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Water chemical effects of forest-fire on lakes

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Abstract

A fire occurred in Mykland, Froland municipality, southern Norway in June 2008. This study is analyzing long and short-term changes in water chemistry in 6 lakes within the burned catchments and 3 outside the burned area during 3 post-fire years. It is known that forest fires can lead to temporary acidification. The area has previous been acidified by long range transported air pollutants, this study evaluate if the fire caused chemical acidification effects. Other studies have observed increases of both cations and anions in surface waters after fire events. The study shows that the lakes were heavily acidified, due to larger increase of anions relative to cations after a large rain event two months after the fire. Accordingly the acid neutralizing capacity (ANC) of the lakes dropped significantly. The spread of dust and ashes after the fire might have caused changes in the reference lakes outside the fire area.

The fire caused elevated concentration in K^+ in burned lakes which likely is linked to its high abundance in biological material; the concentration was near pre-fire levels in the end of the third post-fire year. TOC decreased substantially after the fire, and then increased during all the three post-fire years. The concentration of SO_4^{2-} had a large increase during the first post-fire year in the lakes with burned catchments, but showed significant decreases in all lakes during the next coming years, which might be linked to the general decrease of SO_4^{2-} in this region, or to atmospheric fallout of ash from the burned catchments. It cannot be ruled out that ANC might be higher in the third post-fire year than before the fire.

1 Content

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Bø, 2012-07-16

Rebecka Karlsson

2 Abbreviations

- AD-test –Anderson Darling Normality Test

- Ala –Monomeric Aluminium

- Alc –Aluminium Colloids and Particles

- Ali –Inorganic Aluminium

- Alo –Organic Aluminium

- ANC & ANC_1 –Acid Neutralizing Capacity

- ANC_{oaa} & ANC_2 –Organic Acid Adjusted ANC

- IC – Ion Chromatograph

- TOC – Total Organic Carbon

- T-Test –Paired T-test

- WFD- Water Framework Directive

- W-test –Wilcoxon matched pair test

3 Introduction

In June 2008, a forest fire occurred in Mykland, Froland municipality, southern Norway. Little is known about physical, chemical and biological effects of wildfires on inland waters. As climate change is expected to cause increased frequency of wildfires (Flanningan et al. 2005), more knowledge is required in order to evaluate its effects on the water quality. Accordingly, this thesis is investigating changes in lake water chemistry in 6 wildfire impacted lakes and 3 nearby lakes.

The Water Framework Directive (WFD, EC2000/60) is an important legislative tool designed to improve and protect the water quality in Europe. WFD has the general environmental quality objective "good water status" for all surface waters by 2015. The objective "good water status" means both "good ecological status and "good chemical status." Acidification is one of the factors that are used in the classification of the ecological status of lakes in accordance to Annex V in WFD. Combustion of fossil fuels has been the main source to water acidification, but it is known that fires also may lead to temporary acidification of inland water (Bayley et al. 1992, Eriksson et al. 2003).

Acidification of water leads to decreased pH levels and increased unstable forms of aluminium. High concentrations of inorganic Al can cause damages on the respiratory systems on fish and benthic animals (Wetzel 2001). Total organic carbon (TOC) plays an important role for in which degree low pH can affect organisms since it can function as a pH buffer against strong acid anions as well as detoxifying inorganic Al-forms by formation of organic Al-complexes. Acid neutralizing capacity (ANC) is a parameter that is proven to be a good measure of acidification status of lakes (Reuss and Johnson 1986), as ANC better than pH and/or Al explain biological status in acidified lakes (Bulger et al. 1993, Lien et al. 1996, Lydersen et al. 2004)**. As this thesis primarily is focusing on chemical acidification effects of wildfires on inland water, ANC, pH, TOC and concentration and distribution of Al-species are the most essential parameters evaluated.

3.1 Objective

- Study long and short-term changes in macrochemistry with primarly focus on pH, organic and inorganic aluminium (acute toxic forms), ANC, TOC, colour and UV in inland waters inside and outside a forest fired impacted area.
- Comparison of analytical data due to change in analytical method (Appendix 1).

4 Background

4.1 Changed pattern of forest fires

Wild fires are natural processes in ecosystems and thus an important factor of landscape processes and features. Wild fire affects forest ecosystems by creating space and release nutrients and by reducing trees and organisms. Human's started to use fires for controlling and modifying nature long back in time. The development of agriculture increased the human induced fires. During the last 100 years human has induced forest suppression efforts and accordingly decreased the fire occurrence (Cerdá and Robichaud 2009). The annually burnt areas dropped significantly between 1860 and 1890 in most of the boreal forest in Fennoscandia (Lethonen and Huttunen 1997, Niklasson and Granström 2000). Table 1 shows the global distribution of fires between July 2001 and June 2002 (Roy et al. 2008). It shows that Northern Eurasia had 4, 26% burned area, which is the second lowest frequency.

Table 1: Burned area and active fire area per continent, percentage of burned area and percentage of fire affected area in each continent during the July 2001 & June 2002 (Roy et al. 2008)

Continent	Burned area (x10⁵ km²)	Active fire area (x10⁵ km²)	Precentage of burned area in continent	Precentage of fire affected area in continent
Africa	25,0	13,7	68,41	49,40
Autralia −Oceania	6,32	3,87	17,26	13,94
Northern Eurasia	**1,56**	**2,82**	**4,26**	**10,16**
Southern Eurasia	1,56	2,14	4,27	7,70
North America	0,40	1,43	1,10	5,17
South America	1,72	3,79	4,70	13,64
Total	36,6	27,8	100	100

It is rather rare with wildfires raging over hundreds or thousands hectares in the Nordic countries. However, climate change together with changes in land use and high fuel loads is now assumed to cause increased amounts of forest fires in boreal forests (Cerdá and Robichaud 2009). A significant anthropogenic warming has occurred globally over the past 50 years (IPCC 2007a), and many climate models predict longer periods of droughts in multiple forested regions of the world (IPCC 2007b). Increased amount of wildfires has been documented in several areas around the world during the last centuries (Flanningan et al.

2005, Westerling et al. 2006, Cerdá and Robichaud 2009, Wendler et al. 2011). The most dramatic changes in climate are expected to occur in the Northern Hemisphere were the forest are predominately boreal e.g. in Canada and Russia (Stocks et al. 1998, IPCC 2001). Episodes of drier climate and more severe fire weather are expected to double the burned areas in several boreal regions in Canada (Flanningan et al. 2005) .

4.2 Acidification caused by anthropogenic pollution

Acidification refers to the lowering of pH in lakes, streams, soil, groundwater and precipitation due to anthropogenic pollutions. The lowering of pH occurs due to the loss of cations (e.g. calcium, magnesium and potassium) through the process of leaching and replacement of acidic ions such as aluminium and hydrogen. The effects of acidification are well known and it has caused decreased fish stocks and reduced biodiversity in many parts of Europe and northern America. It is considered as one of the most common anthropogenic change of inland waters (Lampert and Sommer 2010). It is caused by oxidation of nitrogen and sulphur. Nitrogen oxides and sulphur dioxide is converted to nitric acid and sulphuric acid by processes in the atmosphere or on the soil surface. The process of altering nitrogen and sulphur to their acid counterparts can take long time, which makes it possible to transport the acid compounds hundreds of kilometres from the original source. Thus nitric and sulphuric oxides and acids are among the most quantitative important long range transported air pollutants.

Deposition of sulphur primarily from burning coal and other fossil fuels has been the main cause of acidification in the European countries. The deposition of sulphur had a peak level during the 1970 and 1980 (EMEP 2011). Deposition of sulphur was significantly reduced during the late 1980s and lakes in the Nordic countries began to recover. Between the period 1990-1999 the recovery accelerated to 69% in the Nordic countries according to trend analysis of 344 lakes (Skjelkvåle et al. 2001).

 Since 1980 the concentration of sulphate in precipitation has declined with 72-90% in Norway (Schartau et al. 2011), and by 67% from 1986-2008 in southern Norway, the location of the wild fired area (Schartau et al. 2009). The sulphur pollutions has declined significantly in Norway, and accordingly the pH in Norwegian lakes (Figure 1) has increased from 1990-2010 (environment.no 2011).
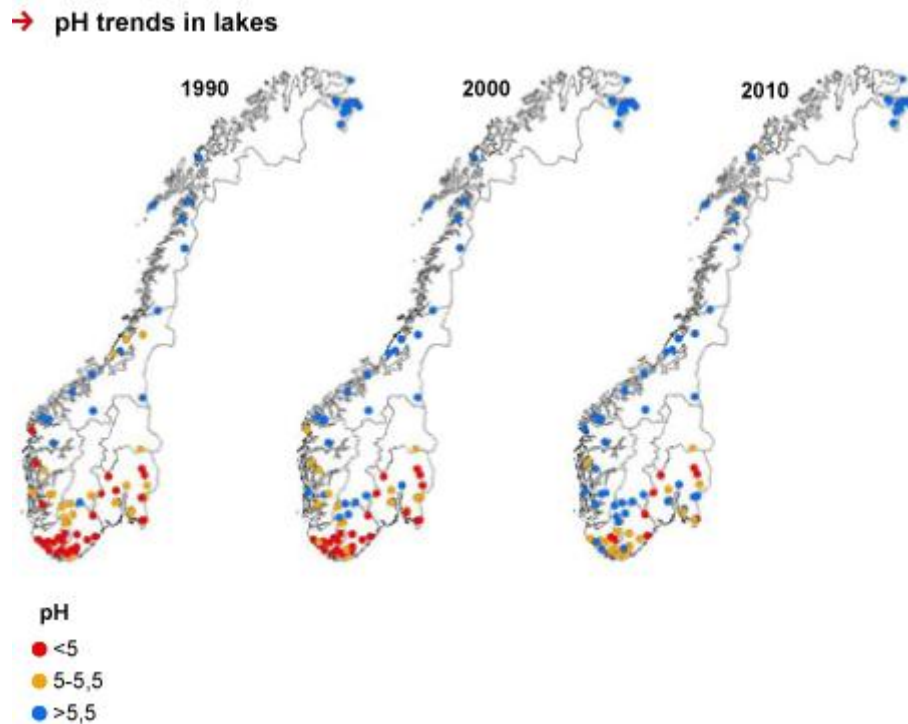
Figure 1: pH trends in Norwegian lakes (environment.no 2011)

Anthropogenic sources of nitrogen include fertilizer production, agricultural activity and fossil fuel combustion. The deposition of nitrogen was highest during the 1900s with a peak level in 1980 in Europe (Tarrason and Schaug 2000, Wright et al. 2001). The concentration of nitrogen in precipitation has declined by 26-46% in Norway from 1980 to 2010 (Schartau 2011). Nitrogen concentrations in Norwegian lakes had the highest decline up to 2005, with small changes after that year (Schartau et al. 2011). Nitrogen has a very complex biogeochemical cycle including uptake from plants and nitrification/denitrificantion processes. Chronic elevated N deposition increase leaching of NO_3 resulting in acidification. Moldan & Wright (2011) added 41 kg N ha^{-1} yr^{-1} in a catchment experiment during a 15 years period. About 5% of the total input went to runoff, 44 % to plants and 51% to soil. Other studies found that about 90% of N deposition was retained in soil and 10% was leached to nearby lakes (Kleemola and Forsius 2007). Studies have indicated threshold deposition of 8-10 kg N ha^{-1} yr^{-1} for elevated leaching of nitrogen. Several countries in Europe exceeds 10 kg N ha^{-1} yr^{-1}, however Norway are below this level (Galloway et al. 2008, INI 2010). The nitrogen concentration in Norwegian lakes are generally low (Skjelkvåle et al. 2001), which imply that atmospheric deposition of nitrogen is an more important factor in other parts of the world.

As mentioned above, sulphur and nitrogen oxides are released into the atmosphere and converted to acid compounds, separated into hydrogen, sulphate- and nitrogen ions (Figure 3). Areas especially sensitive to acidification are characterized by soil with low storage capacity for sulphate and protons, causing high leaking rates to streams and lakes. Acid sensitive soils are shallow post-glacial soil, soil with sandy texture and soil with high humus content (Alewell et al. 2000).

The chemical reactions depend on the content of the soil. Hydrogen ions (H^+) can react with alumino- silicate mineral implying leakage of toxic inorganic aluminium species. H^+ can also react with limestone minerals (Ca^{2+} and Mg^{2+} carbonates) and imply significant leakage of calcium (Ca^{2+}) and magnesium (Mg^{2+}). About 80% of the sulphate deposition will leak unaffected to nearby lakes (Lampert and Sommer 2010). Nitrogen takes part in several biogeochemical processes in the soil, and only about 10% of deposited nitrogen is considered to leak into inland water systems.
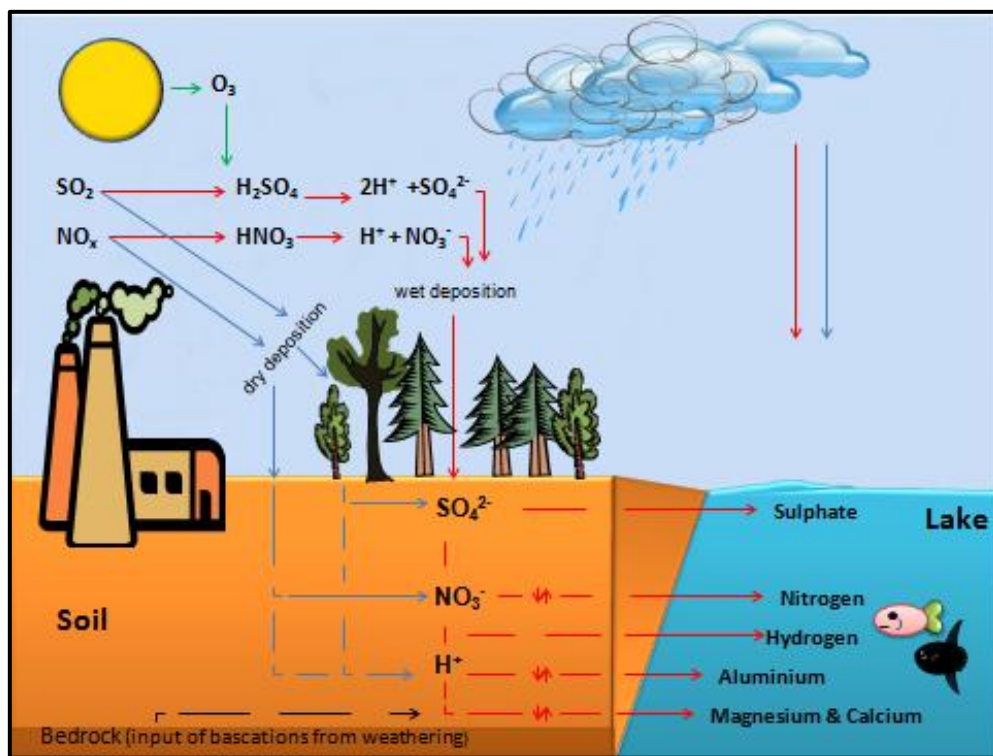

Figure 2: Main chemical reactions of anthropogenic acidification, figure inspired by Pidwirny (2006).

Natural sources to soil and freshwater acidification exist, e.g. nutrient uptake by vegetation, CO_2 production by microorganisms, and oxidation of natural N and S compounds. Acidification can also be caused when cations from seasalt are replacing H^+ ions in inland

waters close to coastal areas (Ulrich 1986). Acidification might also be caused by natural or indirect anthropogenic caused forest fires and droughts.

4.3 Water quality changes after Drought & Fires

Drought

The water table and moister content in soils and wetlands are reduced during long lasting drought. It can increase the evolution of nitrogenous gases in forest soils (Levia et al. 2011), and the shortage of moister increases the emission of NO_x relative to N_2O.

Droughts can further increase the concentrations of SO_4^{2-} and H^+ ions after in wetlands, streams and lakes (Bayley et al. 1992, Dillon et al. 1997, Jeffries et al. 2003). When the retained sulphur is exposed to air it will be oxidized (primarily SO_4^{2-}), this can cause effluxes of oxidized S during rewetting. Resumption of moister content has also resulted in elevated levels of strong acid, base cations and aluminium concentrations (Dillon et al. 1997). Clark et al (2005) further suggest a strong connection between SO_4^{2-} and DOC dynamics during droughts in wetland areas, release of SO_4^{2-} lead to low DOC concentrations in soils and surface water. High ammonium concentrations and nitrogen concentrations was measured in humic water after rewetting dry land in a study conducted by Ryan et al (1998). The production of ammonium was higher than the nitrification. Another study with artificial induced drought with roof constructions in four Norway spruce plantation did not follow the same pattern (Lamersdorf et al. 1998). Only one location showed decreased pH and increased aluminium concentrations but no marked nitrification pulse (surplus production of nitrate) was observed.

Previous studies of forest fires effect on water quality

Bayley et al. (1992) documented the effects of a forest fire and droughts in a stream in Canada, with draining area consisting of low weatherable rocks. The hydrogen concentrations more than doubled, with the highest annual concentrations two years after the fire. Also the annual mean ANC sharply decreased in two years after the fire. Accordingly the concentrations of base cations (Ca^{2+}, Mg^{2+}, Na^+, K^+) increased, but as the ANC decreased significantly, the   increase of in strong acid anions (SO_4^{2-}, NO_3^-, Cl^-) was far from compensated by the base cation increase.

The concentrations of base cations had still not returned to prefered levels nine years after the fire event. Concentrations of acid anions were three times higher than prefered levels two years after the fire. The increase was mainly due to increased sulphate and chloride concentrations. Annual sulphate concentrations was 3,1 times higher in the stream water in five years after the fire, and still higher than prefered values nine years after the fire. Chloride concentrations were 3.9 times higher than prefered concentrations during the fire year, and the concentration stayed elevated one year after the fire. Nitrate concentrations also increased after the fire, Bayley et al. (1992) suggest that enhanced nitrification was caused by the disturbance in the soil, and not by the fire itself since the fire occurred in a nutrient poor boreal forest. Their study further found a relationship between the increase concentration of calcium and magnesium ions versus the increase of sulphate ions during the first year after the fire, which indicated that the leaking of calcium and magnesium was balanced by the leakage of sulphate. Bayley et al (1992) concluded that the pH and ANC concentration in the stream generally recovered from the fire within three years.

Mast and Clow (2008) studied the chemistry in streams in two forest fired affected areas and one reference area within the Glacier National Park (located on bedrock containing carbonate rocks). During the first year after the fire, the nitrate concentration was 10 times higher in one of the burned areas (Coal Creek) compared with an unburned area. The chloride concentrations were 4 times higher and sulphate 2,7 times higher in Coal Creek than the reference area. Chloride concentration returned to similar concentrations as the compared area two years after the fire and the sulphate concentrations returned to prefered levels at the end of the first year, while nitrate concentration was still elevated 4 years after the fire. The differences in chloride sulphate and nitrate concentrations between Coal Creek and the reference were greatest during the winter. The elevated nitrate concentration might be caused by nitrate leaching from the soil into the groundwater due to reduced uptake capacity or increased nitrification in the soil. However, elevated nitrogen concentration in stream-water was also found in other parts of the park. The average DOC concentration was also found to be slightly higher in stream water in burned areas, however Mast and Clow (2008) concluded that the fire had minimal effect on the concentration since the difference was not changing during the study period.

Yang & Hai-qing (2007) studied concentration changes in several ions during May- October in a stream in Songling forest region in Daxing´an Mountains, China after an intense wildfire in 2006. They found higher concentrations of Ca^{2+}, Mg^{2+}, Na^{2+}, K^+, NO_3^-, Cl^- and SO_4^{2-} in

comparison to a unburned reference stream. The average concentration of calcium had the highest increase in the stream water, which existed in high quantities in the soil. The average increase of cation was higher than anion after the fire in Yang & Hai-quing study, which subsequently leads to increased ANC.

Prefire and postfire changes were also investigated at two sites (DC8 & DC2) in southern California by Jung et al. (2009). The catchment consisted of granitic and gneissic bedrocks. Precipitation, groundwater, soil water and stream water were collected before the fire and during the first and the forth post-fired year, Ca^{2+}, Mg^{2+}, Na^{2+}, K^{+}, SO_4^{2-}, Cl^{-} and other parameters were analysed in order to evaluate changes in post-fire watershed fluxes. Significant higher concentrations were measured in the stream in storm periods during pre-fired condition compared with post-fired conditions at DC2, and significant higher concentration of K^{+} and Ca^{2+} was measured during inter-storm periods after the fire. During the first post-fire year site DC8 had a slight increase of Cl^{-}, K^{+}, Na^{2+} and SO_4^{2-} during storm episodes, and decrease of all solute concentrations except K^{+} during inter-storm periods. The concentration of Mg^{2+} and Ca^{2+} was moreover lower during post-fire year 4 than during the prefire and postfire year 1. Their study further indicated high contribution of over land flow during early postfire (100% at two months), and a decrease of soil water and increase of groundwater entering the stream at site DC2.

The effect of wildfire on 4 lotic ecosystems was studied by Eriksson et al. (2003) in Tyresta National Park in Sweden. At all 4 sites, pH decreased and concentration of aluminium increased after the fire. Furthermore, the concentration of nitrogen, phosphorous and several metals increased. High concentration of potassium and chloride was measured in streams a year after the fire, and sulphate and calcium concentrations were also elevated. Increased concentrations of Mg^{2+}, Na^{2+}, K^{+}, SO_4^{2-}, Cl^{-} was also measured in a lake downstream the other sampling sites in two years after the fire. Brackish water was used during the fire fighting which might have contributed to an abnormal increase in Na^{+} and Cl^{-} (directly effect), as well as an abnormal acidification due to cationexchange in soil (indirect effect), as Na^{+} to a certain extent cationexchanges with H^{+} and Al^{n+} in the soil.

