen from figure 56, Na, Mg, K, and Ca were increased from the upstream to downstream location in the later part of the season, and the conductivity is highly increased at the downstream location as seen in figure 67. This may affect the interpretation of general retention in the stream since some of the changes in chemical composition of the water can be a result of dilution from water containing less suspended solids, organics and phosphorus.

Most of the previous studies on river sediments and retention capacities have been conducted on larger rivers; consequently comparisons are done on the assumptions that the processes are transferable to small river systems. Further investigations of the river sediments and P fractionation are necessary for a more substantial conclusion.

7 Conclusion

Differences in lake depth and retention time greatly affect the thermal stratification, mixing regime and nutrient cycling. Due to differences in depth, the mixing barrier is more efficient in Lake Årungen than in Lake Østensjøvannet, which is shallower and more subjected to mixing by wind force. Poorer insolation, due to a higher load of suspended particles in Lake Østensjøvannet also contributed to a less stable stratification regime. As a result, there was more exchange between the epilimnion and hypolimnion and the vertical gradients of chemical substances in the water column became less sharp. The suspended load was determining for the light conditions in both lakes, although water color (humic substances) had a large impact on light penetration in Lake Årungen. The water in Lake Østensjøvannet was more turbid throughout the season due to a higher flushing rate and wind resuspension. Consequently the seechi depth did not increase progressively as in Lake Årungen.

The lake metabolism was higher in Lake Østensjøvannet and intensified mineralization of organic material in the hypolimnion was seen early in the season. Due to the shallower depth, organic matter accumulated over the sediments. Consequently oxygen deficits occurred earlier than in Lake Årungen. The *chl a* concentration, DO saturation and organic suspended solids, indicated a higher production compared to Lake Årungen, however other parameters such as pH and phytoplankton biomass suggested a higher growth in Lake Årungen. The discrepancies are best explained by observations of autotrophic picoplankton in Lake Østensjøvannet, which influences the *chl a*/biomass ratio, since these organisms are not detected by microscopic counts. The limiting factor for phytoplankton during the first part of the season was shallow mixing regimes and low insolation in both lakes. In Lake Årungen nutrient became limiting during the second part of the season, especially by silicon deficits. Conversely nutrient limitation is not supported in Lake Østensjøvannet. Other phytoplankton taxa was most likely subjected to competition of light by the picoplankton population. Further studies are needed to understand the function and interaction of these organisms. The results did not support that the phytoplankton taxa in Lake Årungen was connected with the taxa in Lake Østensjøvannet, by cell migration. Cell import can however be more significant in years with higher precipitation, due to increased flushing rates.

The retention capacity in Lake Østensjøvannet is greater for phosphorus than for nitrogen since the former is more associated with particles. Biological assimilation of nitrogen and denitrification processes in the lake will however reduce downstream transport. Phosphorus is retained by biological uptake and particle sedimentation. Internal phosphorus loading in Lake Østensjøvannet is not supported by the findings of this study, despite the long period of observed anoxia at the sediment-water interface. The overall high concentration of nitrate in the hypolimnion provided an effective buffer by stabilizing the redox conditions. Further studies on the sediment chemistry of the lake, could be useful in order to evaluate the risk internal phosphorus loading in the lake.

There was a net loss in particles and TP in the connecting stream over the season, which shows signs of retention processes in the stream. However during periods of low discharge the phosphate concentration increased downstream which could catalyze phytoplankton growth in Lake Årungen.

Lake Østensjøvannet functions as a buffer of incoming nutrient loads from the catchment, both by retention and transformation of nutrients. The retention capacity in the lake depends largely on the hydraulic retention time. The lake is a phosphorus sink, provided no substantial internal loading. Considering the long period of stagnation in 2013, the lake is less subjected to internal phosphorus loading now, than previous studies have shown.