Summary of potential forest fire impacts on water quality

The fundamental conditions for forest fires include sufficient amount of fuel, a low moisture content (<25%) and ignition (Granström 2009). A fire will cause lower water table and evaporation than a drought, which will lead to further oxidation of S previously stored in the soils. It leads to oxidation of organic carbon to CO_2, organic S to SO_2 and SO_3 and organic N to NO and NO_2 and which later can fall as acid rain or dry deposition (as carbonic acid, sulphuric acid and nitric acids) (Lönnermark et al. 2007). The remaining alkaline ash will contain phosphate, chlorides and heavy metals together with base cations and polycyclic aromatic hydrocarbons (Eriksson et al. 2003, Smith et al. 2011).

 Wildfire also causes changes in soil properties and processes which cause decreased concentration of organic matter, structure, stability and reduced capacity to cationexchange as well as decreased water storage capacity in the soil (Ùbeda and Outeiro 2009, Vergnoux et al. 2011). Furthermore wildfires may also cause water repellent layers in soils which further reduces the infiltration rate (DeBano 2000). Reduced soil infiltration causes increased erosion and runoff, which will result in increased amount of acid substances entering the surface water which can be expected to cause severe acidification soon after the fire. Lakes affected by forest fires tend to be temporarily acidified with subsequent reduced buffering capacity (Pinel-Alloul et al. 2002, Eriksson et al. 2003). Runoff from burned bog dominated catchments after a fire might increase the humic content in lakes with decreased the clarity and reduce the light transmission of the water as a direct consequence (Pinel-Alloul et al. 2002).

The degree of erosion after a wildfire may differ depending on soil type, landscape susceptibility to erosion, magnitude and timing of storms, and the severity of the wildfire itself (Ryan et al. 2011). Various effects on water quality has been observed by different researchers as mentioned above, but almost all studies document increases in both base cations and major anions in streams and lakes. Mast and Cow (2008) and Bayley et al. (1992) showed the highest increase in chloride and sulphate, while Yang and Hai-qing (2007) reported highest concentration increase in calcium in the stream because of high concentration of this element in the soil. Eriksson, et al (2003) measured decreased pH and elevated aluminium concentration in all of their sampling sites, as a result of higher increases in strong acid anions as sulphate and chloride compared with base cations. Figure 3 illustrates general effects of forest fires on bedrock of low weatherable rock.
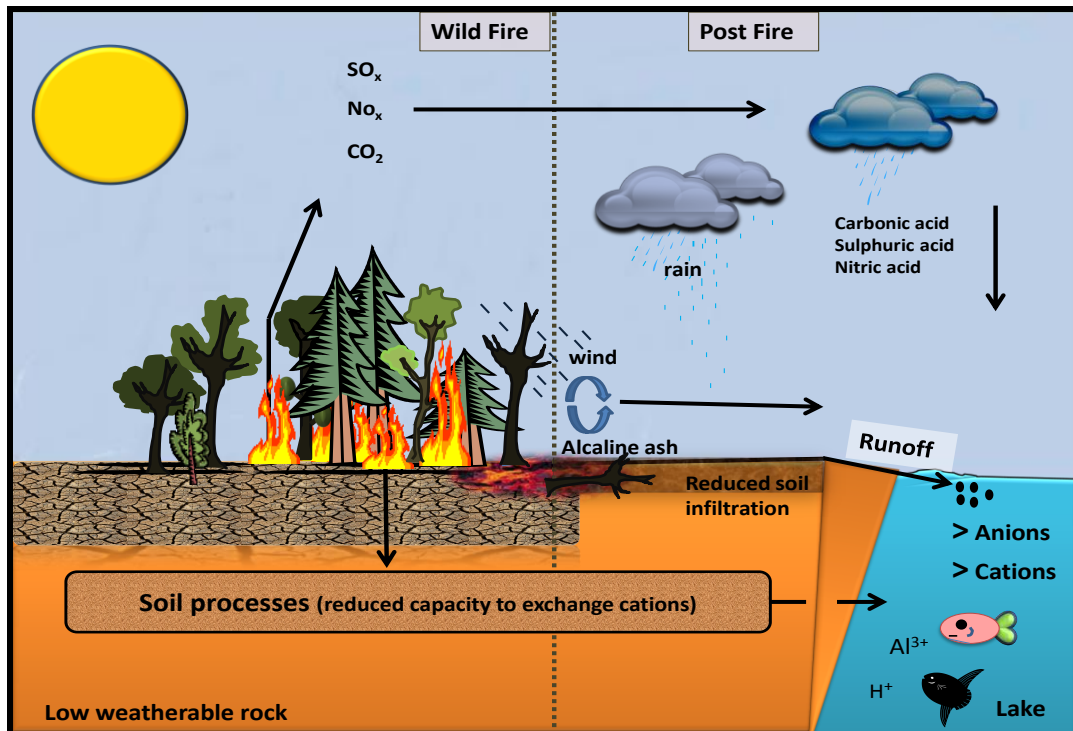
Figure 3: Possible main processes of forestfire effects on lakes.

4.4 Ecological effects of acidification

Organic carbon, pH, UV & Colour

Acidity in water is often measured in terms of pH, a parameter with both direct and indirect effects on water chemistry. On direct effect of pH is linked to critical limit for organisms, i,e, tolerance- and optimum levels for the aquatic organisms. pH will further strongly affect the solubility and speciation of metal ions as aluminium, with direct consequences for the metal toxicity (Hessen and Tranvik 1998). The ability to buffer hydrogen ions and inorganic aluminium is very low in clear water at pH < 5,5 (Laudon et al. 2001).

However, high concentrations of total organic carbon (TOC) are relatively common in boreal forest lakes in the Nordic countries and contribute to natural lowering of pH. As organic carbon consists of numerous organic acids, TOC contributes to a natural lowering of pH (O`Sullivan and Reynolds. 2004). These acids also play an important role for in which degree low pH affects organisms because they are able to complex inorganic aluminium, the acute toxic forms of Al to gillbreeding freshwater organisms. Accordingly it is uncommon that fish-populations are negatively affected in lakes with high TOC concentrations and pH $\geq 6 \pm 0.25$ (Lydersen and Löfgren 2000) .

On the other side, low pH will decrease the solubility of organic acids due to protonation of acid groups, and subsequently lowering their mobility and thereby the concentration of TOC (Laudon et al. 2001). Organic acids further regulate the light and UV absorbancy in lakes. As acidification tends to decrease the dissolved organic carbon, it directly increases the transparency in lakes. Dissolved organic carbon protects aquatic organisms from exposure to UV radiation (Morris et al. 1995, Schindler and Curtis 1997). Humic substances and chlorophyll produce brown and green water colour. The water colour in lakes might vary depending on the source of organic carbon, its solubility in the water, the concentration of iron and hydrological conditions (Arvola et al. 2010). These aspects are highly affected by seasonal variations, increased UV radiation, colour and TOC can be expected during autumn as a consequence of increased biological degradation of organic matter (Lydersen et al. 1996).

Inorganic aluminium

One major effect of acidification is the increased leakage of positively charged, inorganic aluminium (Al_i) to streams and lakes (Wetzel 2001). Al_i easily reacts with negatively charged sites on the surface of gills, forming a jelly formed precipitate on the gill surface (Figure 4). This process both reduces the oxygen uptake and the ionic regulation processes in gillbreeding aquatic organisms. At high concentrations, $Al_i > 50$ μg Al L^{-1}, it might have severe toxic consequences for fish and benthic animals. According to Canadian Guidelines total aluminium should not exceed 5 μg L^{-1} at pH <6.5 and 100 μg L^{-1} at pH ≥6.5 (CCME 2007). However, it is difficult to evaluate the critical concentration of Al_i, since it varies depending on several factors such as pH, temperature, total organic carbon and ionic strength (Lydersen and Löfgren 2000). Furthermore varies the toxic level due to biological factors such as tolerance difference between species, strains, stadium and age. According to the Norwegian guidelines, Al_i concentration > 65 μg L^{-1} indicates bad conditions for fish (Veileder 2009)
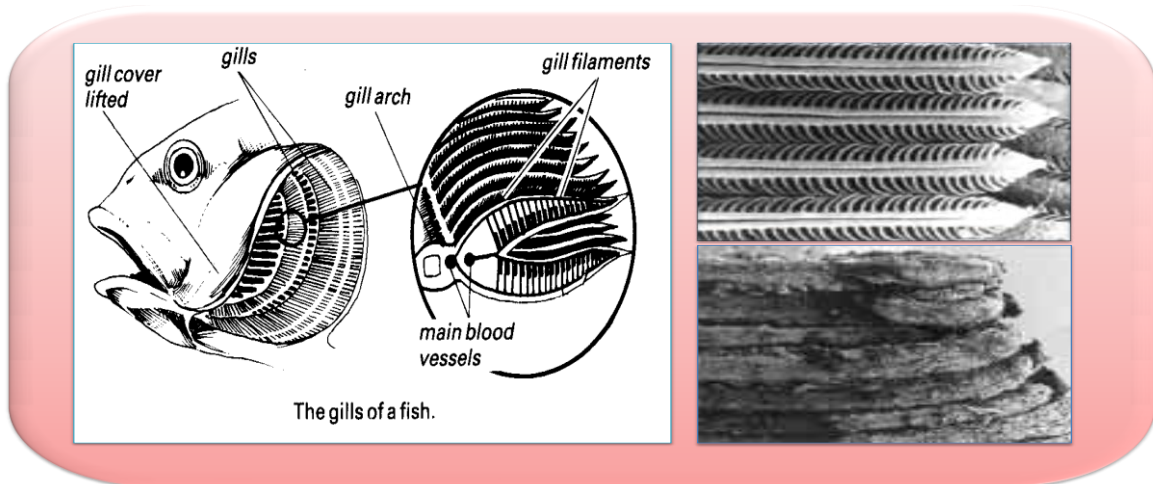
Figure 4: Left panel shows a schematic picture of a fish gill. Right panel shows the gill filaments of a healthy fish (uppermost) and gill filaments covered with inorganic Al (lowermost) (Laudon et al. 2001, arthursclipart.org 2009).

Inorganic aluminium is one of several reactive forms of aluminium occurring in the environment. Aluminium colloids are aluminium ions dispersed into another substance. Organic monomeric aluminium consists of organically-complexed monomeric aluminium and inorganic monomeric aluminium consist the inorganic aluminium and its complexes with sulphate, fluoride and hydroxide.
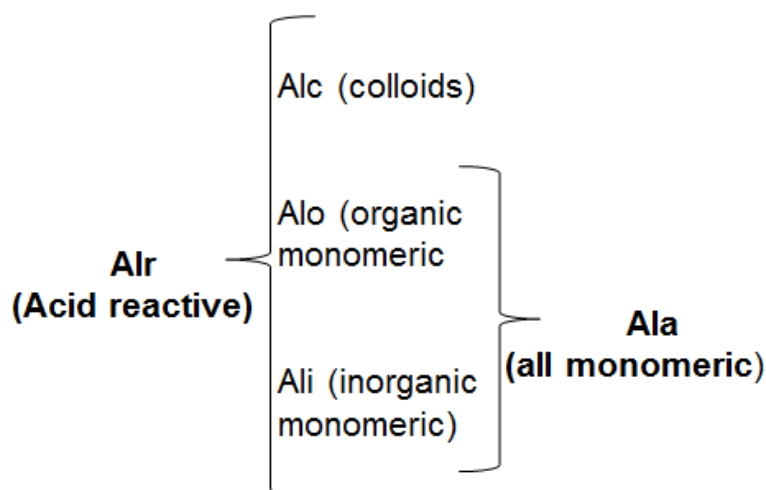

Figure 5: three reactive forms of aluminium.

Acid neutralizing capacity

ANC (acid neutralizing capacity) is documented to be a good measure of biological acidification status of lakes. As ANC is the equivalent differences between base cations (Ca^{2+},Mg^{2+}, Na^+, and K^+) and strong acid ions (SO_4^{2-}, Cl^- and NO_3^-), ANC is not affected by physical chemical factors as dissolved CO_2 which directly effects pH and subsequently the degree of protonation of organic acids and the distribution between toxic Al (inorganic Al: Al_i) and nontoxic Al (organic Al: Al_o)., This fact is likely the main reason why ANC better explain biological status in acidified water than the main toxic factor, Al_i and pH (Bulger et al. 1993, Lien et al. 1996). Fish populations are normally not affected by acidification in freshwaters with ANC >20 µekv L $^{-1}$ (Lydersen et al. 2004). However as organic acids are another main factor for ANC and thus fish status, Lydersen et al. (2004) has included this in the ANC parameter (organic acid adjusted ANC, ANC_{OAA}), and found that it is > 95% probability of no effects on Norwegian fish species   at ANC_{OAA} > 8 µekv L $^{-1}$.   The Norwegian monitoring guideline is still not using ANC_{OAA}, but the traditional ANC, and instead divided ANC in various TOC classes when evaluating water quality in acidified Norwegian waters (Table 2)

Table 2: Concentrations of pH, Al in ANC & TOC indicating the water status for fish according to the Norwegian guideline developed for management of WFD in Norway (Veileder 2009).

Parameter	unit	Very good	Good	Moderate	Bad	Very Bad
pH		>6,5	6,5-5,8	5,8-5,4	5,4-4,9	<4,9
Ali	µg/L	<15	15-30	30-65	65-125	>125
ANC:TOC 2mg/L	µekv/L	>50	30-15	15-5	5- -5	<-5
ANC:TOC 2-5mg/L	µekv/L	>70	70-25	25-10	10- -10	<-10
ANC:TOC >5mg/L	µekv/L	>100	100-55	55-15	15- -10	<-10

5 Study area

24500 hectares of forest and 1500 hectares of lakes were affected by the wild fire in Mykland, June 2008. Mykland is located in Froland municipality in the county of Aust- Agder, southern Norway. This wildfire was the largest in Norway since 1900 (Hogberget et al. 2010). 6 lakes within the wildfire area have been investigated. 5 of these lakes, Lake Heitjenn, Lake Grunnetjenn, Lake Hundsvatn, Lake Fisketjenn and Lake Øyvatn belong to the same draining system, lake Rasvassvatn belong to other draining system (Figure 6). 3 lakes are used as reference, Lake Jordtjenn is located about 1 km upstream from Lake Svarttjenn, and Lake Melestjenn, belongs to another subcatchment. The lakes are located approximately between 191-454 m over sea level (Table 3), and > 40 km from the sea.

Table 3: Altitude (min & max), latitude and longitude for the sampled lakes. Data from the Norwegian Energy Directorate database (NVE 2010).

| Lake	Altitute min m asl	Altitude max m asl	Latitude	Longitude
Heitjenn	min: 271	max: 288	N58 37.073	E8 16.370
Grunnetjenn	min: 231	max: 293	N58 36.906	E8 16.515
Hundsvatn	min: 228	max: 320	N58 36.413	E8 16.770
Fisketjenn	min: 229	max: 245	N58 36.549	E8 17.194
Øyvatnet	min: 255	max: 316	N58 36.675	E8 19.037
Rasvassvatn	min: 173	max: 360	N58 37.446	E8 20.165
Jordtjenn Ref.	min: 228	max: 464	N58 38.956	E8 18.721
Svarttjenn Ref.	min: 191	max: 464	N58 39.118	E8 20.220
Melestjenn Ref.	min: 222	max: 243	N58 37.598	E8 22.063
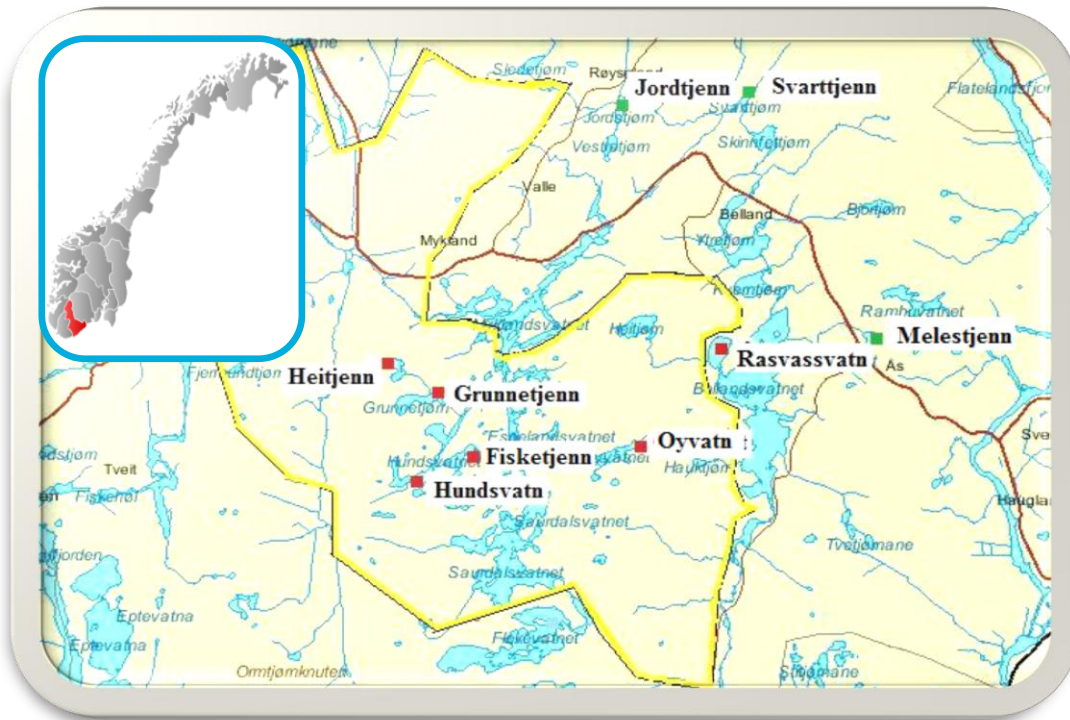
Figure 6: Map over the investigated lakes. Red square: Lakes within the wild fire area; green square: Control lakes. The wild fire is within the yellow line. The map is a modified map from Statens-Kartverk (2011).

All lakes are situated in areas dominated by forest on poor soil and areas with bogs (Table 4). Due to long commercial forestry in the area, 50-60% of the forest consists of young trees (Storaunet et al. 2008). The vegetation is dominated by heather species and Norwegian pine and birch (Lydersen 2011). Peat marshes and bogs are located in the bottom of valleys. The soil cover is mostly shallow but there are thicker layers in the valleys. The bedrock consists of banded gneiss, augen gneiss and migmatite (Figure 7). The slowly weatherable bedrocks in the area are decisive for the low buffering capacity in the surface waters.
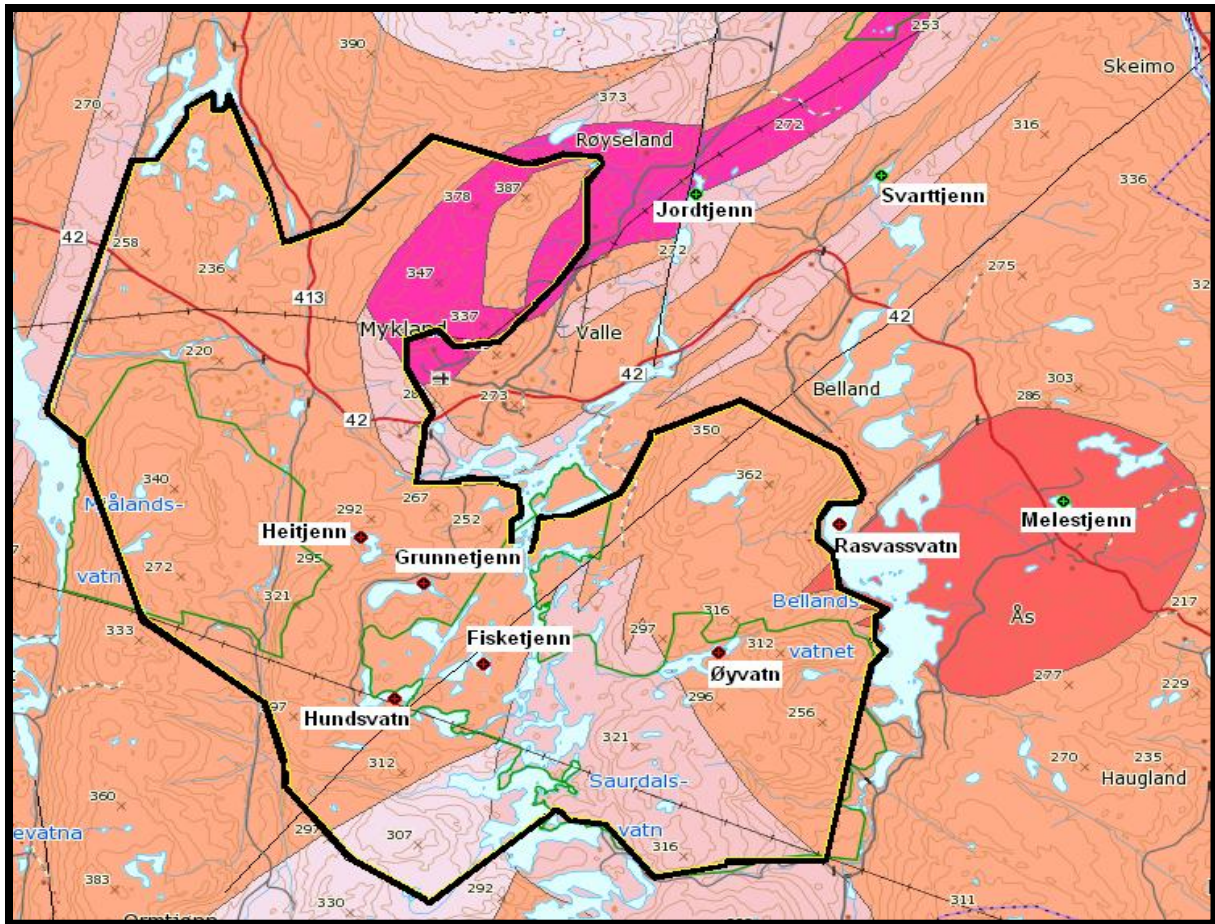
Figure 7: The geology of sampled lakes. Modified map from NGU (2012).