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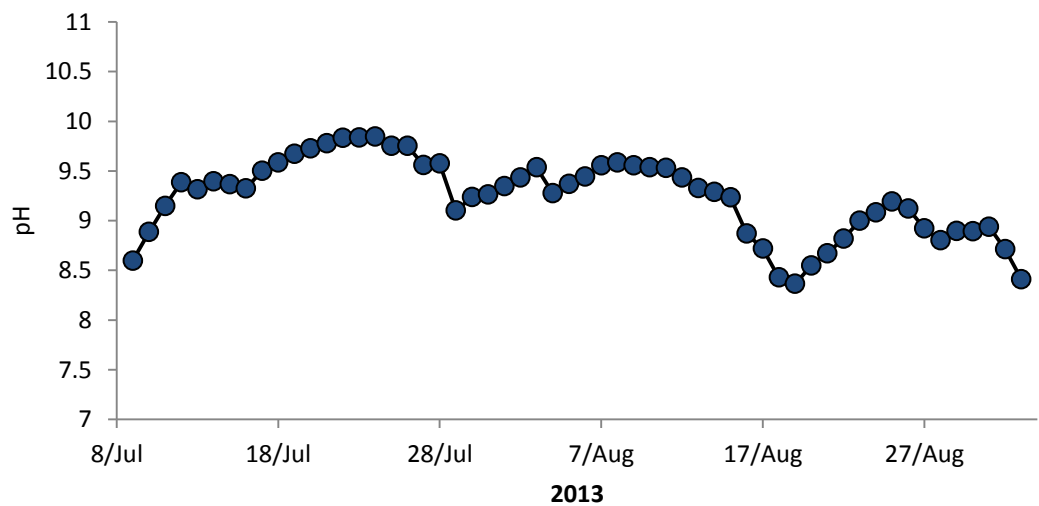
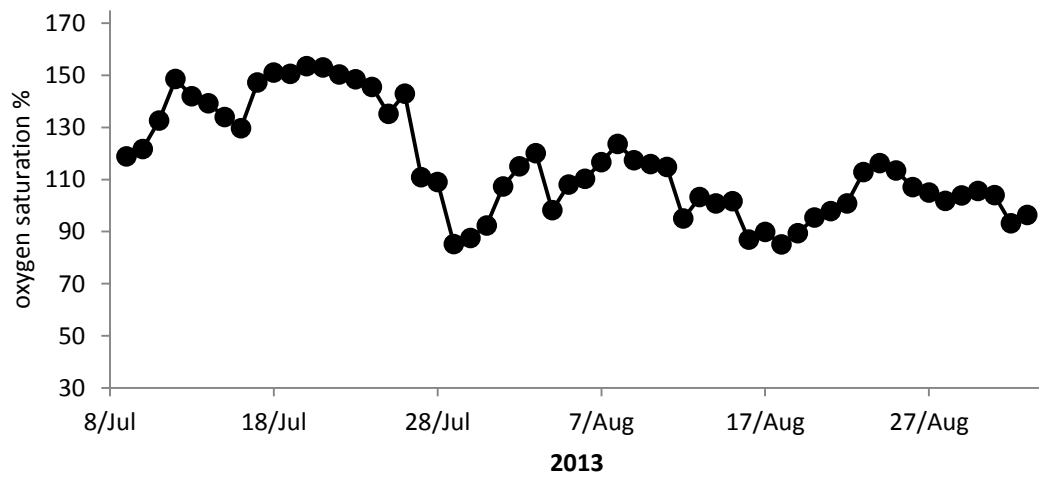
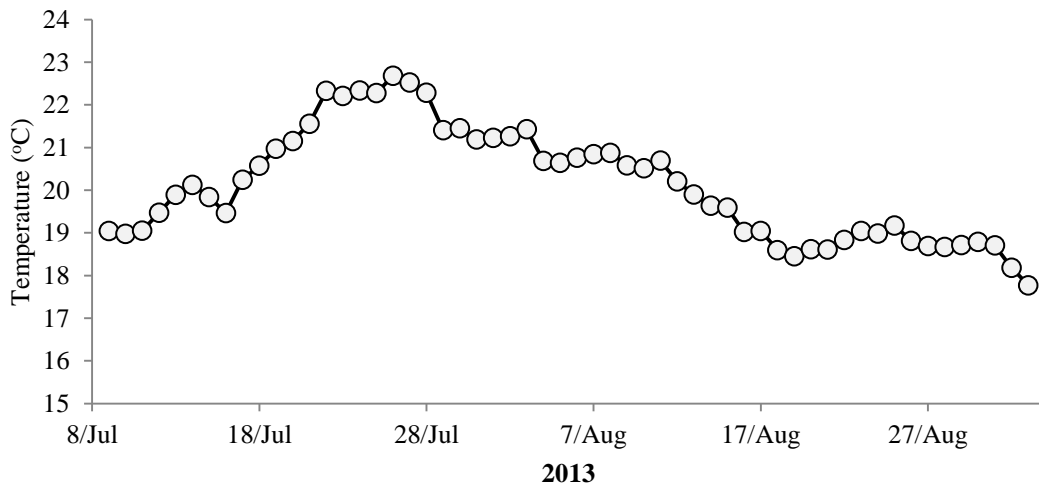
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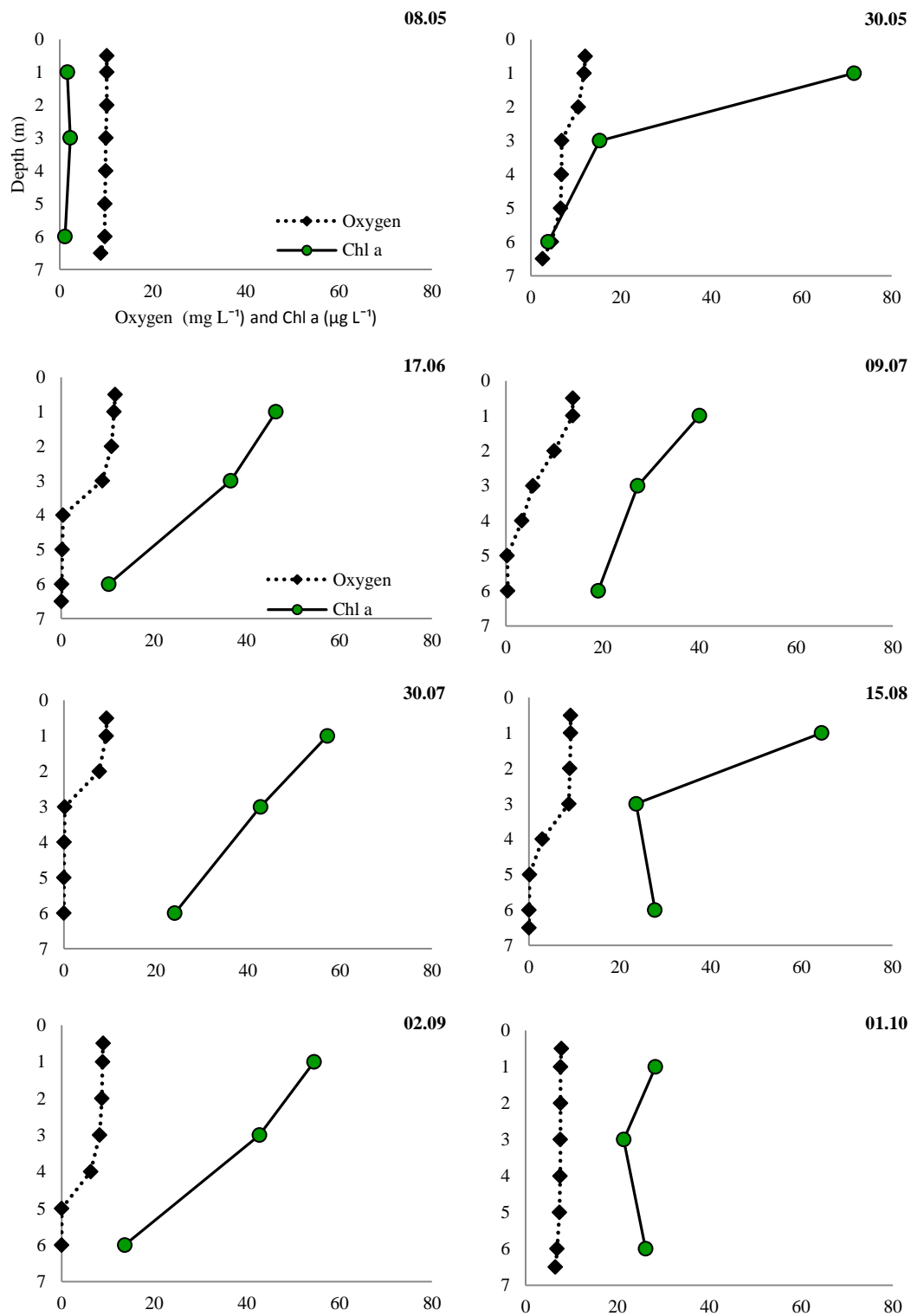
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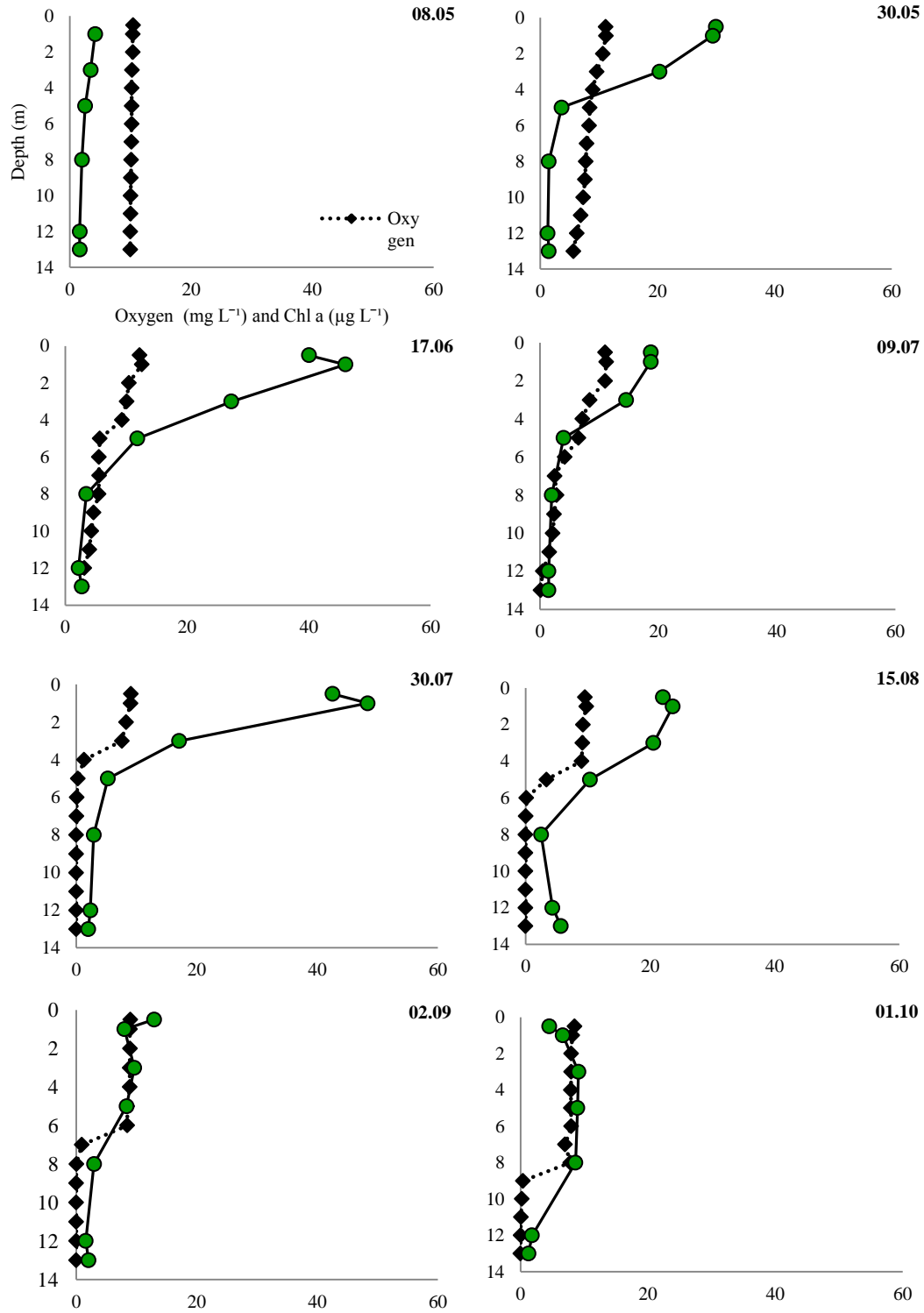
Appendix 1 high resolution data of temperature, oxygen saturation and pH in Lake Årungen