Amphibolite, Hornblende Gneiss, Mica Gneiss

Augen Gneiss, Granite, Foliated Granite

Migmatit, Diorite Gneisses & Granitic Gneisses

Syenite, Quartz Syenite

Granite, Granodiorite

The area has been severely impacted by long range transported pollutions (including acid rain) for several decades (Hogberget et al. 2010), and accordingly several lakes in the area have been limed to reduce the acidification problems and save the original fish species. All the lakes investigated are located above these limed water systems, to avoid this artificial factor. There is almost no agricultural activity or urban areas within the wildfire area (Table 4).

Table 4: Catchment characteristics (covered area in %), data received from the Norwegian Energy Directorate database (NVE 2010).

Lake	Agricultural Area %	Naked Mountain %	Urban Area %	Bogs %	Lake %	Forest %
Heitjenn	0,0	0,0	0,0	3,1	26,4	52,3
Grunnetjenn	0,0	0,0	0,0	10,7	11,9	70,4
Hundsvatn	0,0	0,0	0,0	22,0	12,4	61,0
Fisketjenn	0,0	0,0	0,0	29,8	26,8	37,5
Øyvatnet	0,0	0,0	0,0	11,8	8,1	79,3
Rasvassvatn	0,0	0,0	0,0	7,8	12,5	78,2
Jordtjenn [Ref.]	1,3	0,0	0,0	7,8	3,0	85,4
Svarttjenn [Ref.]	2,0	0,0	0,0	13,3	1,8	81,7
Melestjenn [Ref.]	5,2	0,0	0,0	8,6	15,7	62,4

Lake volume and lake area of the investigated lakes vary between 18 807 – 4 573 287 m^3 and 0,0091- 0,8909 km^2. Lake Rasvassvatn and Lake Hundsvatn are the deepest lakes 15,4 and 13,0 m respectively, while Lake Heitjenn has a maximum depth of only 2,5 m. Lake Svarttjenn has the largest catchment area, i.e. 6,86 km^2 which is the largest of all samples lakes. The residence times ranges between 0,01-3,18 per year (Table 5).

Table 5: Lake characteristics, data measured and calculated by Lydersen (2011).

Lake	Maximum Depth m	Lake Volyme m^3	Lake Area km^2	Catchment area km^2	Residence time per year
Heitjenn	2,5	30417	0,0365	0,14	0,19
Grunnetjenn	3,2	72000	0,0675	0,86	0,07
Hundsvatn	13	647833	0,1495	2,63	0,21
Fisketjenn	5,8	108847	0,0563	0,23	0,40
Øyvatnet	3,2	76587	0,0718	1,09	0,06
Rasvassvatn	15,4	4573287	0,8909	1,23	3,18
Jordtjenn [Ref.]	9,5	74733	0,0236	2,87	0,02
Svarttjenn [Ref.]	6,7	45783	0,0205	6,86	0,01
Melestjenn [Ref.]	6,2	18807	0,0091	0,13	0,12

6 Methods

6.1 Sampling & Analyzes

Investigations of biological and water chemical effects of the Mykland wild fire was funded by the Norwegian Research Council, and thus most of the data used in this thesis have been collected and analysed by others. Lake Rasvassvatn and Lake Hundsvatn have been monthly sampled from June 2008, a week after the termination of the wildfire, until May 2011, while monthly sampling of the remaining lakes started in May 2009. Water samples were taken at the deepest site of the lakes with a Limnos sampler. All lakes were sampled at 1m depth. In addition water samples were also monthly collected in hypolimnion of the stratified dimictic lakes: Lake Rasvassvatn, Lake Jordtjenn and Lake Hundsvatn. Water samples were taken in prewashed polyethylene bottles and stored cold (4°) in darkroom until analysed. An aliquot of each water samples were stored in a freezer in case of reanalysis etc. Aluminium analyses were done according to the Barnes/Driscoll method (Barnes 1975, Driscoll 1984). The ion exchange fractionation of aluminium was performed the same day as the sampling (se figure 8), while the remaining macrochemistry was performed at the laboratory at Telemark University College, according to the methods presented in Table 6. From March 2011 an ion chromatograph (IC) was provided,

Figure 8: Illustration of the separation of inorganic aluminium from other aluminium species (See figure 6).

and from that time Cl^-, SO_4^{2-}, NO_3^-, Ca^{2+}, Mg^{2+}, Na^+, K^+ and NH_4^+ were analysed by IC. All frozen samples before March 2011 was taken up and reanalysed both by the new and old method for intercalibration purpose.
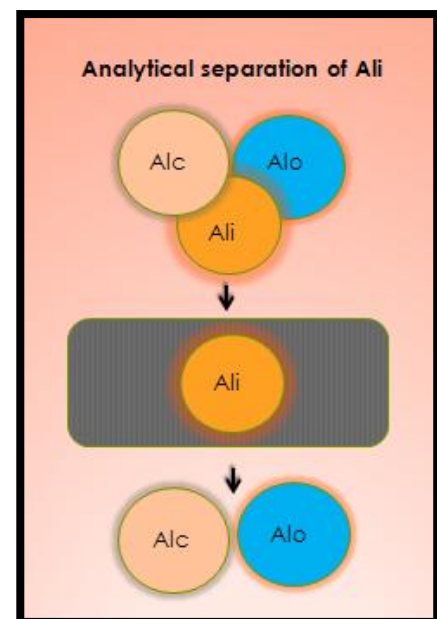
Table 6: Overview of the analytical method used. After March 2011 Ca^{2+}, Mg^{2+}, Na^+, K^+, NH_4^+, SO_4^{2-}, Cl^- and NO_3^- has been analysed with a ion chromatograph (IC).

Parameter	Unit	Reference
PH	-log [H+]	NS 4720, 1979, 2. Edition.
UV-absorbency		Internal method, absorption at 254 nm
Colour	mg Pt L^{-1}	NS 4878 (2002)
Chloride (Cl$^-$)	mg L^{-1}	Potentiometric titration with 0,01 M AgNO$_3$
Sulfate (SO$_4^{2-}$)	mg SO$_4$ L^{-1}	FIA method: Tecator ASTN 15/84
Nitrate (NO$_3^-$)	μg N L^{-1}	FIA method: Tecator ASN 136-01/91
Calcium (Ca^{2+})	mg L^{-1}	NS 4770 + NS 4776
Magnesium (Mg^{2+})	mg L^{-1}	NS 4770 + NS 4776
Sodium (Na$^+$)	mg L^{-1}	Internal method. Emission analysis
Potassium (K$^+$)	mg L^{-1}	Internal method. Emission analysis
Ammonium (NH$_4^+$)	μg N L^{-1}	Internal method based on NS 4746 (1975)
Total organic carbon	mg C L^{-1}	Internal method. OI Analytical 1010
Total Al	μg L^{-1}	Barnes/Driscoll (Barnes, 1975; Driscoll, 1984)
Total monomeric Al	μg L^{-1}	Barnes/Driscoll (Barnes, 1975; Driscoll, 1984)
Organic monomeric Al	μg L^{-1}	Barnes/Driscoll (Barnes, 1975; Driscoll, 1984)

A schematic illustration of the Ion Chromatograph ICS 1100 used, are shown in Figure 8. It consists of liquid eluents, a high-pressure pump, a sample injector, a guard and separator column, a chemical suppressor, a conductivity cell, and a data collecting system (Thermoscientific 2011). The liquid anion eluent contained 1 L distilled water, 2,289 g NaHCO$_3$ and 0,168 g Na$_2$CO$_3$, while the cation eluent contained 10 ml of 1000 mM H$_2$SO$_4$ and 0,999 L distilled water. The water samples were loaded into a sample loop, and the pump pushed the liquid eluent and water sample through the loop, the guard column, and the separation column. Different ion migrated through the separation column at different velocities. When the ions reached the suppression they were selectively detected by the conductivity meter. The data analysis system identified and quantified each ion by evaluating the peak area and peak height and comparing it to peaks from standard solutions. Standard solutions and controls were used for calibration and quality control.
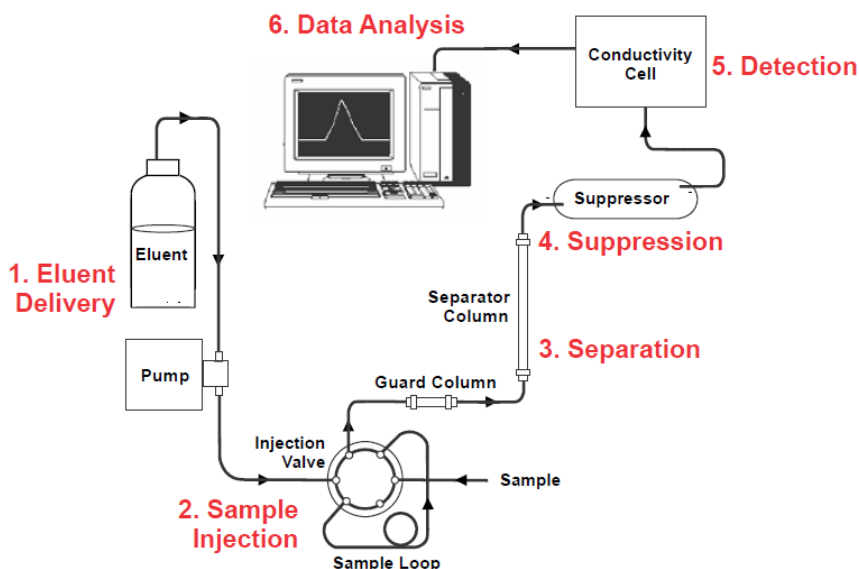
Figure 9: Schematic illustration of Ion Chromatograph ICS 1100 (Thermoscientific 2011).

6.2 Calculations

Aluminium

Aluminium was analysed spectrophotometrically on HQ/MIBK extracts on untreated water, acidified water and cationexchanged water according to the Branes/Driscoll method (Barnes 1975, Driscoll 1984). The samples was analysed at both 395 nm and 600 nm. The absorbancy at 600 nm was used for iron correction, due to iron interference with Al at 395 nm.

Total monomeric aluminium (Al_a) was determined by extraction of a untreated water sample in a hydroxyquinoline (HQ), methyl isobethyl ketone (MIBK) solution for 20 seconds, as described by Barnes (1975). Al_a consist of low molecular weight inorganic (Al_i) and organic aluminium (Al_o). In addition a preacidifed water samples (pH=1 for $\geq$ 24 hrs) underwent a similar extraction in order to obtain a total aluminium concentration, denoted Al_r. Thus the difference between Al_r and Al_a will be a measure of the concentration of aluminium colloids and particles (Al_c). In order to separate total monomeric aluminium (Al_a) into inorganic Al (Al_i) and organic Al (Al_o), the cation exchange procedure described by Driscoll (1984) was used. By analysing aluminium in the cation exchange eluate, Al_o is determined. Then Al_i can be calculated as follows:

$Al_i = Al_a - Al_o$

ANC

The acid neutralizing capacity (ANC) concept was first described by Reuss and Johnsson (1986), and is a robust measurement of acidification status in freshwaters. ANC is the difference between base cations and strong acid anions on equivalent basis, i.e.:

$ANC = \Sigma Ca^{2+}, Mg^{2+}, Na^{+}, K^{+}, - \Sigma SO_4^{2-}, Cl^{-}, NO_3^{-}$

It is a robust method because all parameters incorporated is not affected by CO_2 variations in water, and can therefore be analysed very accurate in soilwater, surface water, groundwater.

ANC_{oaa} is a modification of ANC which also include permanently deprotonated organic acids in water, i.e. an organic acid adjustment of the traditional ANC estimate (Lydersen et al. 2004). This modification might give more reliable ANC values in humic rich water systems were significant amounts of organic acids are present. Calculaton of ANC_{oaa} is done by the following equation:

$ANC_{oaa} = ANC - (1/3\ CD*TOC)$

6.3 Statistics

Descriptive statistics

Many statistical methods assume normal distribution of the data. Anderson Darling normality test (AD-test) can be used to verify normality or non normality of data. The method is considered to be effective at detecting divergent distribution from normality in high and low values (Minitab 2010). The results of AD-test have good consistency in its results, it gave values very close to 0,05 for all samples sizes tested by Yazici and Yolacan (2006). The test rejects the hypothesis of normality when the p-value < 0,05, which further means that it is a 95% possibility that the data does not fit normal distribution. If the assumption of normality is met, a parametric test should be used as this has greater power than non-parametric tests (Wheater and Cook 2000). However, if parametric tests are used on non-parametric data it might lead to unreliable results. The drawbacks with the AD-test, is the poor precision. If the data includes significant number of ties the AD-test diagnoses the data as non-normal, regardless of how the data fits a normal distribution (Taylor 2011). All normality tests have drawbacks and should not entirely substitute histogram or other visual examination of the data material (Statsoft 2012).

Time series plots are a statistical method for analysing observations over time. The aim of time series analysis is to analyse changes that have occurred or to make predictions of future changes. Time series analyses are assumed to consist of systematic patterns and errors, which complicates the identification of the systematic pattern. The time series plots used in this study do not include any techniques for minimizing errors, more than carful sampling and analysing of the parameters. Trend and seasonal patterns are two basic classes of time series patterns (Statsoft 2012). These patterns are expected to coexist in the data in this study, since seasonal changes of temperatures and runoff have great effects on water quality parameters in inland waters.

Standard deviation measures the variation in normal distributed data. A large standard deviation (stdv) in relation to the mean of the data indicates a high variance. The mean ± one stdv will account for 68, 27 % of the data points. Median ± max and min is more suitable to use than the mean and standard deviation when the data points are not normally distributed. The median is less sensitive to the influence of outlines than the mean. The max and min is the largest and smallest observation and will include 100% of the spread of data points. The usefulness depends on if the minimum and maximum values are real values or affected by measurement errors

T-test can be used if the number of observations are small and the variances in the populations from which the two samples were taken are similar (Wheater and Cook 2000). It compares the difference between two means, the difference of the two samples will be divided into explained (mean) and unexplained variation (standard variation). If the standard deviation is large it implies that the data collection contains variability which can't be explained by its membership of the two categories. A big T-value indicates a large difference between means and a small standard deviation, which leads to a small p-value and accordingly a significant difference between the means.

Wilcoxon matched-pairs test (W-test) is a nonparametric test equivalent to paired T-test. Wilcoxon matched-pairs test calculate the difference between pairs of data points and ranking them in relation to the size of difference between paired data. Then the sums of higher and lower ranks are calculated and presented as T^- & T^+. If the positive and negative rank is in balance it will indicate no significant different between the paired samples. The lowest T-value of T^+ and T^- in relation to numbers of pairs will determine if there are significant different or not. The pairs will be considered different from each other if the p-value is less than 0,05. The W-test is more sensitive for small sample sizes than paired T-test, as it might

miss detection of small differences if the sample size is too small. In the thesis paired T-test and W-test were used in order to evaluate difference between samples taken at the same months at different year (2009 and 2010).

Bivariate Methods

Correlation and regression are the main methods for analysing relationships. Correlation can be used to test the strength of association between two variables, while regression predicts the values of one variable (dependent) from another (independent).

In the thesis, correlation was used when comparing analytical data obtained by different analytical methods. Pearsons correlation measures the strength of the linear relationship between two quantitative variables (McKenzie and Goldman 2005). The correlation lies between -1 and 1, were 1 means perfect positive relationship between two variables. A correlation coefficient was chosen since it cannot be assumed that there is cause and effect relationships between the results obtained with the two analyze methods. The correlation coefficient is compared with the probability (p-value) to test if there is a significant relationship between two variables. Both the sample size and the result of the correlation coefficient influence the p-value (Wheater and Cook 2000). It means that a large sample size can obtain a significant relationship even at a quite low correlation coefficient. Spearmans rank correlation is a non parametric method. This method ranks each variable separately and then compares the ranks for each data pair. In regression it is possible to draw a line of best fit through the data.

7 Results and Discussion

7.1 Time series plots

Almost all concentrations of cations and anions increased substantially during a heavy rain period in August 2008, two months after the forest fire (Table 7). The larger increase of ions in Lake Hundsvatn can be linked to the higher catchment area/lake area ratio (CA:LA-ratio) in this lake (CA:LA=17,6) compared with Lake Rasvassvatn (CA:LA-ratio) in this lake (CA:LA=1,38), which means a larger impact of wildfire on Lake Hundsvatn compared with Lake Rasvassvatn. However, Lake Hundsvatn had a shorter residence time (0,21 yr) compared to Lake Rasvassvatn (3,18 yr), which might be the cause of the lower concentrations of most of the parameters in Lake Hundsvatn after the first post-fire months and forward. The concentrations of cations and anions showed seasonal variation but gradually decreased in most of the parameters after the peak in August 2008, except for some temporarily high values linked to hydrological episodes as snowmelt and heavy rain and following drought periods (Figure 11).

Table 7: Factor increases of ion concentrations, conductivity and factor decrease of ANC$_2$ two months after the forest fire.

	SO_4^{2-} mg L^{-1}	Cl$^-$ mg L^{-1}	NO_3^- μg L^{-1}	Ca^{2+} mg L^{-1}	Mg^{2+} mg L^{-1}	Na$^+$ mg L^{-1}	K$^+$ mg L^{-1}	NH_4^+ μg L^{-1}	H$^+$ μekv L^{-1}	Ali μg L^{-1}	ANC2 μekv L^{-1}	Cond. μS/cm
Rasvassvatn	1, 3	1,49	4,4	1,55	1, 92	1, 28	1,81	1,68	1,95	2,33	1,36	1,58
Hundsvatn	4, 6	2,53	1,77	3,08	3, 94	1, 84	5, 25	4,06	7,93	6,39	16,08	3,51

After the peaks in August 2008, there were no substantially higher peaks of Ca^{2+}, Na$^+$ and Mg^{2+} in Lake Rasvassvatn and Lake Hundsvatn compared with the reference lakes (Figure 10). It can however be concluded that the calcium concentration was lower in Lake Rasvassvartn and Lake Hundsvatn in the end of the investigated period (until September 2011) then in the three reference lakes, it was further a bit larger variations of Mg^{2+} concentration in Lake Rasvassvatn and Lake Hundvatn then in the reference lakes. The potassium concentration was moreover substantially higher in Lake Rasvassvatn and Lake Hundsvatn (also in the rest of the burned lakes see Annex 2) compared with the reference lakes. There was moreover a visible decrease of the concentration of K$^+$ and Na$^+$ in the end of the investigated period.
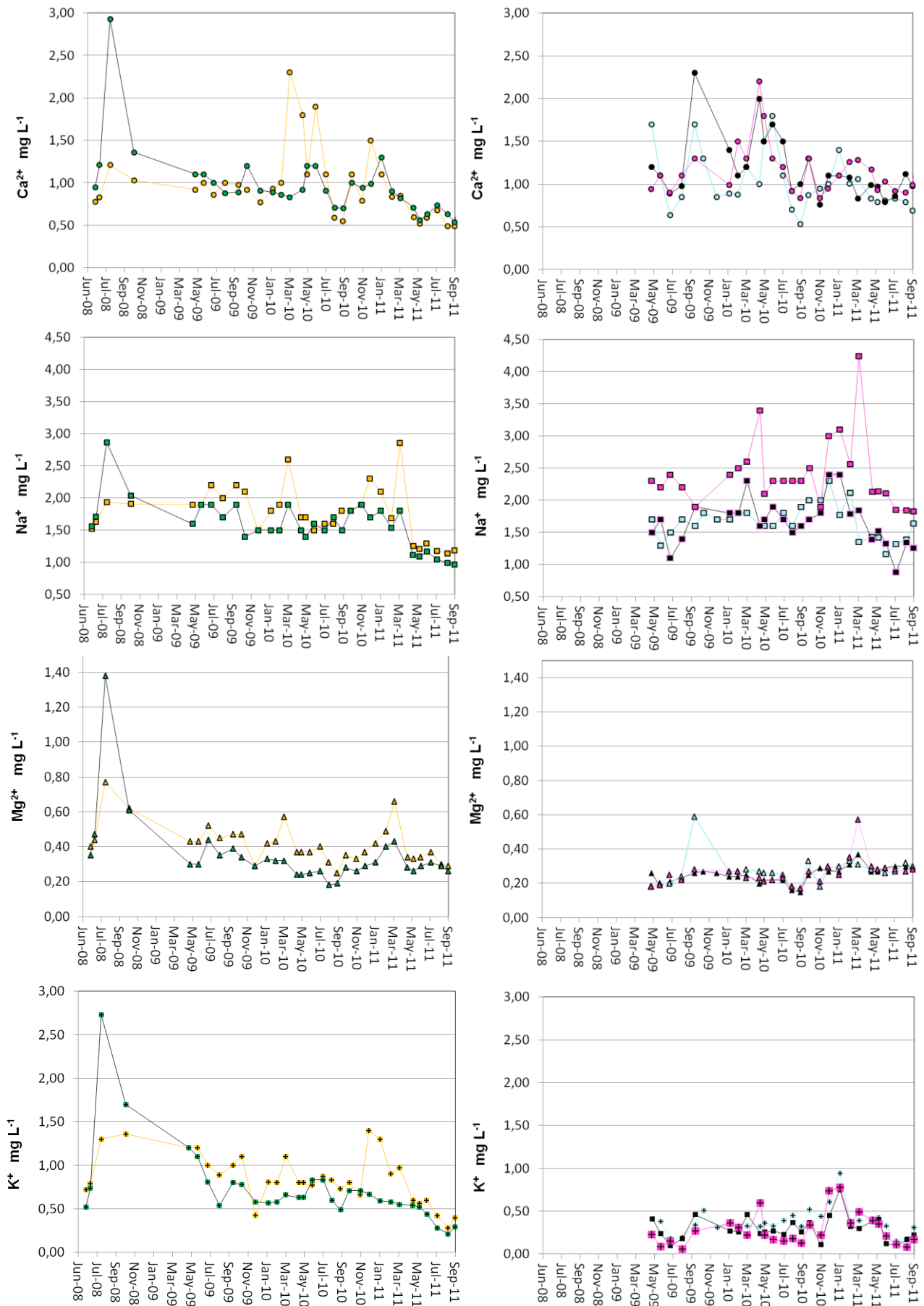
Figure 10: Time series plot of Ca^{2+}, Mg^{2+}, Na^+ & K^+ (mg L^{-1}) in the burned lakes Rasvassvatn (yellow) and Hundsvatn (green), and the reference lakes Melestjenn (pink), Jordtjenn (blue) and Svarttjenn (black).
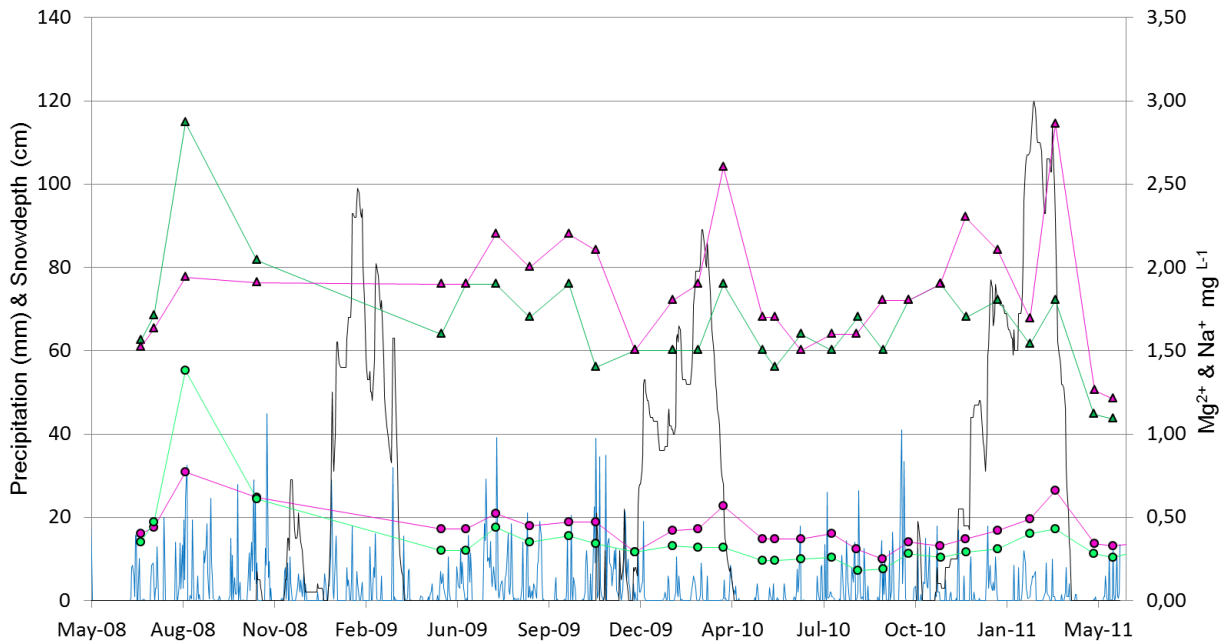
Figure 11: Snow depth (black line) and precipitation (blue line)(met.no 2012), in relation to Mg^{2+} (circle) and Na^+ (triangular) concentration in Rasvassvatn (pink) and Hundsvatn (green).

The sulphur concentration in Hundsvatn decreased with approximately 10 mg L^{-1} from August and September 2008 and then varied between 5,1 – 1,27 mg L^{-1}, while lake Rasvassvatn varied between 7,0-1,89 mg L^{-1} during the same period (Figure 12). The concentration of SO_4^{2-} seemed to be a bit elevated in Lake Rasvassvatn and Lake Hundsvatn compared with the reference lakes. Lake Rasvassvatn and Lake Hundsvatn showed visible decrease of chloride and sulphate concentrations starting in May 2011, however the reference lakes did also follow the same decreasing trend of the SO_4^{2-} concentration. Lake Melestjenn had larger Cl^- concentrations then all other lakes. It is likely that the higher concentrations is linked to road salting, since the lake is close to a road and shows a large increase during snowmelt after the cold winter 2011, this would further explain the higher concentration of sodium in this lake.
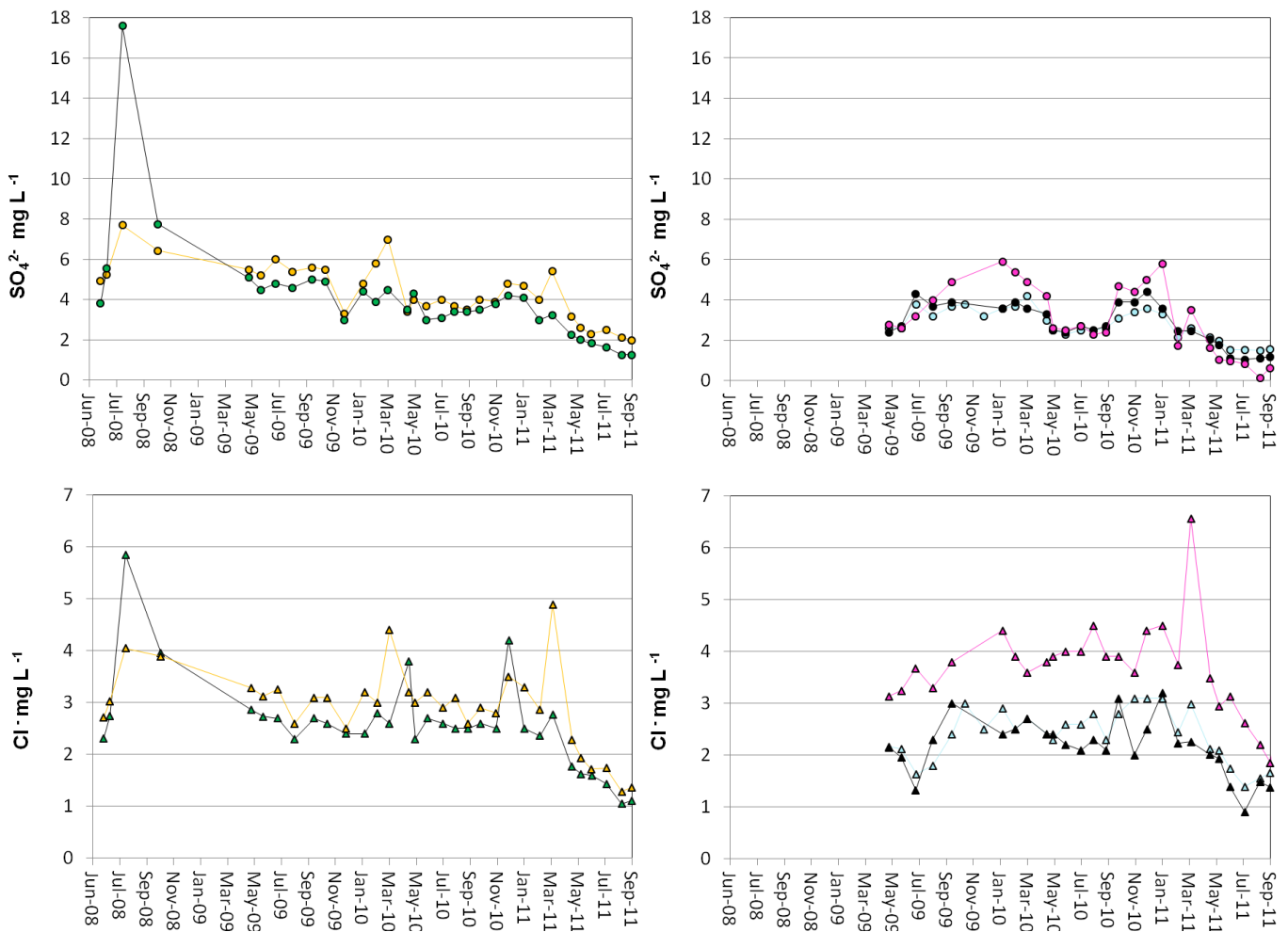
Figure 12: Time series plot of SO_4^{2-} and Cl^- (mg L^{-1}) in the lakes Rasvassvatn (yellow) and Hundsvatn (green) (right panel), and the reference lakes (left panel) Melestjenn (pink), Jordtjenn (blue) and Svarttjenn (black).

The nitrogen concentration showed larger peaks and variability thought the year, than during the two months after the forest fire for both Lake Rasvassvatn and Lake Hundsvatn (figure 13), and the other burned lakes seemed to follow the same pattern (Appendix 2) as well as the three reference lakes. The concentration of NO_3^- was slightly higher in the fired lakes then in the reference lakes (Appendix 2 & 4). The ammonium concentration showed also more variability through the year, but seen in Appendix 2 the differences was more site specific then for the nitrogen concentration.
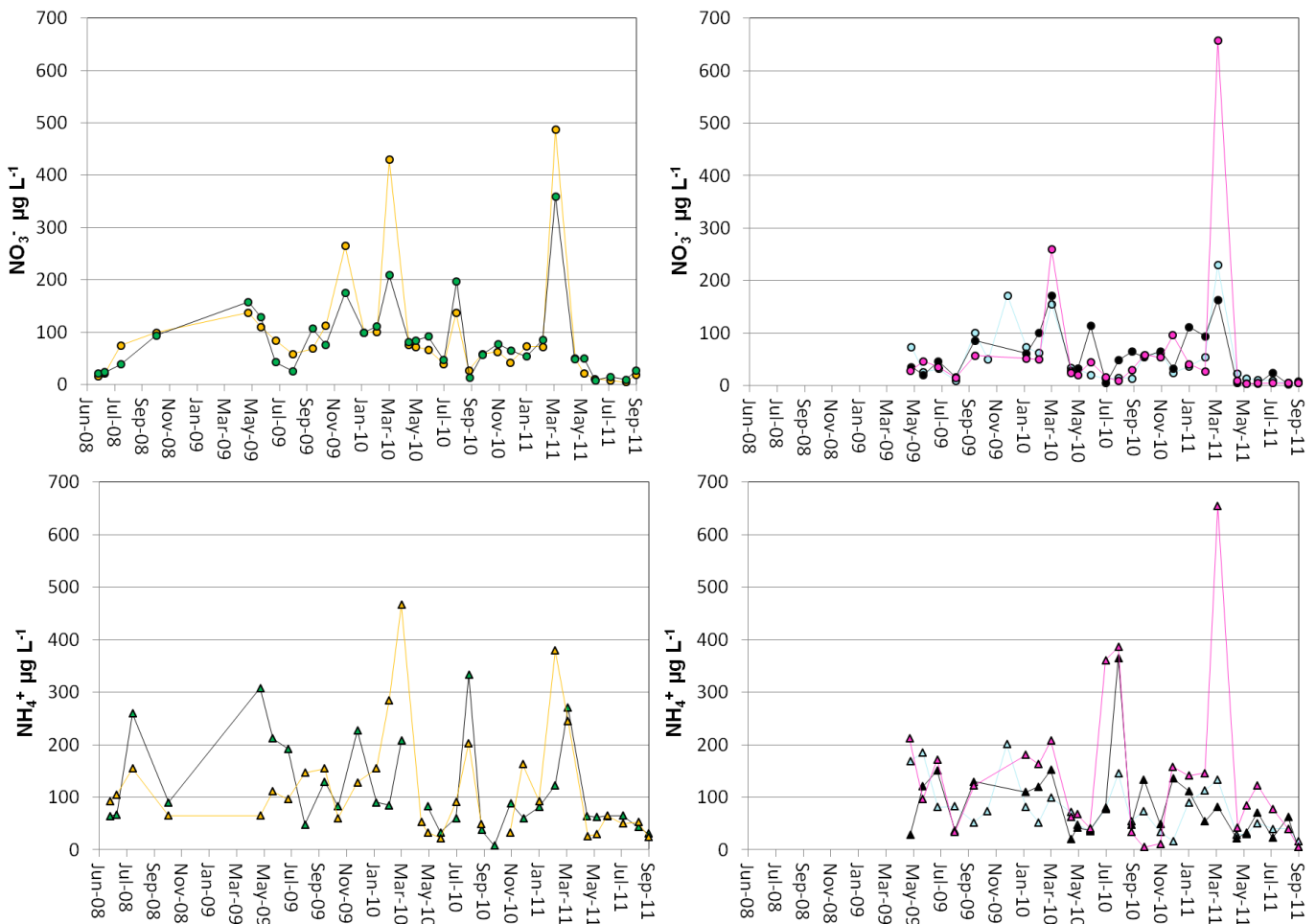
Figure 13: Time series plots of NO_3^- (circle) and NH_4^+ (triangular) μg L^{-1}, in the lakes Rasvassvan (yellow) and Hundsvatn (green) (right panel), and the reference lakes (left panel) Melestjenn (pink), Jordtjenn (blue) and Svarttjenn (black).

Figure 14 show that the concentration of hydrogen followed a similar pattern in both Lake Rasvassvatn and Lake Hundsvatn. The reference lakes had lower H^+ concentrations than Lake Rasvassvatn, Lake Hundsvatn and most of the other burned lakes (Appendix 2) during the first sampling in May 2009 and forward except for some peaks. The lakes ANC_2 values were more differentiated in each lake then the hydrogen ions. It can further be concluded that the two burned lakes had bad condition for fish since the ANC_2 value was under zero and lower in both of the lakes in all measurements from 2008 to 2009, and continued to have low values in 2010 except for some increasing peaks in Lake Rasvassvatn. The reference lake, Lake Melestjenn was however the lake with the most decreasing peaks of ANC_2 compared with all other lakes. The concentration of ANC_2 had an increasing trend in most of the lakes during the end of 2011 (Appendix 2).

The high peak of Ali two months after the forest fire was more than double of what is considered to cause very bad condition for fish, and thus must have caused extensive biological damage in the lakes. Moreover was the concentration of Al_i much higher in Lake

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Rasvassvatn then in Lake Hundsvatn, the reference lakes and all other lakes (Figure 14 and Appendix 2). This was probably due to its low TOC concentration and rather high concentration of strong acid ions compared with the other lakes. As mention before, this lake has the longest residence time (3,18 yr), and inflowing elements from the burned area stayed much longer in the system.
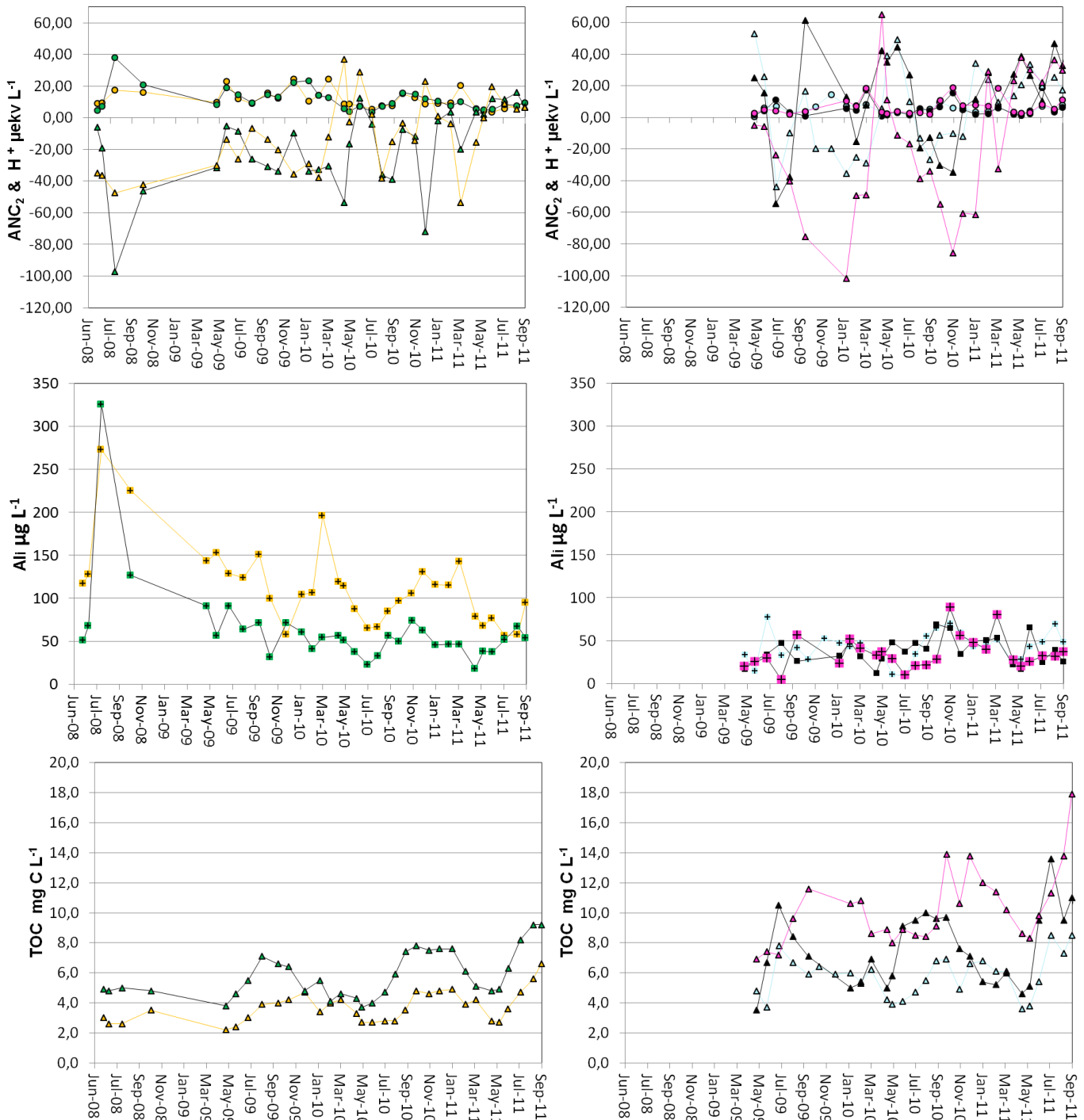

Figure 14: Time series plots of H^+ µekv L^{-1}, ANC_2 µekv L^{-1}, Al_i µg L^{-1} and TOC mg C L^{-1} in Lake Rasvassvatn (pink) and Lake Hundsvatn (green) (right panel), and the reference lakes (left panel) Melestjenn (pink), Jordtjenn (blue) and Svarttjenn (black)

The TOC concentration increased during autumn and decreased during summer in Lake Rasvassvatn and Lake Hundsvatn (Figure 14), which indicate natural seasonal variations probably caused by changes in biological production and water flows. The peak during autumn has slightly increased for each year after the fire in both of the lakes. This small trend can be seen in most of the wildfire affected lakes (Appendix 2). Lake Melestjenn had a larger variation then other reference lakes, which might be caused of a larger agricultural area compared with the other lakes (5,2%), or other site specific characters. Figure 15 further show that Colour and UV most often followed the same pattern. Colour and UV declined when the catchment had snow cover and increased during snow melt. The humic substances that reach the lakes during snowmelt gradually dissolves and dilutes during the summer period creating clearer water and during autumn new discharge of leafs and other organic particles reaches the lakes.
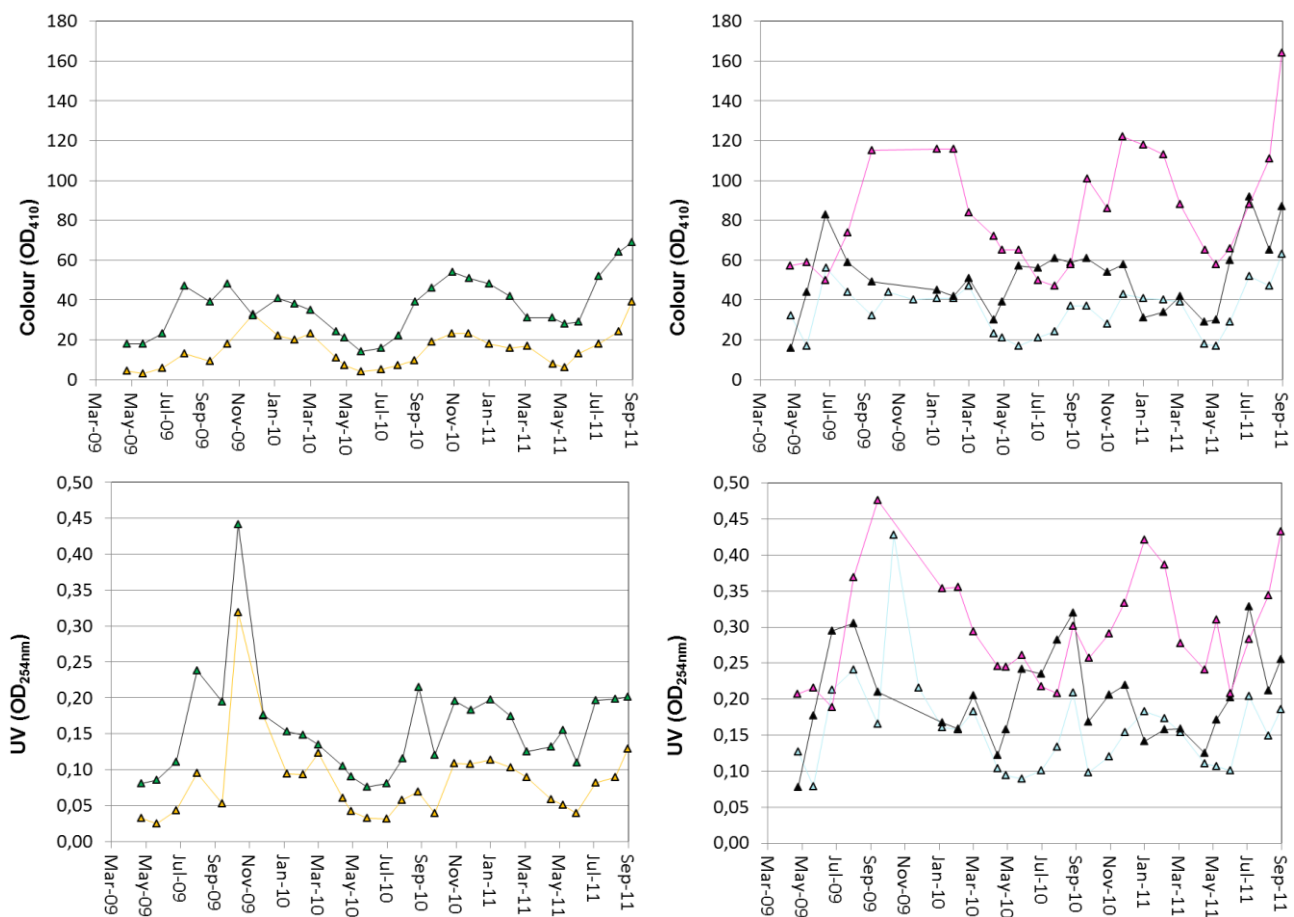