Appendix 2 Vertical profiles of oxygen and chl a in Lake Østensjøvannet and Lake Årungen

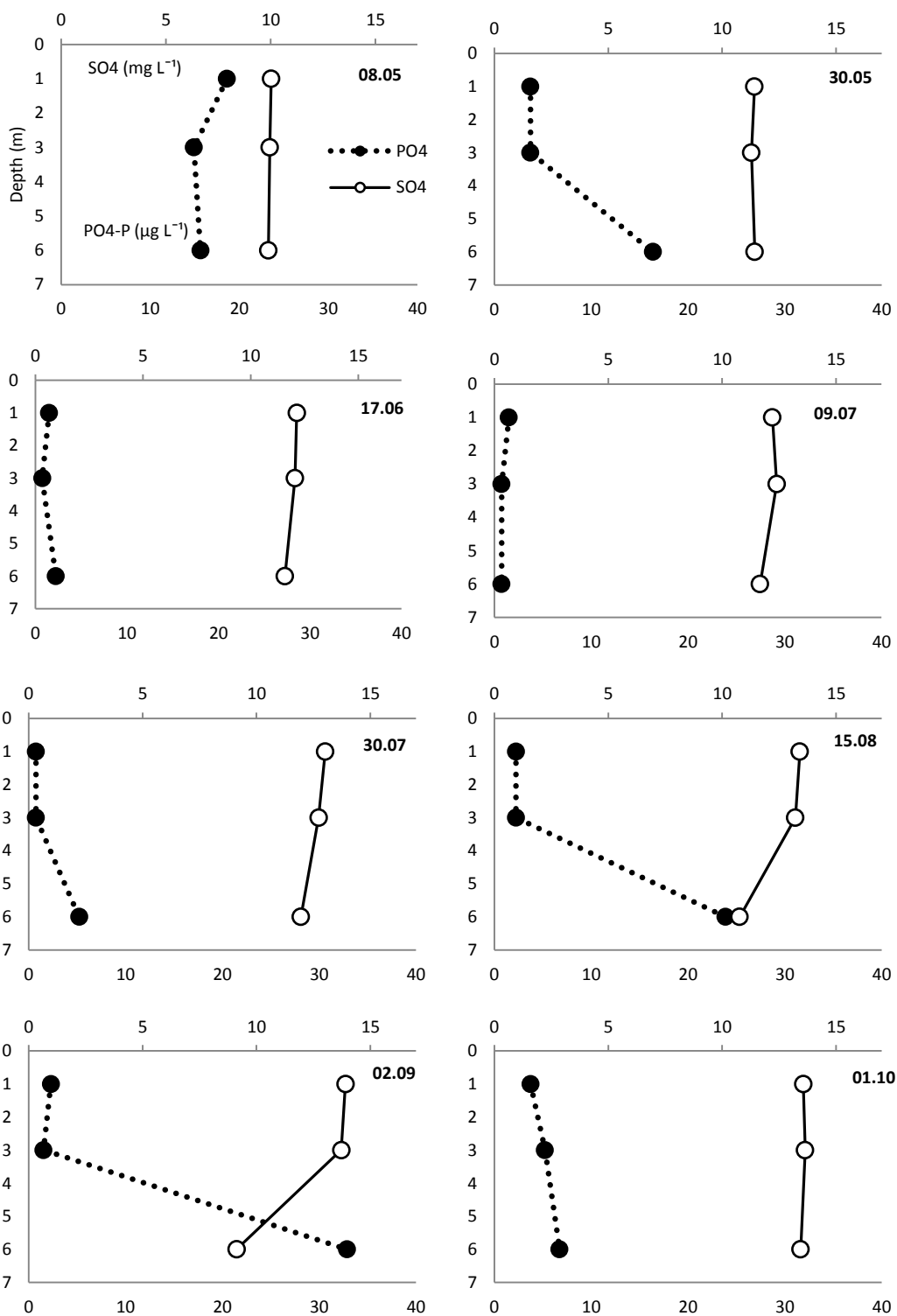


Vertical series of oxygen (black markers) and chl a (green markers) in Østensjøvannet during the period of May 8th and October 1st 2013.