Figure 15: Time series plots of Colour and UV in Rasvassvatn (yellow) and Hundsvatn (green) (right panel), and the reference lakes (left panel) Melestjenn (pink), Jordtjenn (blue) and Svarttjenn (black)

The colour/TOC ratios do also follow the same pattern similar as TOC, with lower values during summer and high concentrations during autumn. The high colour concentration in relation to TOC concentration during autumn might have been caused by higher molecular size of the organic matter and lower concentration of dissolved organic matter during that period.
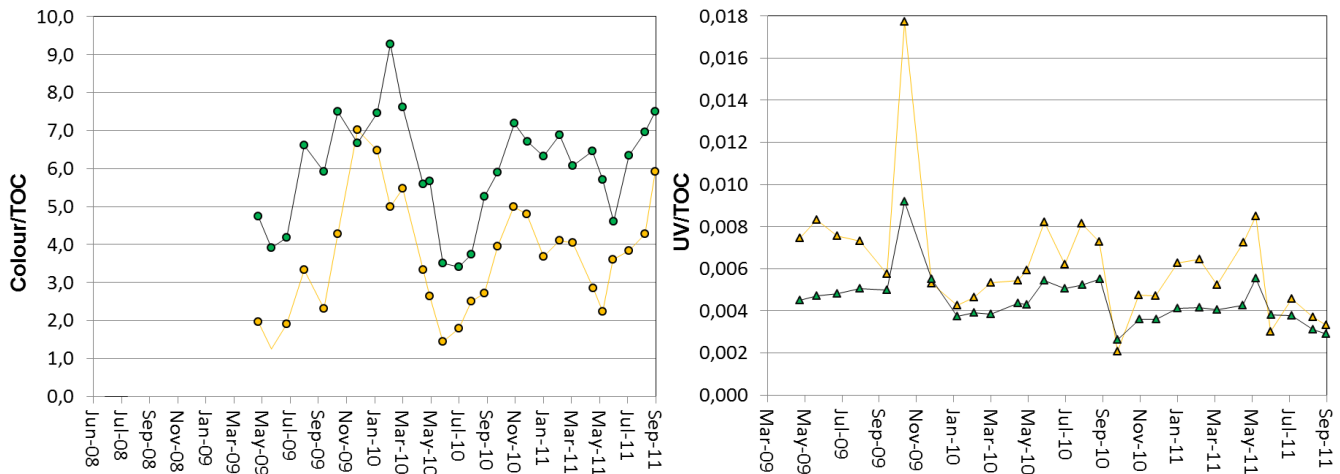

Figure 16: Time series plot of Colour/TOC ratio and UV/ TOC ratio in Rasvassvatn (yellow) and Hundsvatn (green).

7.2 Normality

Normality was analysed in relation to measured parameters in each lake and measured parameters in each month, and for average concentrations (Appendix 5). The results indicate that the normality of the parameters is higher in relation to month than to specific sampling sites, except for aluminium fractions and organic parameters. Chloride, nitrate, calcium, sodium, potassium and ammonium has a low normal distribution in relation to sampling site, which leads to that these parameters should as much as possible be analysed with non-parametric methods. The normality in relation to months is considered to be sufficient in order to use parametric methods in all cases except ammonium, Ali, TOC, UV and colour (Table 8 & 9).

Table 8: Normal distribution of inorganic parameters in relation to sampling points and in relation to sampled months.

	SO_4^{2-} mg L^{-1}	Cl$^-$ mg L^{-1}	NO_3^- µg L^{-1}	Ca^{2+} mg L^{-1}	Mg^{2+} mg L^{-1}	Na^+ mg L^{-1}	K^+ mg L$^-$	NH_4^+ µg L^{-1}
Sampling points	10/12	2/12	2/12	5/12	7/12	5/12	2/12	-
Months	10/12	6/12	8/12	11/12	11/12	9/12	9/12	4/12

Table 9: Normal distribution of organic parameters, ANC, aluminium species, hydrogen ions and conductivity in relation to sampling points and sampled months.

	H^+ µekv L^{-1}	Cond. µS/cm	Ali µg L^{-1}	Alo µg L^{-1}	Ala µg L^{-1}	TOC mg C L^{-1}	UV OD$_{410nm}$	Colour OD$_{254nm}$	ANC1 µekv L^{-1}	ANC2 µekv L^{-1}
Sampling points	3/12	5/12	10/12	9/12	9/12	11/12	6/12	7/12	12/12	12/12
Months	9/12	11/12	4/12	6/12	9/12	4/12	4/12	3/12	12/12	12/12

The statistical methods used further in this paper will be considering the normality of the parameters. In order to create consistency in the method no result has been taken away in order to create normality, since it cannot be certain that outliers are not signs of real fluctuations in the water chemistry, moreover no normality plot would have changed to normality if only a few outliers were taken away.

7.3 Time trends
Estimated yearly changes during period May 2009- May 2011

With the method linear regression a yearly average decrease in SO_4^2 were estimated to be -0,99 mg L^{-1} yr^{-1}, and -0,93 mg L$^-$1 yr^{-1} based on median values in the wildfire affected lakes during the period May 2009 – May 2011 (Table 10). The fired lakes showed different declining SO_4^{2-} patterns, but all the trends were significant (Table 11). However Lake Hundsvatn (9m) was not normal distributed. Lake Jordtjenn was the only reference lake with significant yearly decrease in SO_4^2, which might be due to the fact that a small part of its catchment was affected by the fire. The chloride concentration had average significant increase of 0,28 mg L^{-1} yr^{-1} in the reference lakes, but only Lake Jordtjenn had individually a significant increase (0,27 mg L^{-1} yr^{-1}). Lake Hundsvatn had a significant decrease of -22,2 µg L^{-1} yr^{-1} NO_3^-, most of the other lakes was nonparametric. The median concentration of Mg^{2+} increased significantly with 0,04 mg L^{-1} yr^{-1} in the reference lakes, while the trend decreased nonsignificant in the fired lakes. Lake Rasvassvatn had a significant decrease of -0,07 mg L^{-1}

Mg^{2+} yr^{-1} in hypolimnion. No other yearly trends can clearly be stated for the inorganic parameters due to the high quantity of nonparametric data. It is however visible to see that several ions probably have a declining trend per year in several fired lakes.

Table 11: Linear regression of average and median concentrations in fired lakes and reference lakes.

Parameters	SO$_4^{2-}$ mg L^{-1}		Cl$^-$ mg L^{-1}		NO$_3^-$ µg L^{-1}		Ca^{2+} mg L^{-1}		Mg^{2+} mg L^{-1}		Na$^+$ mg L^{-1}		K$^+$ mg L^{-1}		NH$_4^+$ µg L^{-1}	
	const. coef.	Δ yr	const. coef.	Δ yr	const. coef.	Δ yr	const. coef.	Δ yr	const. coef.	Δ yr	const. coef.	Δ yr	const. coef.	Δ yr	const. coef.	Δ yr
Fire Lakes (average)	4,88	-0,99***	2,79	-0,28	111	-5,51	1,09	-0,06	0,35	-0,02	1,79	-0,09	0,86	-0,13*	190	-37.23
Reference Lakes (average)	3,70	-0,05	2,62	0,28*	52,2	14,45	1,18	-0,05	0,22	0,03	1,76	0,20*	0,24	0,10*	119	-7,56
Fire Lakes (median)	4,79	-0,93***	2,81	-0,12	107	-2,48	1,09	-0,11	0,35	-0,02	1,80	-0,1	0,86	-0,15***	172	-34,2
Reference Lakes (median)	3,59	-0,42	2,38	0,20	55,7	1,28	1,25	-0,09	0,21	0,04*	1,67	0,13	0,26	0,09*	133	-25,22

*p-value ≤ 0,05, **p-value ≤ 0,01, ***p-value ≤0,005,   \   = assumption of normality is violated

Table 10: Linear regression of inorganic parameters in the lakes.

Lake	SO$_4^{2-}$ mg L^{-1}		Cl$^-$ mg L^{-1}		NO$_3^-$ µg L^{-1}		Ca^{2+} mg L^{-1}		Mg^{2+} mg L^{-1}		Na$^+$ mg L^{-1}		K$^+$ mg L^{-1}		NH$_4^+$ µg L^{-1}	
	const. coef.	Δ yr	const. coef.	Δ yr	const. coef.	Δ yr	const. coef.	Δ yr	const. coef.	Δ yr	const. coef.	Δ yr	const. coef.	Δ yr	const. coef.	Δ yr
Heitjenn	4,68	-1,33***	2,56	-0,04	118	-10,59	0,78	-0,12	0,26	-0,02	1,62	-0,06	0,81	-0,14*	398	-158,78**
Grunnetjenn	4,92	-1,08***	2,66	-0,11	135	-30.3	1,24	-0,07	0,41	-0,04	1,78	-0,17	0,91	-0,20***	109	2,48
Hundsvatn 1m	4,90	-1,05***	2,78	-0,16	103	-1,28	1,05	-0,11	0,33	-0,03	1,74	-0,13	0,87	-0,17***	174	46,72
Hundsvatn 9m	7,98	-2,88***	4,12	-1,17***	97,5	-22,2*	1,52	-0,42***	0,49	-0,13***	2,30	-0,50***	1,35	-0,45***	313	-125,20***
Fisketjenn	4,64	-0,92***	2,90	0,04	87,9	10,8	0,9	-0,05	0,31	-0,003	1,80	-0,06	0,73	-0,06	245	-62,05
Øyvatn	4,49	-0,51*	2,68	0,04	53,7	29,2	1,60	-0,01	0,37	-0,02	1,90	-0,05	0,86	-0,10	149	-5,48
Rasvassvatn 1m	5,57	-1,00***	3,16	-0,07	112	-6,94	1,15	-0,10	0,44	-0,03	2,38	-0,34	1	-0,10	130	4,38
Rasvassvatn 14m	7,88	-1,83***	4,28	-0,71***	76	-23,58	1,29	-0,26***	0,52	-0,07*	2,00	-0,12**	1,27	-0,19*	446	-66,43
Jordtjenn 1m (Ref)	3,47	-0,46*	2,24	0,27*	60,5	-2,66	1,21	-0,14	0,22	0,03	1,60	0,12	0,28	0,11*	120	-34,68*
Jordtjenn 7m (Ref)	4,42	-1,09***	2,67	0,07	84,6	-15,95	1,01	-0,01	0,24	0,02	1,98	-0,15	0,43	0,00	202	-89,06***
Svartjenn (Ref)	3,56	-0,41	2,23	0,09	52	9,38	1,42	-0,18	0,26	0,01	1,59	0,14	0,25	0,07	107	-27,01
Melestjenn (Ref)	4,10	-0,51	3,47	0,42	22,2	48,55	1,21	-0,02	0,19	0,06*	2,20	0,27	0,13	0,16*	125	-12,45

The average concentration of H$^+$ decreased significantly with -3,61 µekv L^{-1} yr^{-1} in the fired lakes, Al$_i$ decreased significant with -13,03 µg L^{-1} yr^{-1} in average and -13,1 µg L^{-1} yr^{-1} in median, and the average ANC$_1$ concentration increased with 11,09 µekv L^{-1} yr^{-1} (Table 12). These changes are indicating large improvements in the water quality in the burned lakes. The improvements are specifically visible in Lake Hundsvatn with decrease of -3,87 µekv L^{-1} H$^+$ yr^{-1}, -3,18 µS cm^{-1} yr^{-1} conductivity, and -16,97 µg L^{-1} Al$_i$ yr^{-1} at 1m depth (Table 13). Lake Hundsvatn had further significant increase in ANC$_1$ and ANC$_2$ with 29,82 µekv L^{-1} yr^{-1} and 26,46 µekv L^{-1} yr^{-1} respectively in hypolimnion and Lake Jordtjenn had also significant increase of ANC$_1$ and ANC$_2$ of 17,08 µekv L^{-1} yr^{-1} and 15,44 µekv L^{-1} yr^{-1} at hypolimnion, which indicating large decrease of strong acid ions in these lakes.

Significant decrease of aluminium fractions was found in Lake Fisketjenn at 1 m depth (-13,36 µg L^{-1} Al_i yr^{-1}) and Lake Rasvassvatn at hypolimnion (-43,43 µg L^{-1} Al_a yr^{-1}). Fisketjenn had a significant decrease of Al_i and nonsignificant increase of Al_o which caused only a small decrease of the total monomeric aluminium fractions, while Lake Rasvassvatn have a much larger nonsignificant decrease of Al_o, a small decrease of Al_i and the largest significant decrease of Ala compared to all other lakes. Lake Fisketjenn has a significant increase of 1 mg C L^{-1} yr^{-1} TOC, which increases the organically bound aluminium fractions in relation to Al_i. In the case of Lake Rasvassvatn it is more likely that some of the aluminium fraction has left the system, however Lake Rasvassvatn has the largest constant coefficient for Al_a in hypolimnion which might suggest that this lake still has large quantities of reactive aluminium fractions in the system.

Table 12: Linear regression of average and median concentrations in fired lakes and reference lakes.

Parameters	H+ µekv L⁻¹ const. coef.	Δ yr	Conductivity µS/cm const. coef.	Δ yr	Ali µg L⁻¹ const. coef.	Δ yr	Alo µg L⁻¹ const. coef.	Δ yr	Ala µg L⁻¹ const. coef.	Δ yr	ANC1 µekv L⁻¹ const. coef.	Δ yr	ANC2 µekv L⁻¹ const. coef.	Δ yr
Fire Lakes (average)	14,8	-3,61*	26,5	-2,45	67,1	-13,03*	56	2,92	123	-10,11	-4,86	11,09*	-19,3	9,13
Reference Lakes (average)	5,95	0,02	18,4	2,16	32,4	6,13	101	3,83	134	9,93	9,09	9,6	-14,4	8,4
Fire Lakes (median)	14,8	-3,36	26,4	-2,55	60,2	-13,1*	52,8	6,75	113	-7,52	-6,92	9,34*	-20,7	6,94
Reference Lakes (median)	6,45	-0,46	18,8	0,91	31,8	6,53	99,1	0,51	139	0	7,6	11,79	-13,8	12,15

*p-value ≤ 0,05, **p-value ≤ 0,01, ***p-value ≤0,005, \ = assumption of normality is violated

Table 13: Linear regression of H^+, conductivity, aluminium species and ANC in the lakes.

Lake	H+ µekv L⁻¹ const coef.	Δ yr	Conductivity µS/cm const coef.	Δ yr	Ali µg L⁻¹ const coef.	Δ yr	Alo µg L⁻¹ const coef.	Δ yr	Ala µg L⁻¹ const coef.	Δ yr	ANC1 µekv L⁻¹ const coef.	Δ yr	ANC2 µekv L⁻¹ Const coef.	Δ yr
Heitjenn	19,9	-3,08	26,4	-3,38	67,0	-14,97	36,0	10,91	103	-4,02	-26,6	17,96*	-38,9	13,94
Grunnetjenn	8,17	2,18	24,5	-2,52	44,6	-6,57	37,5	10,59	82,1	4,02	8,43	8,25	-4,3	6,39
Hundsvatn 1m	15,3	-3,87*	25,6	-3,18**	70,4	-16,97***	71,7	1,28	142	-15,7	-10,4	9,05	-27,5	7,19
Hundsvatn 9m	16,1	-3,42	37,5	-10,55***	100	-34,75***	84,4	6,21	185	28,54***	-38,9	29,82***	-53,9	26,46***
Fisketjenn	22,3	-6,42***	27,0	-2,41	52,8	-13,36**	38,0	8,91	90,8	-4,45	-17,7	10,48	-32,2	7,08
Øyvatn	1,60	0,55	23,6	0,71	14,5	5,48	39,1	9,60	53,6	15,11	41,6	3,72	23,9	3,58
Rasvassvatn 1m	15,5	-3,47	29,4	-2,36	131	-19,49	67,9	2,41	199	17,08	-7,83	8,21	-19,0	7,12
Rasvassvatn 14m	7,03	0,00	39,0	-6,21***	62,3	-6,79	261	-36,87	323	-43.43*	-47,2	21,39	-68,8	21,02
Jordtjenn 1m (Ref)	5,96	-0,77	17,7	0,97	38,8	2,26	91,3	-5,33	130	-3,03	15,2	5,55	-4,5	6,42
Jordtjenn 7m (Ref)	6,86	-0,71	20,8	-0,49	61,5	-6,02	85,4	7,01	147	1,02	-6,36	17,08*	-22,4	15,44*
Svartjenn (Ref)	5,40	-0,35	17,9	1,19	30,4	7,19	112	-4,12	142	3,07	26,7	5,15	1,7	6,39
Melestjenn (Ref)	4,42	2,28	18,8	4,75*	24,9	10,51	102	20.56	127	31,1	-9,6	15,22	-38,0	10,99

No yearly median and average decreases could significantly be proven for the organic parameters (Table 14), but three of the fired lakes have significant yearly increases of TOC (Table 15). Lake Hundsvatn had further significant increases of UV, colour and the Colour/TOC fraction. The large increase of colour in Lake Hundsvatn (12,19 OD_{254}), might be caused of an increase of dissolved humic particles (DOC) in relation to humic matter of lager molecular size.

Table 14: Linear regression of average and median concentrations in fired lakes and reference lakes.

Parameters	TOC mg C L⁻¹		UV OD410nm		Colour OD254nm		UV/TOC		Colour/TOC		UV/Colour	
	const. coef.	Δ yr	const. coef.	Δ yr	const. coef.	Δ yr	const. coef.	Δ yr	const. coef.	Δ yr	const. coef.	Δ yr
Fire Lakes (average)	4,25	0,57	0,12	-0,003	19,7	3,98	0,03	-0,002	4,31	0,35	0,01	-0,001
Reference Lakes (average)	6,91	0,36	0,24	-0,02	51,3	1,13	0,03	-0,003	7,18	-0,27	0,005	-0,0004
Fire Lakes (median)	4,13	0,63	0,11	0,001	18,5	4,49	0,03	-0,002	4,20	0,034	0,01	-0,001
Reference Lakes (median)	6,91	-0,07	0,23	-0,03	48,2	-2,34	0,03	-0,004	7,14	-0,36	0,005	-0,0004

*p-value ≤ 0,05, **p-value ≤ 0,01, ***p-value ≤0,005, \ = assumption of normality is violated

Table 15: Linear regression of organic parameters in the lakes.