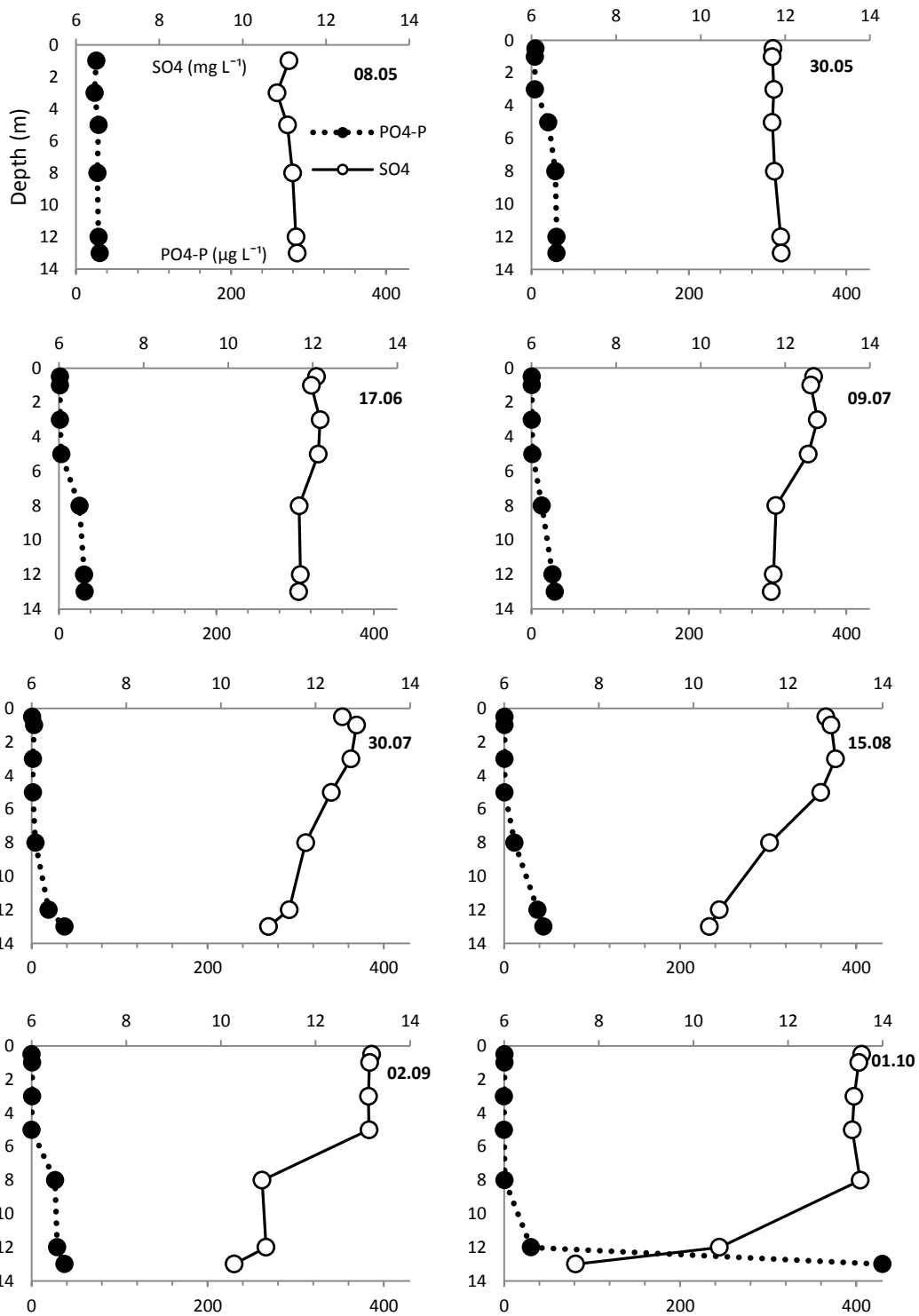


Vertical series of oxygen (black markers) and chl*a* (green markers) in Årungen during the period of May 8th and October 1st 2013.

Appendix 3 Vertical profiles of sulfate and phosphate in Lake Østensjøvannet and Årungen

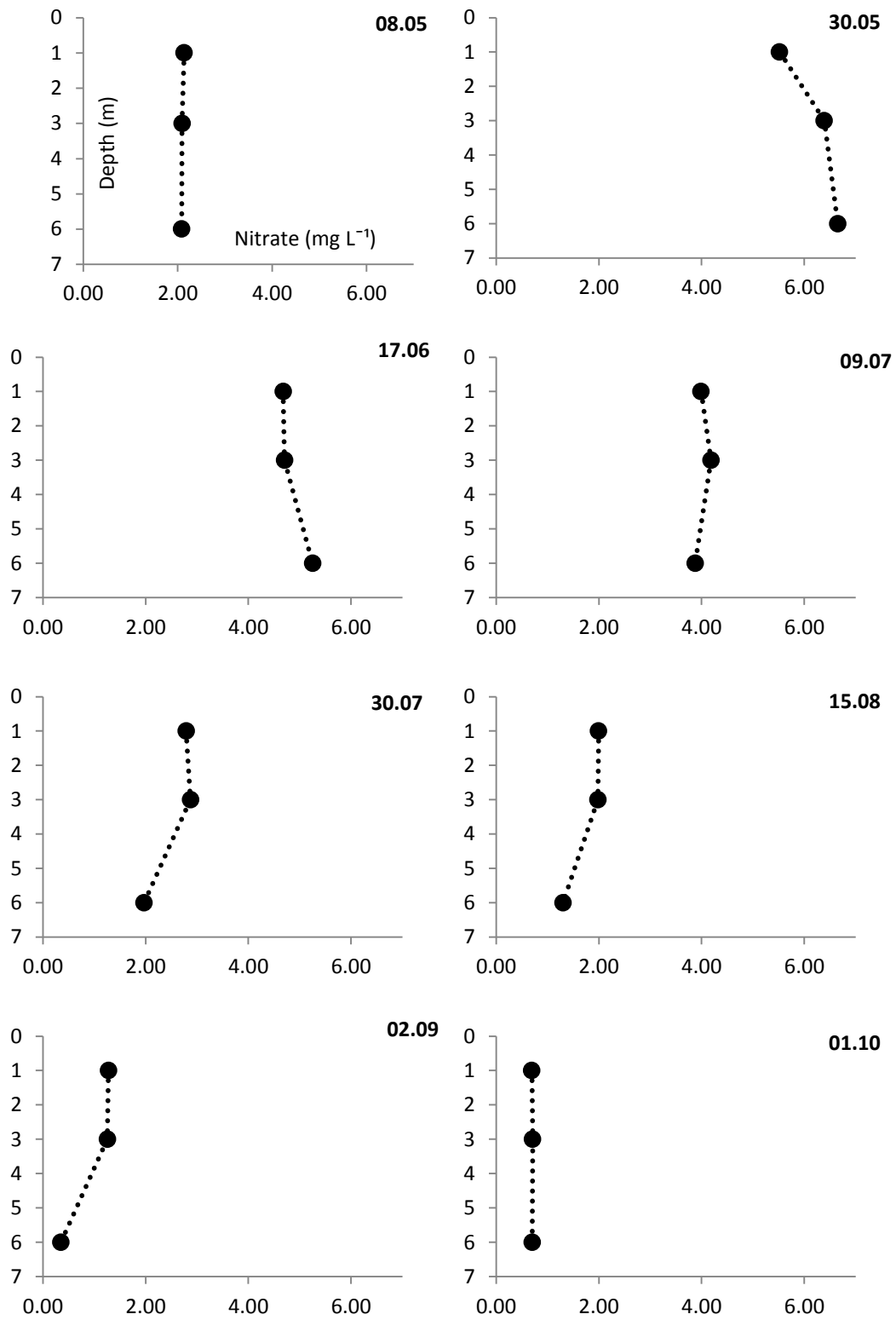


Vertical depth profiles with SO_4^{2-} in mg L^{-1} (open markers) and PO_4^{3-} in $\mu\text{g L}^{-1}$ (black markers) in Lake Østensjøvannet.

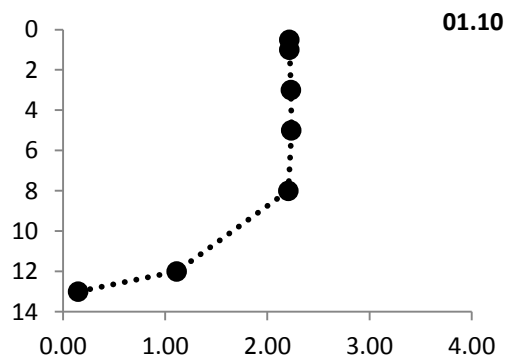
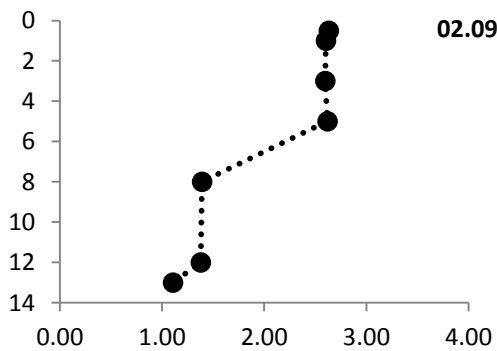
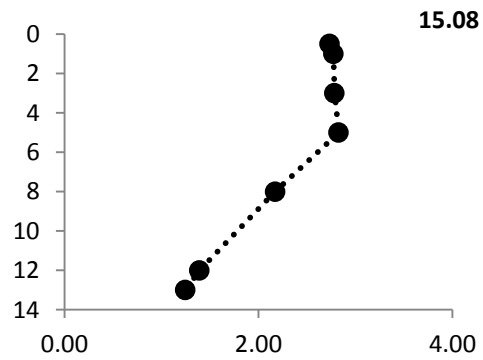
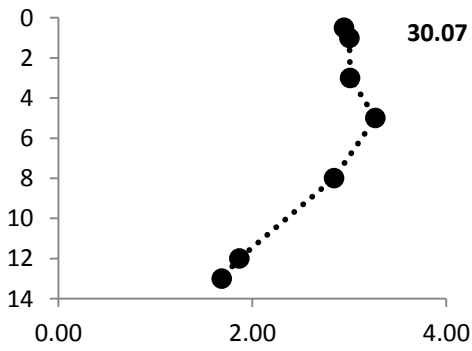
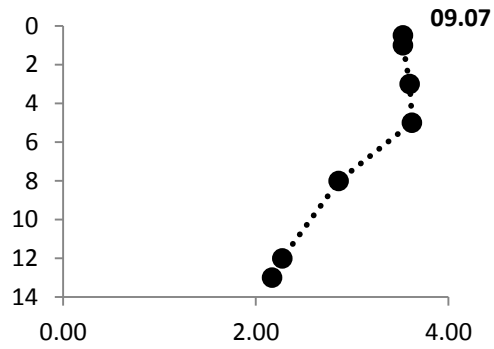
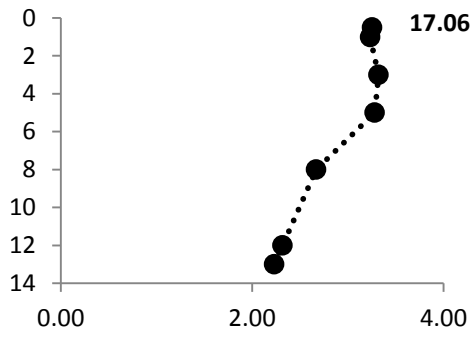
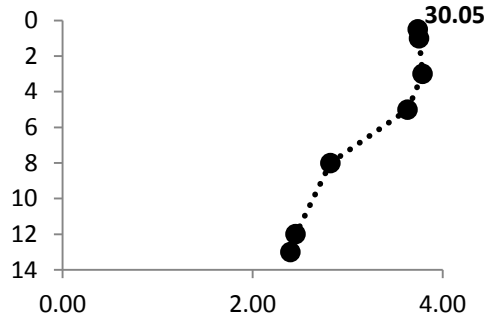
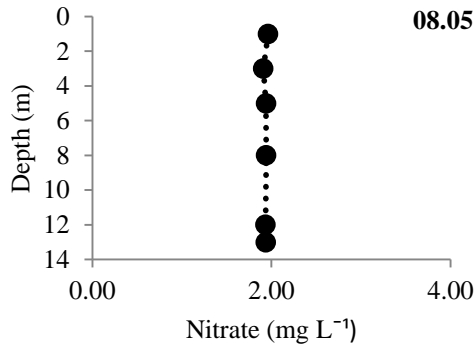


Vertical depth profiles with SO_4^{2-} in mg L^{-1} (open markers) and PO_4^{3-} in $\mu\text{g L}^{-1}$ (black markers) in Lake Årungen.