	TOC mg C L⁻¹		UV OD410nm		Colour OD254nm		UV/TOC		Colour/TOC		UV/Colour	
	const coef.	Δ yr	const coef.	Δ yr	const coef.	Δ yr	const. coef.	Δ yr	const. coef.	Δ yr	const. coef.	Δ yr
Heitjenn	3,62	1,18*	0,07	0,03	15,2	5,91	0,02	0,002	3,68	0,4	0,005	0
Grunnetjenn	3,75	0,54	0,06	0,02	12,1	4,93	0,02	0,003	3,05	0,77	0,006	0
Hundsvatn 1m	5,04	0,55	0,16	-0,01	28,8	4,67	0,03	-0,003	5,63	0,29	0,005	-0,001
Hundsvatn 9m	4,40	0,98***	0,08	0,05***	20,1	12,19***	0,02	0,005**	4,7	1,25***	0,004	0
Fisketjenn	4,25	1,00**	0,07	-0,03*	15,4	8,76*	0,02	0,003	3,58	0,81	0,005	0
Øyvatn	5,23	0,04	0,12	0,01	24,1	4,16	0,02	0,002	4,7	0,66	0,005	0
Rasvassvatn 1m	3,27	0,32	0,09	-0,01	11,8	1,68	0,02	-0,002	3,24	0,26	0,004	0
Rasvassvatn 14m	6,35	0,11	0,14	-0,004	40,9	-12,81	0,02	0,001	5,89	-1,79*	0,003	0,003*
Jordtjenn 1m (Ref)	5,79	-0,25	0,19	-0,03	38,1	-4,64	0,03	-0,004	6,46	-0,55	0,005	0
Jordtjenn 7m (Ref)	4,72	-0,48*	0,14	0,02	31,1	2	0,03	0,001	6,57	-0,18	0,004	0
Svartjenn (Ref)	7,35	-0,37	0,21	-0,02	51,9	-4,75	0,03	0	6,9	-0,19	0,004	0
Melestjenn (Ref)	8,36	1,24	0,28	0,02	72,4	7,85	0,03	-0,001	8,49	-0,25	0,004	0

Difference between second and third post-fire year

Wilcoxon matched pair test shows a significant decrease of SO_4^{2-} in all lakes during postfire year 3 compared with the postfire year 2. The burned lakes has an average median decrease of -1,22 mg L^{-1} at 1m depth, while the reference lakes has an average median decrease of -0,76 mg L^{-1} at 1m depth. The largest decreases was found at hypolimnion in the stratified lakes, and the reasons for that might be that the water quality in this layer are linked to depositions and concentrations in surface water during the previous mixing period. This might also be the reason for the significant decrease of Cl^- at hypolimnion in Lake Rasvassvatn and Lake Hundsvatn (Table 16).

 The median calcium concentration decreased significantly with -0,23 mg L^{-1} at hyplomnion in lake Rasvassvatn and Hundsvatn in the third year compared with the second year. This was probably caused by retained levels of calcium from the large input of ions during the rain event after the fire in Lake Hundsvatn and an increase of the element after spring flood 2010 in Lake Rasvassvatn (see Table 7 and Figure 10). Lake Hundsvatn further decreased with -0,57 mg L^{-1} Na^+ in hyplomnion, however no other lakes showed any significant change. Lake Hundsvatn has the largest catchment area (2,63 km^2) compared with the other burned lakes, which might cause of larger inflow of Na^+ after the fire episode.

 The potassium concentration increased significant in Lake Jordtjenn and showed only a significant decrease of -0,42 mg L^{-1} in hyplomnion in Lake Hundsvatn with Wilcoxon matched pair test. This indicates that there was no clear decrease of K^+ in the other burned lakes during the postfire years 2 and 3. The ammonium concentration decreased significantly with -114 µg L^{-1} in Lake Heitjenn, -82,2 µg L^{-1} in Lake Fisketjenn and with -99,2 µg L^{-1} in hyplomnion in Lake Jordtjenn. It does not seem to be any clear trends distinguishing the burned lakes from the reference lakes when it comes to ammonium concentrations.

Table 16: Wilcoxon matched pair test, inorganic parameters.

Lake	SO_4^{2-} mg L^{-1}		Cl^- mg L^{-1}		NO_3^- µg L^{-1}		Ca^{2+} mg L^{-1}		Mg^{2+} mg L^{-1}		Na^+ mg L^{-1}		K^+ mg L^{-1}		NH_4^+ µg L^{-1}	
	n	median dif.	n	median dif.	n	median dif.	n	median dif.	n	median dif.	n	median dif.	n	median dif.	n	median dif.
Heitjenn	8	-2,1**	9	-0,02	9	-36,5	9	-0,09	9	-0,05	9	-0,15	9	-0,13	9	-114***
Grunnetjenn	9	-1,3***	9	-0,02	9	-43	9	0,17	9	-0,06	9	-0,15	9	-0,12	9	15,6
Hundsvatn 1m	11	-1,13*	11	-0,1	11	-23,5	11	0,01	11	-0,04	11	-0,05	11	-0,05	10	-63
Hundsvatn 9m	9	-3**	9	-1,22*	9	-13,4	9	-0,23***	9	-0,14	9	-0,57*	9	-0,42*	9	-73,1
Fisketjenn	9	-1,2*	9	0,02	8	-20,9	9	-0,01	9	-0,02	9	-0,16	9	-0,07	8	-82,2*
Øyvatn	8	-0,98*	8	0,18	8	-12,3	8	-0,02	8	-0,06	8	-0,1	8	-0,05	8	-13,3
Rasvassvatn 1m	11	-1,52**	11	-0,04	11	-34,6	11	0,0	11	-0,04	11	-0,2	11	-0,13	10	-26,9
Rasvassvatn 14m	9	-2,2***	9	-0,6*	9	-28,5	9	-0,23*	9	-0,1	9	-0,38	9	-0,2	8	-38,7
Jordtjenn 1m Ref.	11	-0,61**	11	0,34*	11	-15,8	11	-0,01	11	0,02	11	0,19	11	0,13*	11	-26,3
Jordtjenn 7m Ref.	8	-1,19*	8	0,08	8	-18	8	-0,03	8	0,01	8	-0,11	8	0,02	8	-99,2*
Svattjenn Ref.	9	-0,72*	9	0,13	9	1	9	-0,18	9	0,03	9	0,01	9	0,05	9	-30,8
Melestjenn Ref.	9	-0,95***	9	0,43	9	-8	9	0,01	9	0,03	9	0,1	9	0,07	9	36

*p-value ≤ 0,05, **p-value ≤ 0,01, ***p-value ≤0,005*

The median concentration of hydrogen ions decreased significantly in 3 of the 6 burned lakes, approximately with -5,49 µekv L^{-1} in Lake Hundsvatn, with -5,67 µekv L^{-1} in lake Fisketjenn and with -3,52 µekv L^{-1} in Lake Rasvassvatn (Table 17). This can likely be linked to a recovery from the acidic deposition after the fire. 4 of the 6 burned lakes had also significant decreases of conductivity, which further shows that the overall concentration of inorganic dissolved solids was higher during the second year compared with third year after the fire in most of the burned lakes.

The aluminium fractions did not show any significant change during the second and third year after the fire in the fired lakes, the reference lake Svarttjenn had however a significant increase of 12,4 µg L^{-1} Al_i in the third year compared with the second year. Lake Hundsvatn had significant increases of TOC, UV and colour at hyplomnion, while Lake Jordtjenn had a significant decrease of UV and Lake Meletjenn had a significant increase of TOC. There were furthermore significant increases of ANC_1 and ANC_2 in Lake Heitjenn, Lake Grunnetjenn and Lake Hundsvatn (hyplomnion). Wilcoxon matched pair test only indicated a significant median increase of ANC_1 in Lake Rasvassvatn, paird t-test showed however a significant increase also in ANC_2 (Appendix 6).

Table 17: Wilcoxon matched pair test of hydrogen ions, conductivity, aluminium fractions, organic parameters and ANC.

Lake	n	H⁺ µekv L⁻¹	Cond. µS/cm	Ali µg L⁻¹	Alo µg L⁻¹	Ala µg L⁻¹	TOC mg C L⁻¹	UV OD₄₁₀nm	Colour OD₂₅₄nm	ANC1 µekv L⁻¹	ANC2 µekv L⁻¹
		median dif.	median dif.	median dif.	median dif.	median dif.	median dif.	median dif.	median dif.	median dif.	median dif.
Heitjenn	9	-3,06	**-5,7****	-17	-1	-21,9	0,55	-0,0002	-2,5	**31,3****	**29.9****
Grunnetjenn	9	-2,74	**-3,4***	-15,8	-2,8	-22,8	0,3	0,004	-0,55	**27,9*****	**26,1*****
Hundsvatn 1m	11	**-5,49****	**-3,3***	-19	-1,9	-22,6	0,68	-0,01	1,5	19,4	17,3
Hundsvatn 9m	9	-1,33	**-10,35***	-30,2	14,7	-23,1	**0,7****	**0,03****	**8***	**34,6*****	**32,4****
Fisketjenn	9	**-5,67*****	-4,45	-11,4	1,9	-13,6	0,55	-0,003	1,6	12	9
Øyvatn	8	-0,04	-1,15	-2,5	-1	-0,4	0,25	-0,01	1,25	5,9	5,5
Rasvassvatn 1m	11	**-3,52*****	**-3,75*****	-26,4	-7,4	-32,6	0,15	-0,01	-1,6	15,4	15,2
Rasvassvatn 14m	9	0,32	**-6,15***	-15,6	-24,9	-40,2	0,6	-0,05	-17	**34,3***	31,3
Jordtjenn 1m Ref..	11	-1,72	0,65	1,3	-5,4	-3,9	-0,25	**-0,05***	-6,5	17,3	18,3
Jordtjenn 7m Ref.	8	-0,51	-0,78	-0,5	8,8	7,5	0,20	-0,004	-0,75	19,8	17,8
Svattjenn Ref.	9	-1,3	19	**12,4*****	7,4	19,1	0,75	-0,01	-1	13,4	9,7
Melestjenn Ref.	9	-0,27	**1,6*****	3,8	9,6	7,8	**1,1***	0,01	-1,5	18,7	13,7

*p-value ≤ 0,05, **p-value ≤ 0,01, ***p-value ≤0,005

8 Final Discussion

Other studies have observed increases of both cations and anions in surface waters after fire events. Williams and Melack (1997) found that the concentration of the cations; Na^+, K^+, Ca^{2+} and Mg^{2+} increased by a factor of 1.3, 1.8, 3.0 and 2.9 respectively during the first postfire year in a stream in Sierra Nevada, Carlifornia. Bayley (1992) estimated factor increases of 1.74, 3.08, 1.89 and 1.89 respectively for the same cations in a study in a stream in northwestern Ontario in Canada. A study conducted by Eriksson et al (2003) did further measure elevated concentrations of mostly Na^+, Ca^{2+} and K^+ after the fire, and Nordland (2011) concluded among other things that the concentrations of Na^+ was approximately less than 10% higher after the fire and the concentration of K^+ was 71-85% higher in streamwater during the first year compared to a reference creek. In common for some of these studies is however a larger increases of strong acid anions in relation to cations after fires in surface waters. Sulphate concentrations increased in streams by factors of 16, 3.8 and 2.7 and chloride increased by factors of 4, 3.9 and 4 during the first year in previous studies (Table 18) (Bayley et al. 1992, Williams and Melack 1997, Mast and Clow 2008).

Table 18: Documented factor increase of some ions during post-fire year 1, and factor increase in Rasvassvatn (R) and Hundsvatn (H) during the large rain event in August 2008.

Article	SO_4^{2-}	Cl^-	NO_3^-	Na^+	K^+	Ca^{2+}	Mg^{2+}
Mykland, Norway Factor increase two months after fire **Lakes** *(Mykland)*	H: 4,6 R: 1,3	H: 2,53 R: 1,49	H: 4,4 R: 1,77	H: 1,84 R: 1,28	H: 5,25 R: 1,81	H: 3,08 R: 1,55	H: 3,94 R: 1,92
William and Melack (1997) Factor increases first postfire year in **Stream** *(Tharp Creek)*	16	4	2	1.3	1.8	3.0	2.9
Bayley et al. (1992) Factor increases first postfire year **Stream** *(NW stream)*	3.8	3.9		1.74	3.08	1.89	1.89
Mast and Clow (2008) Factor increases first postfire year in **Stream** *(Coal Creek)*	2.7	4	10				

The concentration of most of the anions and cations was also elevated in Mykland, Lake Rasvassvatn and Lake Hundsvatn peaked in august 2008 during intense rain event two months after the forest fire. It is however difficult to evaluate the total increase of ions. The concentrations measured during the two months before the rain event cannot be seen as

background values in these lakes since it rained before the larger rain event. It is therefore reasonable to believe that the fire had larger impact on the lake chemistry during the first months then showed by the factor increase during the large rain event. The lakes outside the fired area might also have been influenced by the fire. Lake Jordtjenn had a small part of its catchment within the fired area. It is also possible to suspect a wind dispersal of dust and ashes and a subsequent atmospheric transport and dry deposition after the combustion. The spread of dust and ashes are affected by topography, wind speed, season and the size of the fire (Lönnermark et al. 2007). Most studies dealing with this topic evaluate long transport of particles (Cheng et al. 1998, Hu et al. 2006), but atmospheric fallout of ash in close located reference lakes was also suspected by Gerla and Galloway (1998).

The lakes in Mykland responded differently after the fire event, however all lakes showed a decreasing trend in SO_4^{2-} concentration during the years after the fire. Mykland area has been suffering from acidification caused by long range transport of pollutions which probably have caused large accumulation of SO_4^{2-} in the bogs, which was released during the fire. It cannot be ruled out that atmospheric fallout of ash caused an increase of SO_4^{2-} in all lakes after the fire. This could explain the significant decrease of SO_4^{2-} three years after the fire compared with two years after the fire in all measured lakes (W-test), but the reference lakes have rather low residence time, which might limit the atmospheric impact from the burning event. The generally declining SO_4^{2-} trend might also be consequence of the continuous declining SO_4^{2-} inputs in this region. The yearly decrease of SO_4^{2-} was estimated to be larger and more significant in the fired lakes compared with the reference lakes (linear regression). This indicates that runoff from the burned catchment caused the largest fraction of the increase of SO_4^{2-}, and the subsequent decrease during the following years.

Changes in chloride concentrations except the peak in August 2008 were not statistically proven, except the decrease in hypoliminion in Lake Rasvassvatn and Lake Hundsvatn during post-fire year 3 and the yearly increases in the reference lakes, which might imply a difference between reference lakes and some of the fired lakes. Bayley et al (1992) measured elevated chloride concentration direct after the fire and the increase remained one year after the fire. They suggested that the increase was caused by the combustion of mosses. Williams and Melack (1997) and Mast and Clow (2008) thought that an increase of chloride concentrations was caused by burning of plant tissue. Since chloride is considered to be a rather mobile ion it might be reasonable to expect an increased concentration to leave the

catchments during the first spring melt, but in the study conducted by Mast and Clow (2008) was the chloride concentration elevated until the second year and in Williams and Melack (1997) study was the concentration elevated in 3 years. In this study had all lakes a decreasing pattern of Cl- after the spring melt during the third post-fire year (Appendix 2).

The fired lakes had slightly higher concentrations of NO_3^- than the reference lakes (Apendix 2 & 4). Bayley et al (1992) observed elevated concentrations of NO_3^- immediately after the fire and during following spring, which was suggested to be caused by enhanced nitrification. A slight higher concentration of total nitrogen was further found by Nordblad (2011) compared with a reference. This study cannot prove any enhanced nitrification in the burned catchments, and both the fired lakes and reference lakes had peaks of NO_3^- during spring 2010 and 2011. Since the catchments in Mykland were nutrient poor and the release of NO_3^- was limited, was the most of the released NO_3^- as well as NH_4^+ most likely retained by new vegetation.

Large increases of calcium has been found after fires in catchments with high quantities of the compound in the soil (Yang and Hai-qing 2007). The Mykland area is however located on a low weatherable bedrock and the quantities of calcium are therefore low in the catchments. All the lakes had low or very low lime content during the time they were monitored. The highest peak of 2,93 mg L^{-1} Ca^{2+} was however found in Hundsvatn during the large rain event in august 2008, which suggest that a release of calcium was enhanced direct after the fire.

No statistical significant changes of Mg^{2+} were found between the second and third post fire year. The fired lakes had however a slight higher concentration than the reference lakes (Appendix 2 & 4), as well as the consistent peak in august 2008. William and Melack (1997) continued to have elevated concentration during the third post-fire year and Bayley et al (1992) concluded that the concentration of cations continued to be higher than the pre-fire levels even 9 years after the forest fire. Nordblad (2011) suggested that magnesium might continue to be released from burnt plant remainders two years after a forest fire. This might also be the case in Mykland, since burned trees and other burnt organic matter was left in the burned catchments. Elevated concentration of K^+ is however more visible than changes of Mg^{2+} and Na^+ concentrations, seen in Appendix 2 and 4 there is large differences of K^+ concentrations between the reference lakes and the fired lakes. Potassium is an important nutrient for plant growth, and it is the most common cation in plant cells, plant dry weight

consist of up to 10% of the compound (Véry and Sentenac 2003). Nordblad (2011) concluded that an elevated concentration in K^+ in a burned creek likely could be linked to its high abundance and uptake in biological material. The concentration of K^+ in Mykland had no significant decrease between post-fire year 2 and 3 (W-test), there was however lower concentrations in spring 2011 which might imply a larger recovery of plant grow and subsequent uptake of K^+ by new vegetation. The decrease of K^+ and other ions during spring and summer 2011 was further most likely linked to the large snowmelt 2011. Runoff from the catchments was followed by peaks of ions, which most likely decreased the concentrations of ions stored in the catchments. It could explain a limited release of these ions during the next following months.

Median and average concentration of hydrogen was higher in the burned lakes compared with the reference lakes throughout the three post-fire years (Appendix 4). A small increase of H^+ was also documented by Bayley et al (1992) and by Nordblad (2011), while Williams and Melack (1997) observed a small insignificant decrease during the two first post-fire years. Hogberget mention that some of the lakes in Mykland was more acidified then other lakes prior to the fire, e.g was Lake Rasvassvatn more acid then Lake Hundsvatn (Hogberget et al. 2010). Bayley et al (1992) concluded that catchments with high amount of stored sulphur from past or large land areas of peat might have higher losses of H^+ and sulphur after fires. Lake Fisketjenn and Lake Hundsvatn has the largest areas of bogs compared with the other lakes, while Lake Rasvassvatn was more acidified then other lakes prior the fire. It is difficult to evaluate the losses of H^+ in the catchments since there is no available data prior to the fire and there is only data for Lake Rasvassvatn and Lake Hundsvatn during the first post-fire year. It is however visible to see that these lakes together with Lake Heitjenn had the largest increase of H^+ during the a peak in May 2009 (Appendix 2) (probably linked to the shorter drier period with no rain). The three lakes also had significant decreases of H^+ during post-fire year 3 (W-test), however the three lakes had not the highest input during peaks related to snow melt in the next coming year and they had not the largest decrease of sulphur compared with other lakes.

The leaching of ions further caused elevated conductivity after the fire. In this study had 4 of 6 fired lakes significant decrease of conductivity in post-fire year 3 compared with post-fire year 2, and the median and average concentration of conductivity is most divergent during post-fire year 1 and 2, compared with post fire year 3 (Appendix 4), which suggest that

conductivity might be close to pre-fire levels during the third post-fire year. This is in line with Nordblads (2011) results, which indicated that conductivity was elevated during two years after the fire.

The large leakage of acid ions during the rain event in August 2008 decreased the ANC value in the fired lakes. It is however difficult to further evaluate the ANC pattern in the lakes, since the average and median concentrations of ANC sometimes was higher in the fired lakes then in the reference lakes and vice versa, but the lower average values of ANC in the reference lakes might be caused by Lake Meletjenn, which suffering from road salting. Four of the fired lakes showed significant increases of ANC during post-fire year 3, and Lake Hundsvatn and Lake Rasvassvatn had higher ANC values in Jun 2011 then in Jun 2008 (before the large rain event). It cannot be ruled out that ANC values might be higher during the third post-fire year than pre-fire values. Williams and Melack (1997) documented an increased ANC value during the first post-fire year compared with pre-fire values, while Eriksson et al (2003) observed decreased alkalinity, and Bayley et al (1992) found a sharp decrease of ANC during 2 post-fire years. All lakes in Mykland have had ANC_2 values harmful for fish (lower than 8 µekv), but seen in Figure 17 there is increasing pattern by the end of 2011. The shifting ANC values within Mykland area was caused by the decrease of acid ions during the third post- fire year.
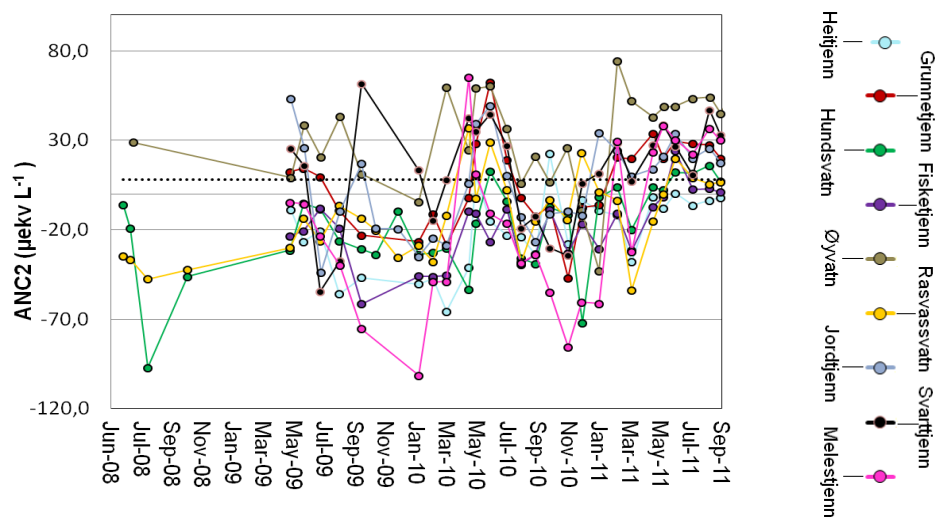
Figure 17: Time series plot of ANC$_2$ in the lakes, 95% of the fish not harmed at >8 µekv (dotted line).