Appendix 4 Vertical nitrate profiles in Lake Østensjøvannet and Lake Årungen

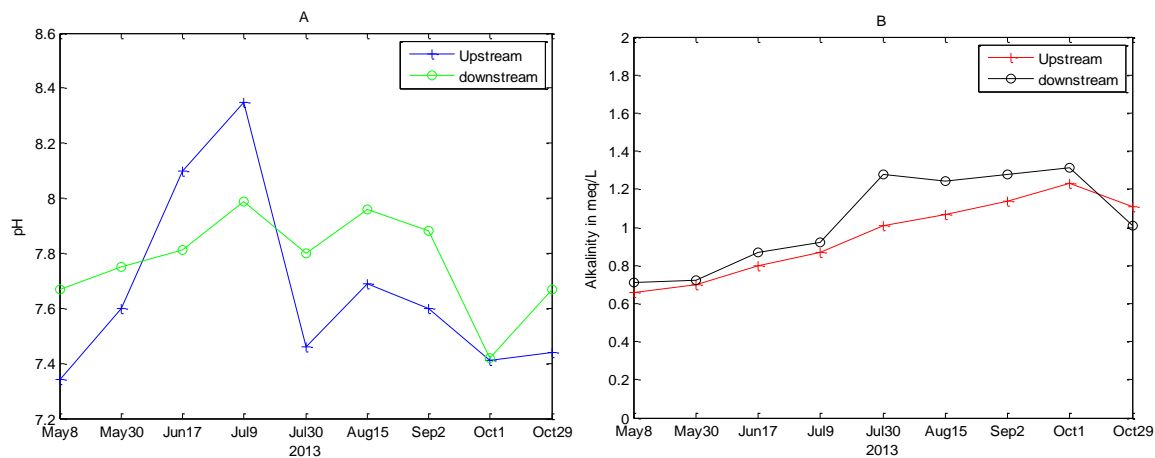


Vertical nitrate series in Lake Østensjøvannet

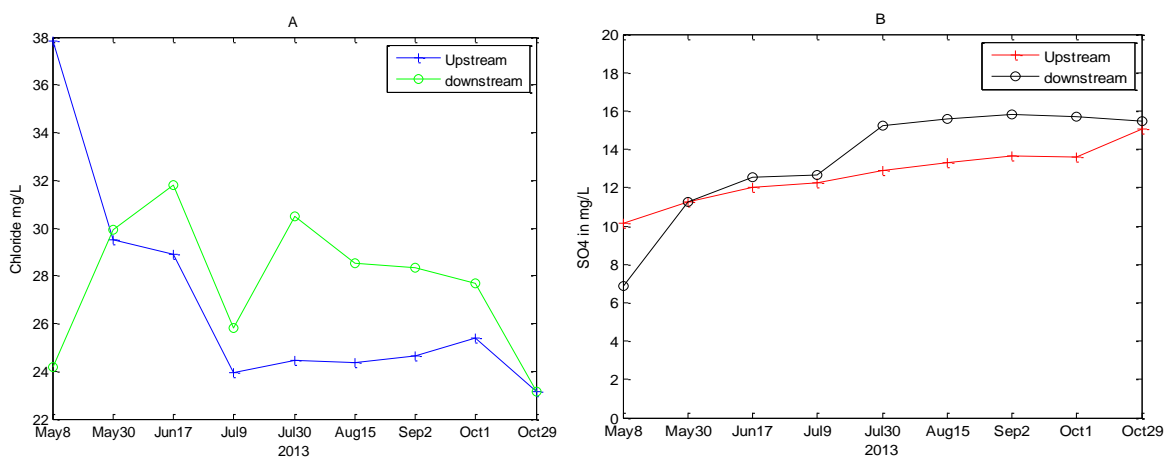


Vertical nitrate series in Lake Årungen

Appendix 5 Series of pH, alkalinity, chloride and sulfate in Bølstadbekken



A) pH and B) alkalinity (meq L^{-1}) at the upstream and downstream location in Bølstadbekken, from 8th of May until 1st of Oct 2013.



A) Chloride (mg L^{-1}) and B) sulfate (mg L^{-1}) at the upstream and downstream location in Bølstadbekken, from 8th of May until 1st of Oct 2013.

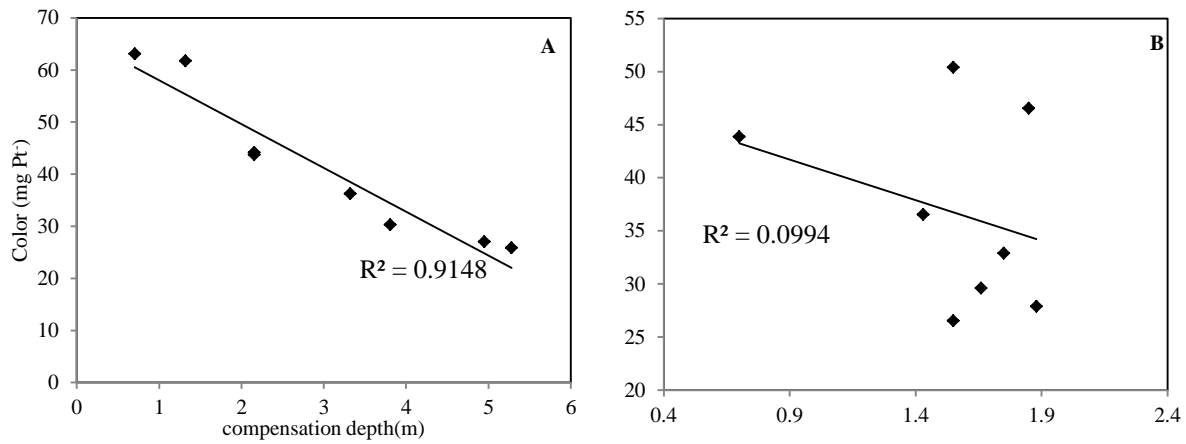
Appendix 6 Cations in Østensjøvannet, Bølstadbekken and Årungen

	Na	Mg	Al	Si	P	K	Ca	Mn	Fe	Cu
	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	ug/L
Østensjøvannet										
8. mai										
1m	23	4,1	2,2	7,3	0,11	4,3	17	0,14	2,6	4,9
6m	23	4,2	2,5	8,0	0,13	4,4	17	0,16	2,9	5,2
30. maj										
1m	18	4,7	0,57	4,0	0,084	4,3	21	0,035	0,49	3,6
6m	17	5,1	1,6	6,9	0,087	4,9	21	0,11	1,5	4,3
17. juni										
1m	18	4,8	0,27	3,1	0,068	4,2	22	0,048	0,31	3,7
6m	18	4,9	0,78	5,6	0,048	4,7	21	0,23	0,76	3,7
9. juli										
1m	15	4,7	0,24	3,4	0,060	4,3	21	0,023	0,26	3,6
6m	18	5,0	1,1	6,2	0,091	4,9	22	0,57	1,1	3,8
30. juli										
1m	16	4,8	0,25	3,4	0,075	4,4	22	0,062	0,37	3,6
6m	18	5,2	1,1	6,7	0,10	4,9	24	0,93	1,5	3,7
15. aug										
1m	16	4,8	0,17	3,3	0,072	4,3	22	0,052	0,26	3,5
6m	18	5,3	0,50	5,7	0,12	4,9	25	1,0	1,3	3,2
2. sep										
1m	16	4,8	0,20	3,4	0,080	4,5	23	0,096	0,36	3,5
6m	18	5,4	0,19	4,9	0,19	4,8	26	2,2	2,5	2,6
1. oct										
1m	17	4,9	0,25	3,8	0,080	4,6	23	0,15	0,63	3,1
6m	17	5,0	0,38	4,1	0,085	4,6	23	0,21	0,81	2,8

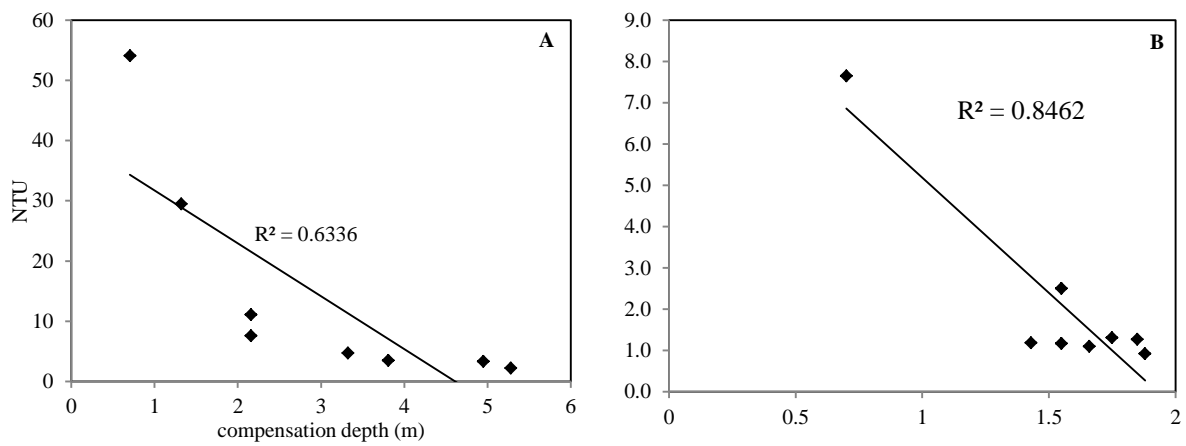
	Na	Mg	Al	Si	P	K	Ca	Mn	Fe	Cu
	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	ug/L
Bølstadbekken										
8 may										
output	22	4,0	2,0	6,9	0,11	4,4	17	0,13	2,1	4,4
input	21	4,3	1,7	6,7	0,087	4,0	18	0,089	2,0	3,8
30 may										
output	18	4,6	0,47	3,9	0,051	4,4	20	0,023	0,41	3,1
input	18	4,6	0,62	4,4	0,052	4,1	20	0,041	0,73	3,1
17. juni										
output	18	4,7	0,11	2,6	0,041	4,2	21	0,017	0,13	2,8
input	20	4,9	0,33	3,4	0,050	4,2	22	0,049	0,49	2,9
9. july										
output	16	4,6	0,26	3,5	0,049	4,3	21	0,026	0,28	3,1
input	17	4,9	0,24	3,6	0,043	4,3	22	0,030	0,34	3,1
30. july										
output	15	4,8	0,17	3,1	0,067	4,2	22	0,073	0,27	3,0
input	19	5,8	0,18	3,4	0,064	5,0	27	0,058	0,35	2,8
15. aug										
output	16	4,9	0,14	3,2	0,066	4,2	22	0,046	0,23	2,9
input	19	5,5	0,13	3,2	0,048	4,8	26	0,037	0,26	2,7
2. sep										
output	16	4,8	0,14	3,3	0,064	4,4	22	0,083	0,26	2,7
input	19	5,4	0,050	3,1	0,040	4,9	26	0,030	0,15	2,4
1. oct										
output	17	4,9	0,16	3,6	0,080	4,6	23	0,12	0,55	2,5
input	20	5,6	0,046	3,4	0,039	4,8	27	0,032	0,21	2,1

	Na	Mg	Al	Si	P	K	Ca	Mn	Fe	Cu
	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	ug/L
Ärungen										
08.may										
1m	19	4,5	1,8	6,4	0,11	4,4	19	0,14	2,1	5,1
13m	20	4,5	1,8	6,6	0,11	4,4	20	0,14	2,2	4,2
30. may										
1m	19	4,6	0,86	4,8	0,077	4,3	20	0,022	0,84	3,7
13m	20	4,5	1,3	5,7	0,089	4,3	20	0,086	1,5	3,7
17. jun										
1m	20	4,5	0,12	2,5	0,045	4,3	21	0,0010	0,089	3,7
13m	21	4,3	1,0	5,5	0,084	4,4	20	0,16	1,1	3,5
9. july										
1m	19	4,6	0,29	3,2	0,041	4,3	22	0,012	0,27	3,3
13m	21	4,4	0,66	4,9	0,069	4,3	20	0,16	0,63	3,2
30.july										
1m	19	4,7	0,13	2,2	0,063	4,4	22	0,024	0,14	3,1
13m	21	4,5	0,57	4,9	0,11	4,2	22	0,52	1,0	3,0
15. aug										
1m	20	4,8	0,072	0,18	0,039	4,3	23	0,033	0,088	2,9
13m	21	4,5	0,54	5,1	0,19	4,4	22	0,88	1,6	2,8
2. sep										
1m	21	4,8	0,036	0,20	0,025	4,3	22	0,016	0,048	2,9
13m	21	4,5	0,40	4,7	0,13	4,3	22	1,1	1,1	2,7
1 oct										
1m	21	4,7	0,051	0,70	0,022	4,3	22	0,055	0,080	2,9
13m	22	4,7	0,30	5,2	0,76	4,5	24	2,4	5,7	8,1

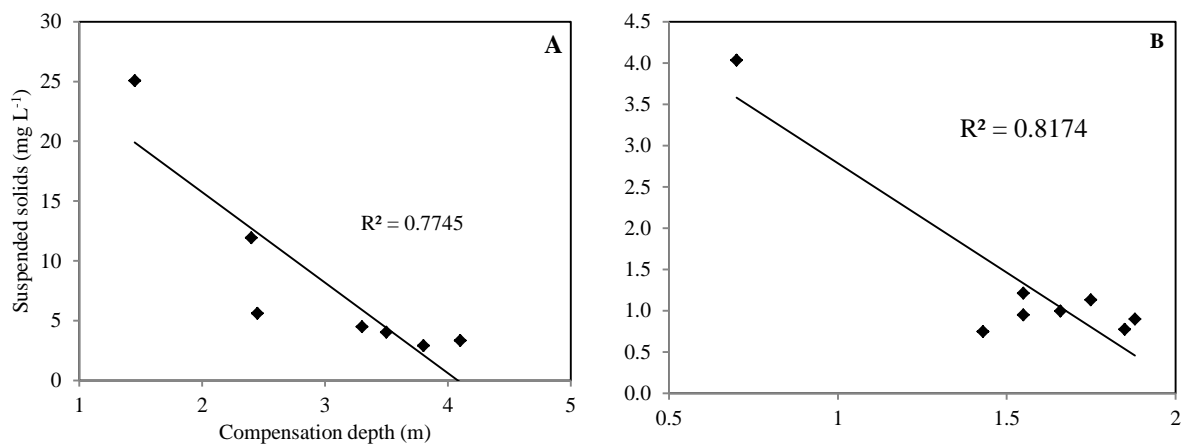
Appendix 7 Light attenuating variables in Lake Østensjøvannet and Lake Årungen



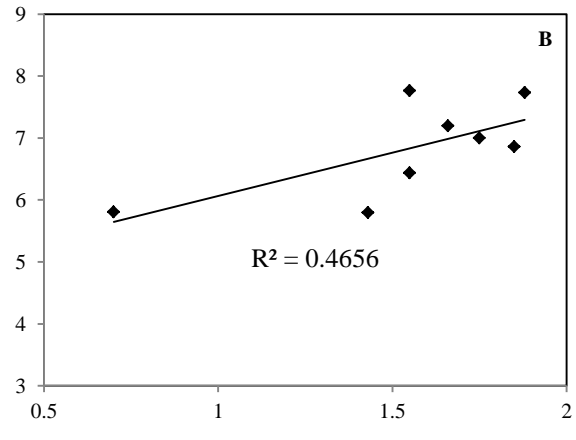
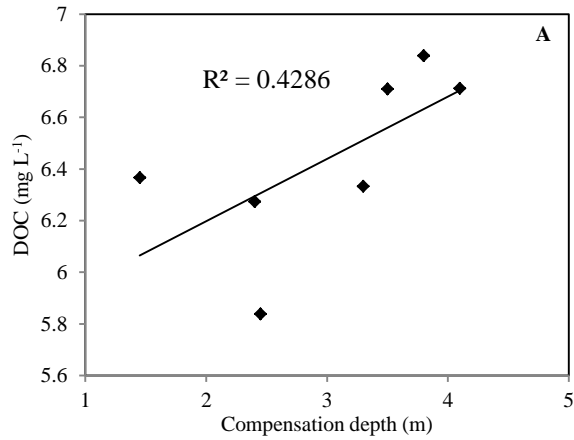
Correlation between color and transparency in A) Årungen and B) Østensjøvannet