There was no peak of TOC like the other parameters in august 2008, which probably was due to reduced carbon pool after the fire. The time series plots further indicated a slight increase of TOC for every year in all lakes after the fire, which might be linked to new biomass production after the fire and the reduced sulphate concentrations in all of the lakes, with subsequent decrease of acidification. W-test and T-test indicate significant increase of TOC in one of the burned lakes and one reference lake during the post-fire year 2 compared with the post-fire year 3. Regression indicated yearly increases in three of the burned lakes and one of the reference lakes. Most of the reference lakes had a slight larger constant coefficient then the fired lakes, suggesting that the reference lakes have a larger pool of TOC than the fired lakes. However, this is not certain. It is however likely that the burned catchment had a loss of carbon since carbon must have been lost through combustion during the fire. A slight decrease of TOC in burned catchments was also found by Nordblad (2011), but a slight higher concentration was found by Mast and Clow (2008), however Mast and Clow concluded that the increase in their study not was linked to the fire event. Eriksson et al (1999) documented a decrease of TOC and Colour direct after the fire, and a increase during high water flows. This shows that the effect of TOC discharge from burned catchments is rather complex and varies between different studies. Mykland area had a shallow soil cover and the fire was intense which made it possible for an extensive combustion of organic matter, it is likely that not all peats was burned, since it might lead to lower TOC levels than was observed in this study.

Aluminium is available in large amount in soils and biological material and thus released during forest fires. The concentration of TOC further affects the concentration of inorganic aluminium in the lakes. Both Lake Rasvassvatn and Lake Hundsvatn had large increase of Al_i during the rain even in August 2008. The average concentration of Al_i was substantially higher in the burned lakes then the reference lakes until approximately May 2010 (Appendix 4). There were no significant changes during post-fire year 3 compared with post-fire year 2, but two of the fired lakes had significant yearly decreases. The reference lakes and the burned lakes had however larger differences in Al_o concentrations which clearly shows that the reference lakes had a larger fraction of aluminium bound to organic matter, which are linked to the larger quantity of TOC in these lakes.

Short-term acidification caused by seasalt episodes or spring flood often cause biological degradation such as fish kills and lower taxonomic richness of invertebrates (Hindar et al. 1994, Lepori et al. 2003, Lydersen and Henriksen 2005). Forest fires might also create short-

term acidification. The fire in Mykland caused acute concentrations of toxic aluminium, increased leakage of hydrogen ions and further decreased the lakes buffer capacity which made the lakes more sensitive for other disturbances. The extreme Al_{in} concentrations (>250 µg L^{-1}) two months after the fire are considered lethal for fish in the effected lakes. The study has however not included any biological data which is needed for further evaluate the biological effect in the lakes. The area was suffering from acidification prior to the forest fire, and fish might have acclimatized to elevated metal concentrations and H^+ concentrations in the lakes, which might have limit the adverse effect of increased acidification. Furthermore was the extreme acidification event during a short period, which is known to limit the toxic effect (Mount et al. 1990). Lakes with pH $\leq$ 4,75 and >50 µg Al_i have often extinct or reduced fish populations (Lydersen et al. 1994). Figure 18 shows that the concentration was not permanently over these values in most of the lakes.
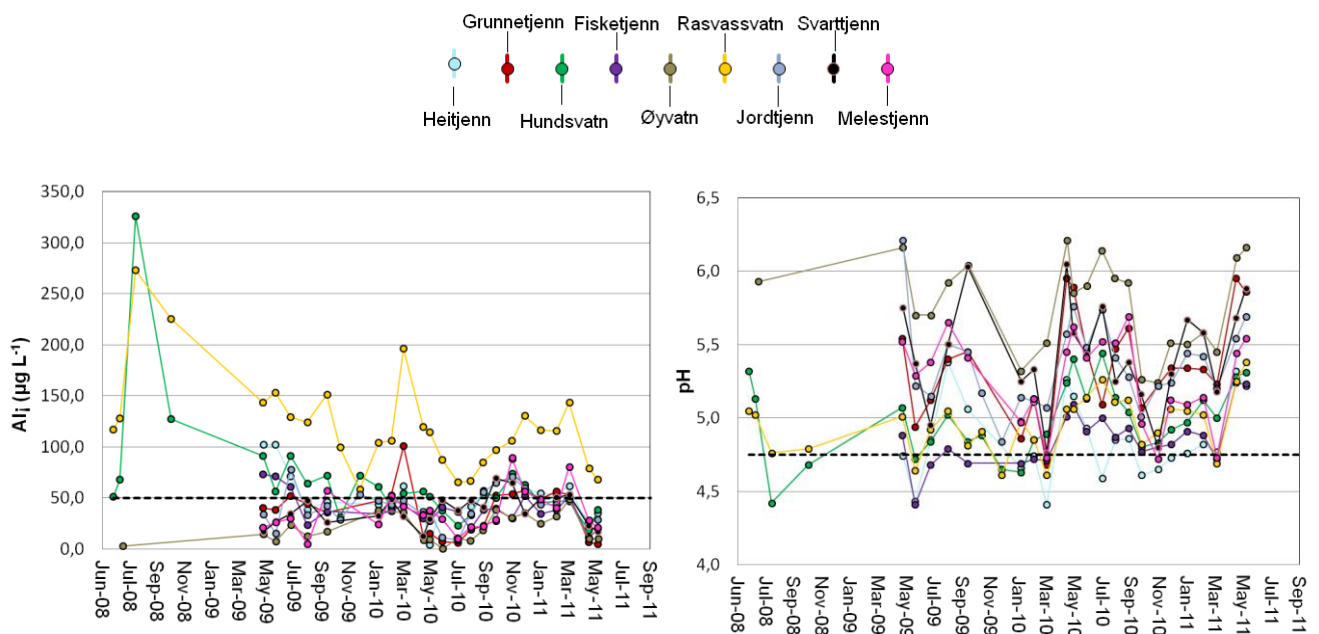
Figure 18: Time series plot of Al_i and pH in the lakes, when lakes have pH $\leq$ 4,75 and >50 µg Al_i (dotted line) it might indicate extinct or reduced fish population.

9 Conclusion

- The large rain event 2008 and the snow melt 2011 are important weather event causing release of ions from the fire affected catchments.

- All lakes were affected by atmospheric fallout of ash, or by the general decrease of SO_4^{2-} in this region.

- This study suggest in general a minor leach of Na^+ and NH_4^+, a substantial leach of Mg^{2+}, Ca^{2+}, Cl^-, and a large leach of K^+ and SO_4^{2-} in the fire affected lakes in Mykland. There is however differences among the lakes.

- The difference of leach pattern was most likely linked to catchment size, residence time, previous acidification, vegetation, bedrock and size of bogs.

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11 Appendix

11.1 Appendix 1: Comparison of old and new method

Correlation between unfrozen samples analyzed with IC and with the older methods showed that the calcium concentration correlated reasonably between the different techniques ($r = 0,88$, $p<0,0001$). On the other hand was the correlation very low for the frozen samples ($r = 0,38$, $p<0,0001$), which might indicate contamination of the frozen samples. The results are sometimes better for the frozen samples with larger sample size and sometimes better with the unfrozen samples with smaller sample size (Figure 19). The result from both frozen and unfrozen samples indicates that the methods are most likely comparable for potassium, magnesium and chloride. The result from the unfrozen samples gives further reasonable correlation for sodium, calcium and nitrate, whereas the correlation for sulphate and ammonium can be questionable for both the unfrozen and frozen samples.

Table 19: Pearson correlation between IC and old methods in unfrozen (a) and frozen (b) samples).

	Ca^{2+}		K^+		Mg^{2+}		Na^+		NH_4^+		Cl^-		SO_4^{2-}		NO_3^-	
	a	b	a	b	a	b	a	b	a	b	a	b	a	b	a	b
n	21	205	21	205	21	205	21	205	21	218	21	266	21	251	21	204
r- statistics	0,88	0,38	0,78	0,91	0,98	0,89	0,96	0,45	0,70	0,58	0,95	0,82	O,45	0,77	0,85	0,46
T-statistics	7,94	5,92	5,45	31,6	23	28,6	14,5	7,19	4,3	10,3	13,3	23,6	2,19	19,07	7,02	7,46
2-tailed p	<0,0001	<0,0001	<0,0001	<0,0001	<0,0001	<0,0001	<0,0001	<0,0001	<0,0001	<0,0001	<0,0001	<0,0001	0,0409	<0,0001	<0,0001	<0,0001

The figures below are showing results of IC plotted against the results of old methods. The result shows that the calcium concentrations are a bit higher with the old method than with IC in unfrozen samples. The IC gave further a bit higher concentration for K^+ and a substantial higher concentration of Mg^{2+}. The concentration of Na^+ was higher in the old method in the frozen samples, on the other hand was the concentration for the two methods similar in unfrozen samples. The result further indicates a higher concentration of SO_4^{2-}, Cl^- and NH_4^+ with the old method compared with IC.
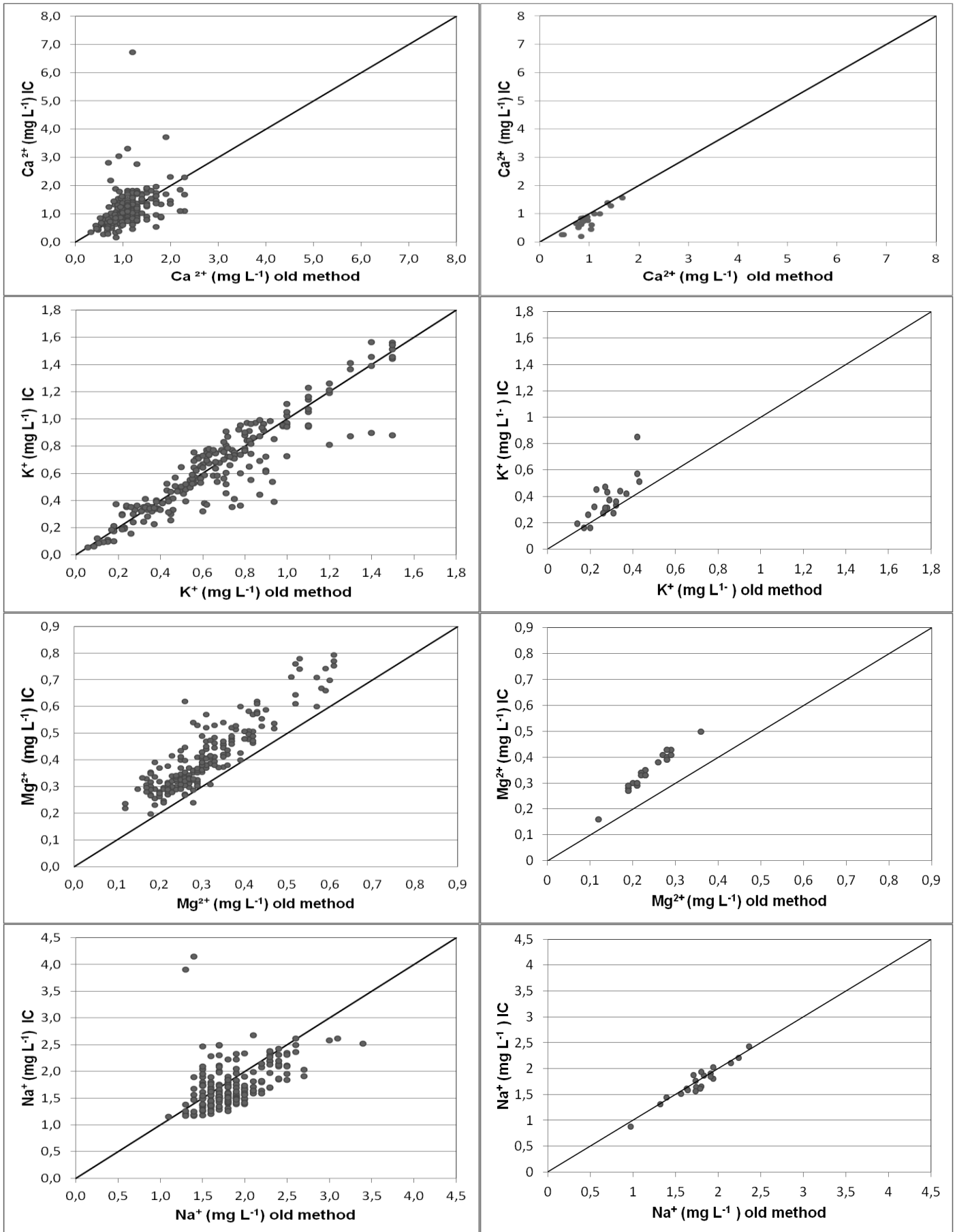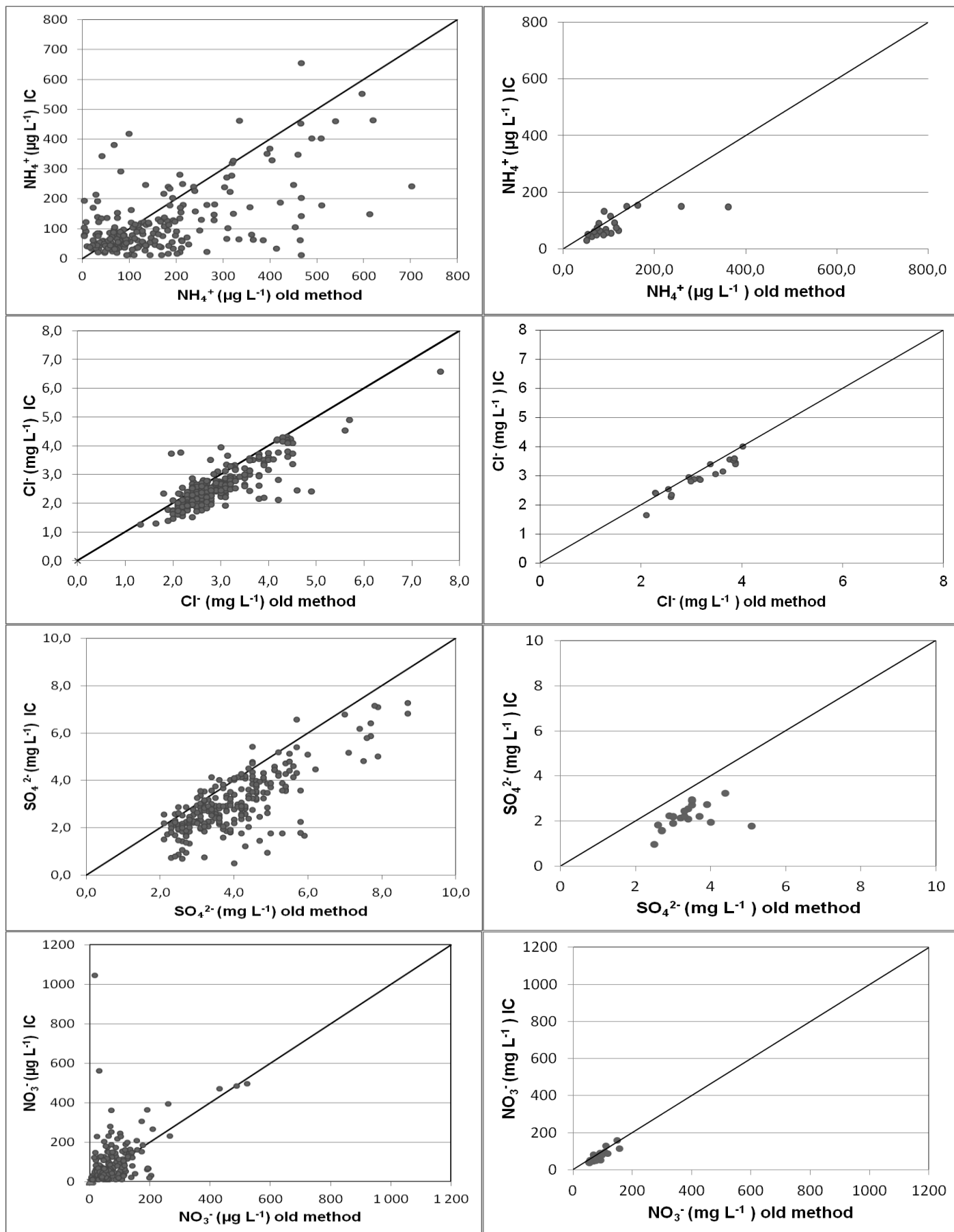
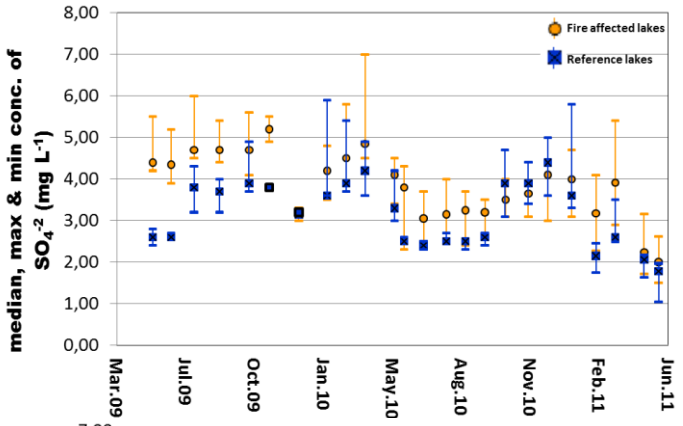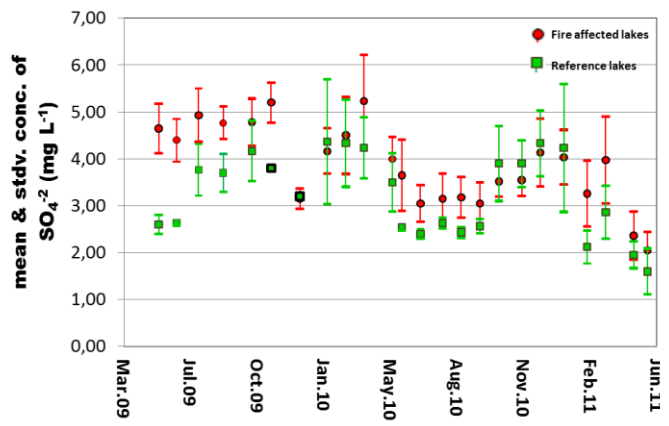*Figur 19: Comparison between IC and the old methods. At left frozen samples reanalyzed with IC, at right unfrozen samples reanlyzes with the old methods.*