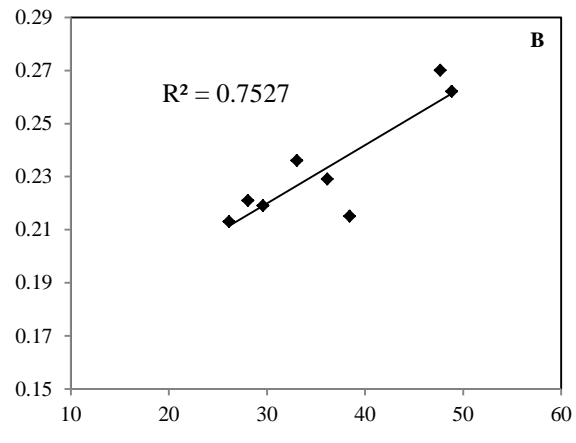
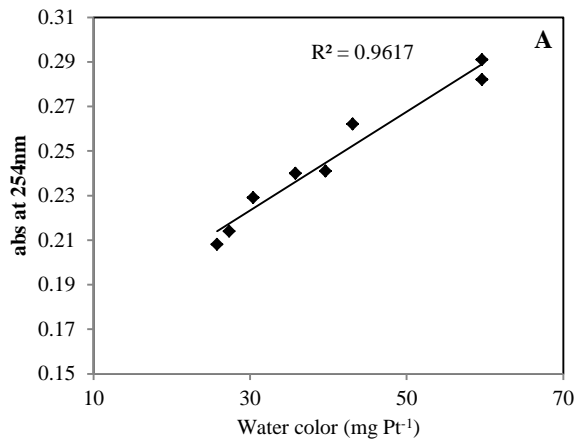
Correlation between turbidity and transparency in A) Årungen and B) Østensjøvannet



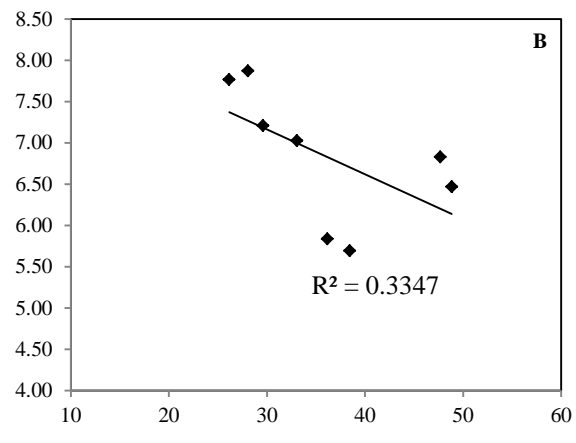
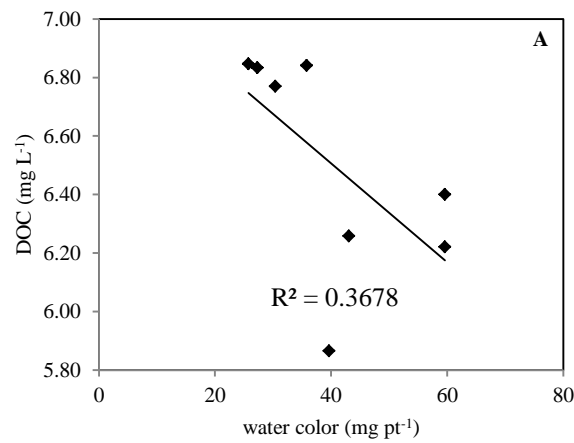
Correlation between suspended solids and transparency in A) Årungen and B) Østensjøvannet



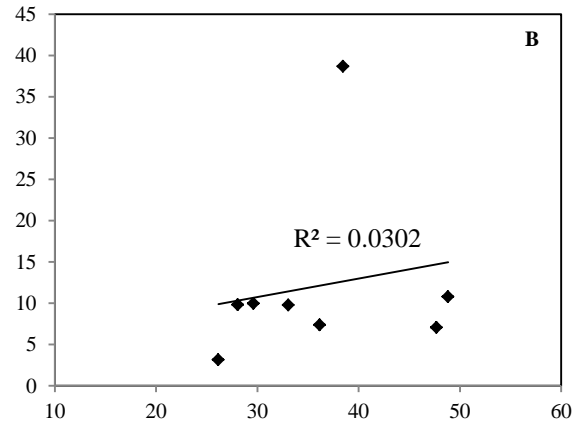
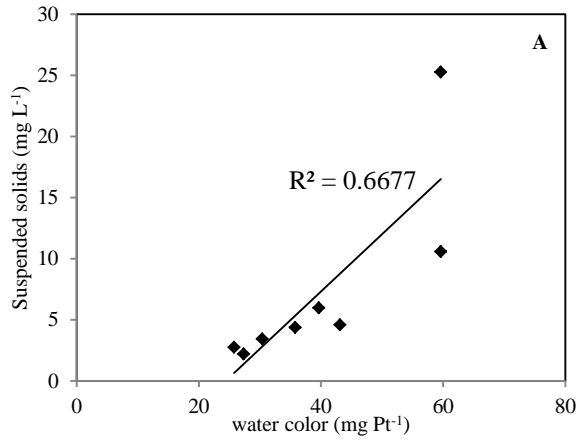
Correlation between DOC and Transparency in A) Årungen and B) Østensjøvannet



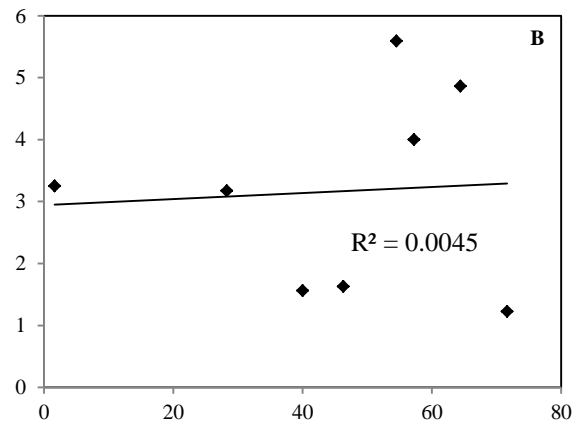
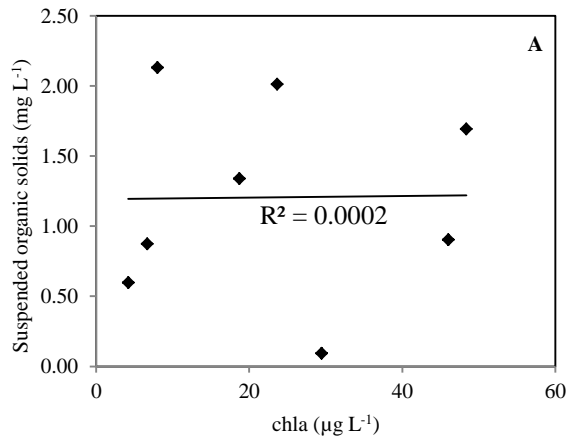
Correlation between water color and abs at 254nm in A) Årungen and B) Østensjøvannet



Correlation between water color and DOC in A) Årungen and B) Østensjøvannet



Correlation between water color and suspended solids in A) Årungen and B) Østensjøvannet



Correlation between chl a and organic suspended solids in A) Årungen and B) Østenjøvannet



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