11.2 Appendix 2: Time series plots
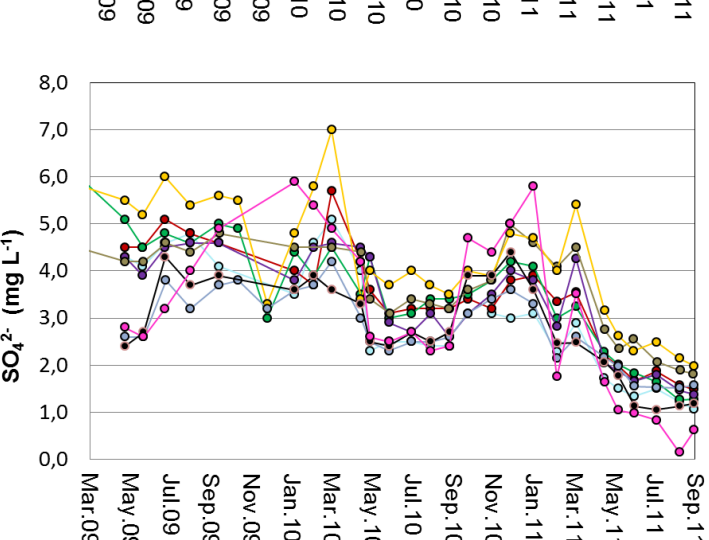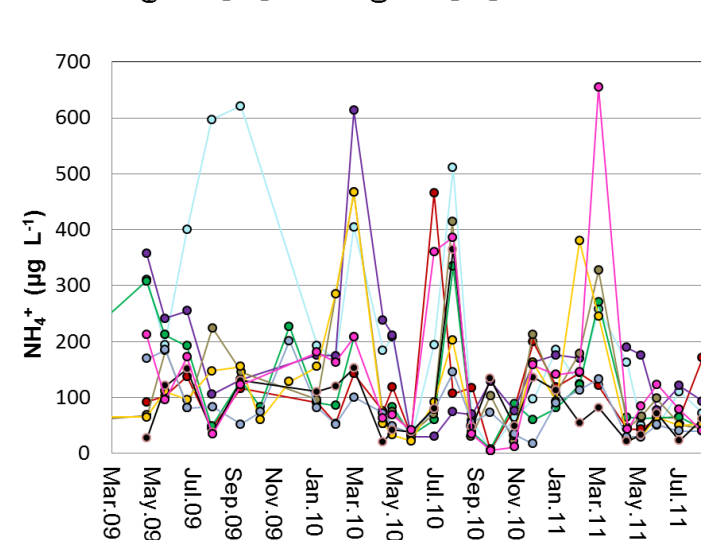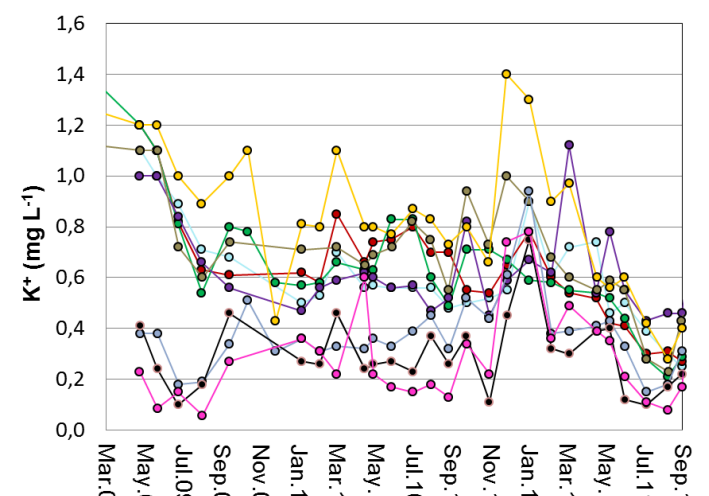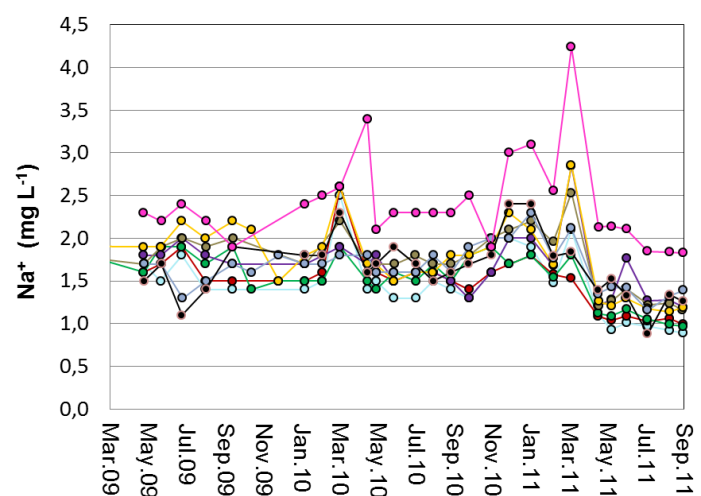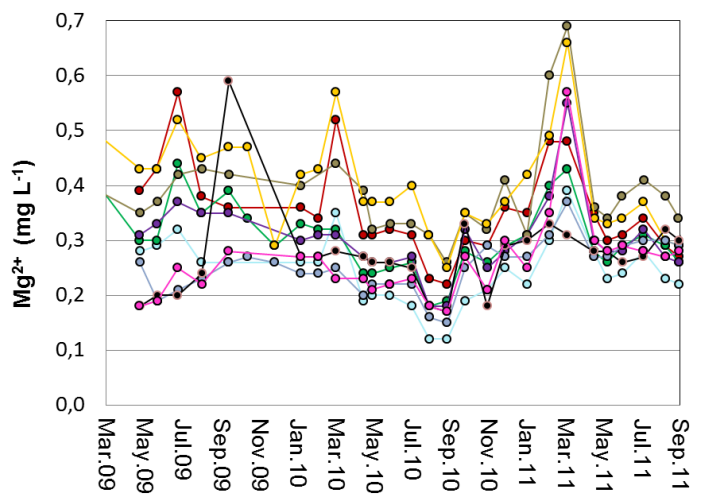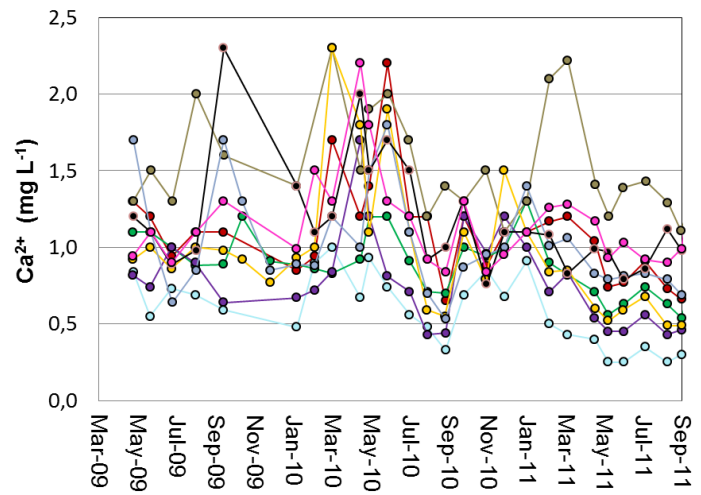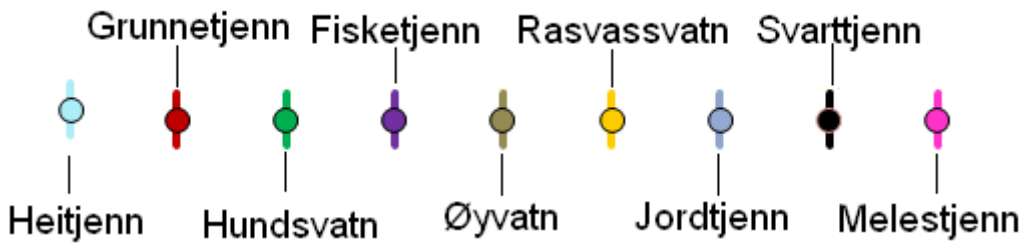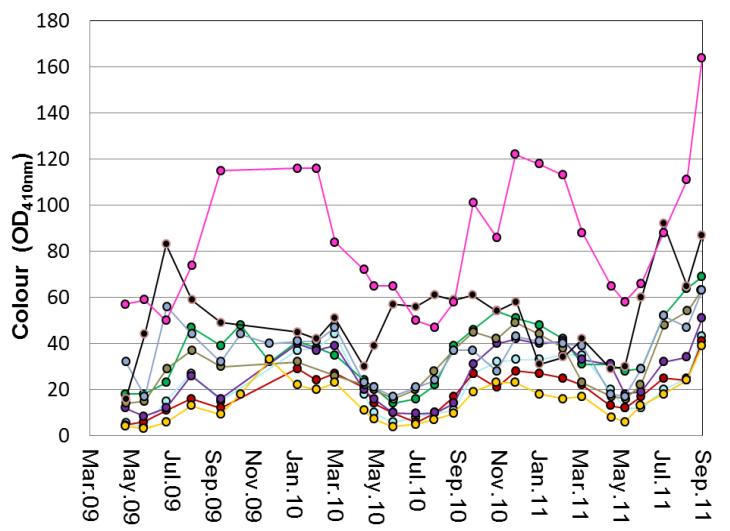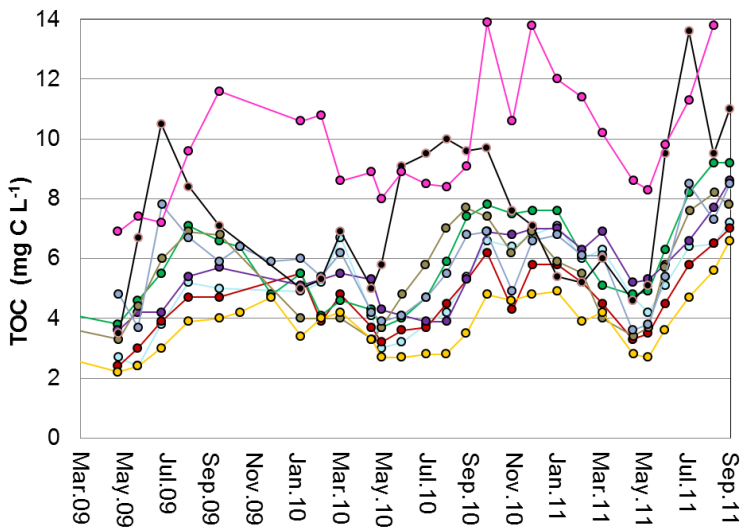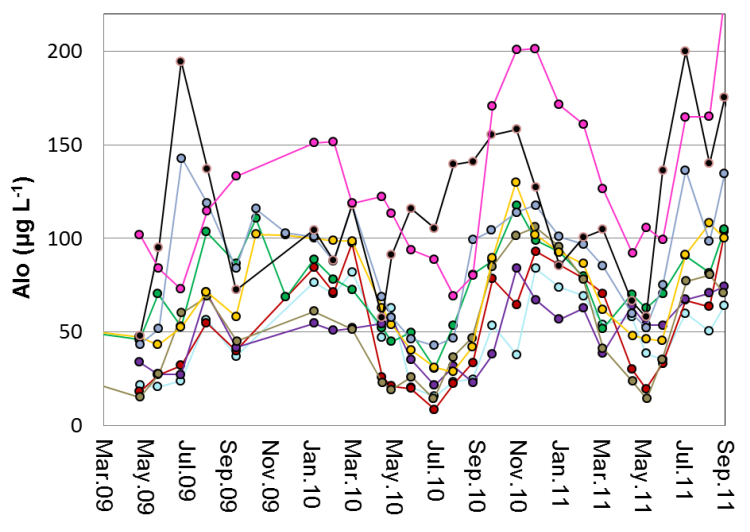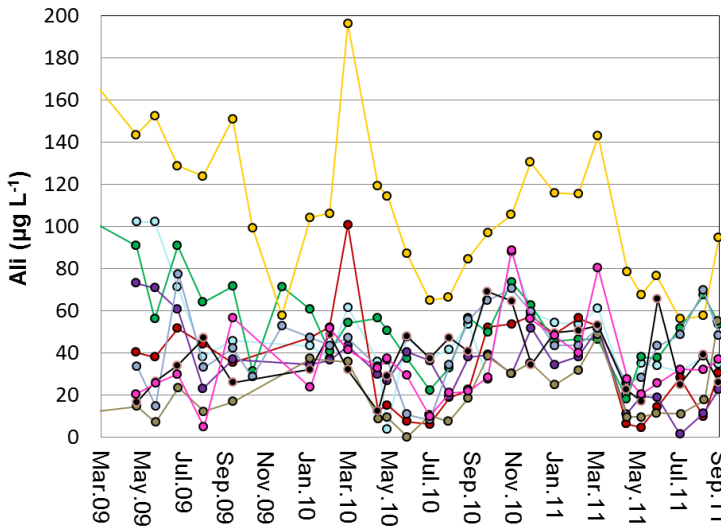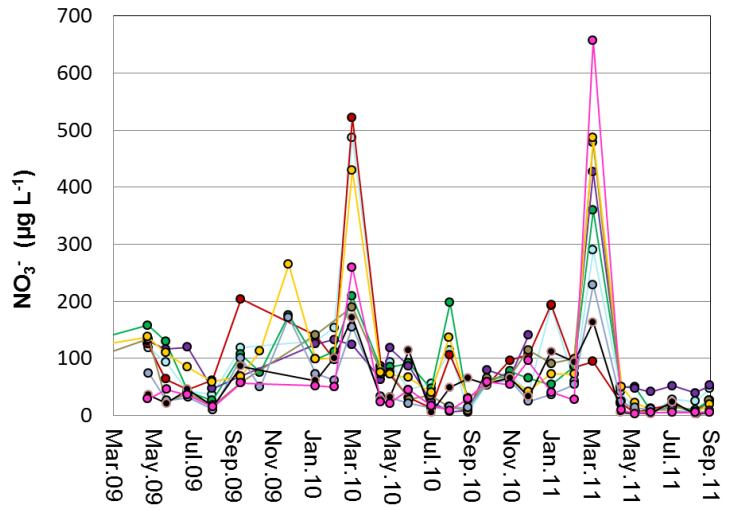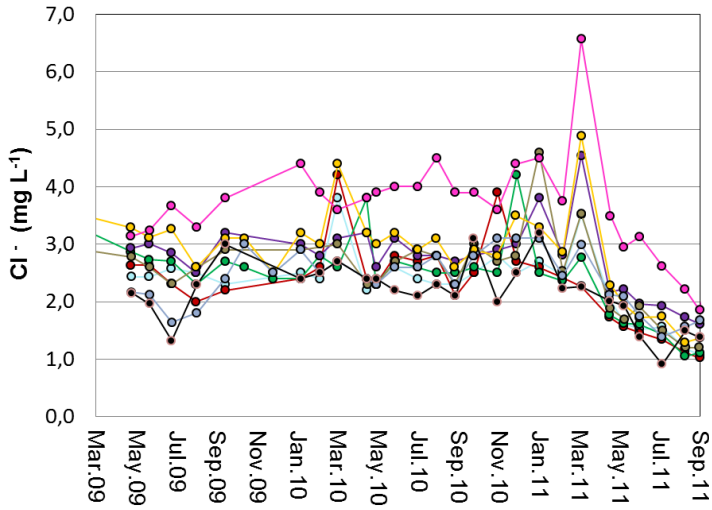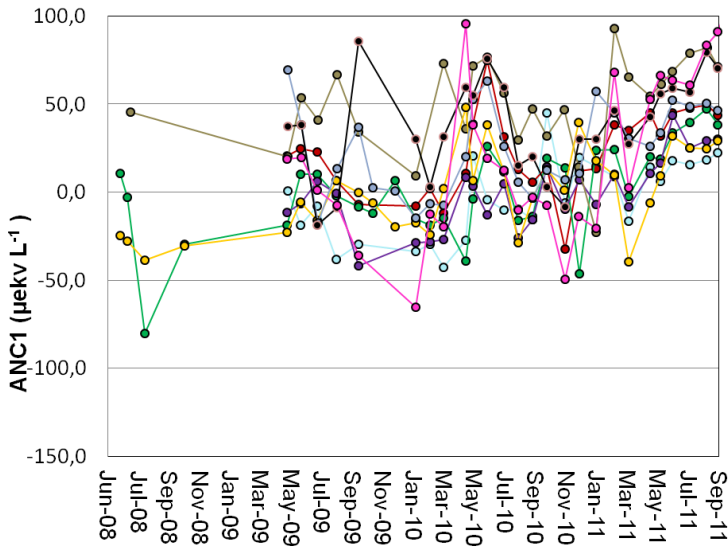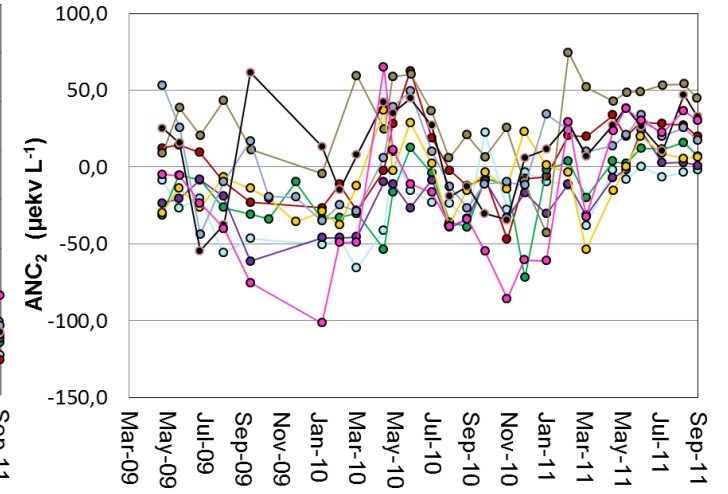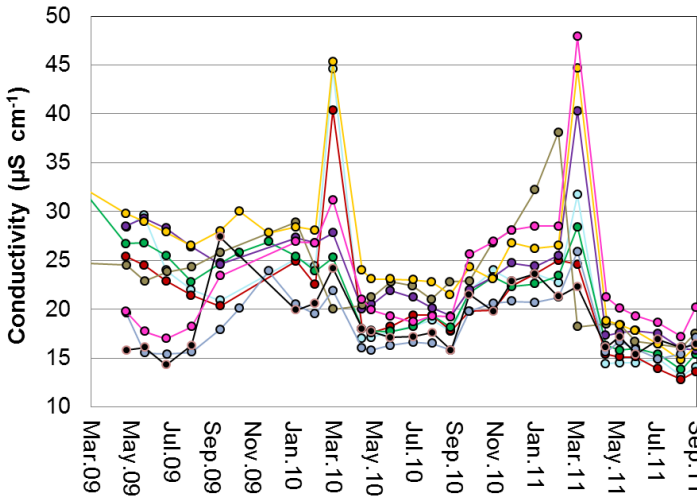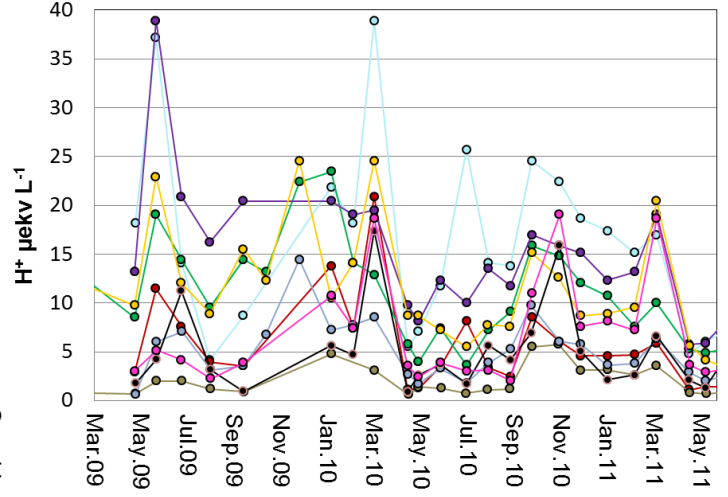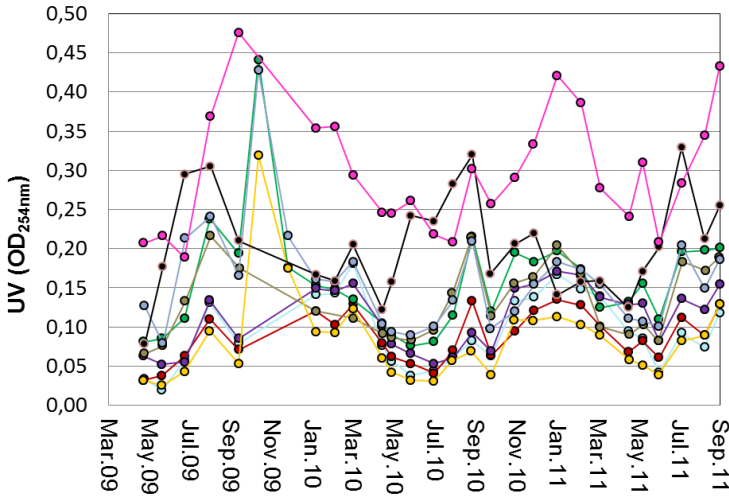
11.3 Appendix 3: Average and median Concentrations
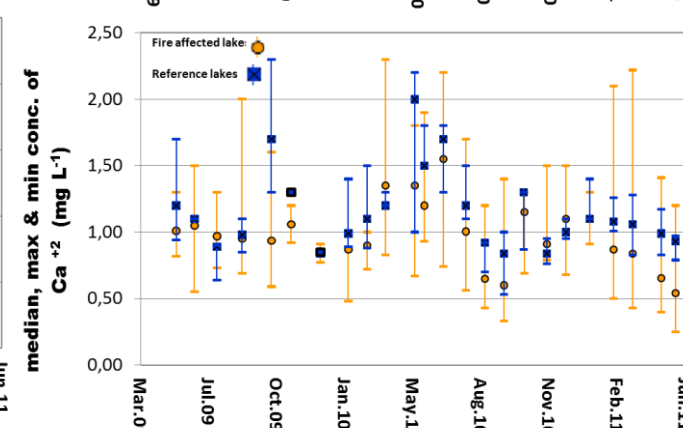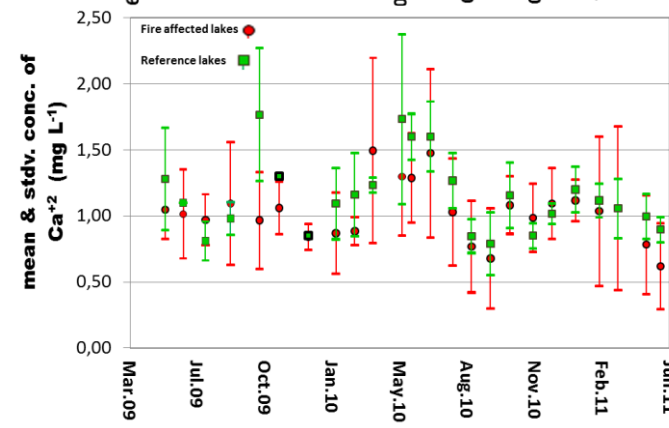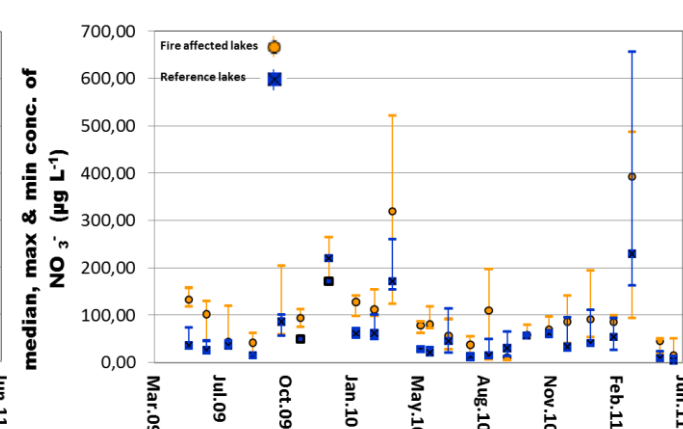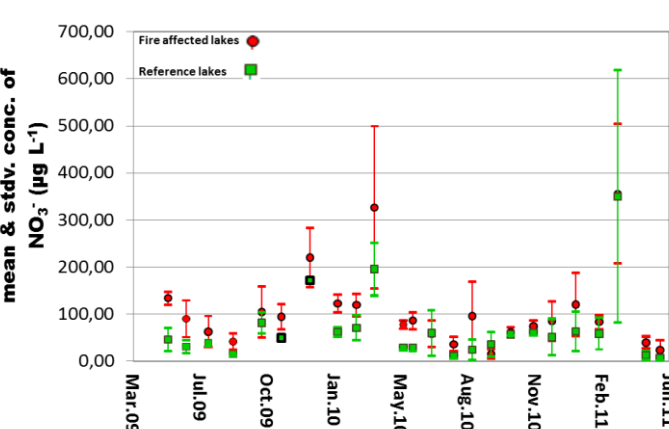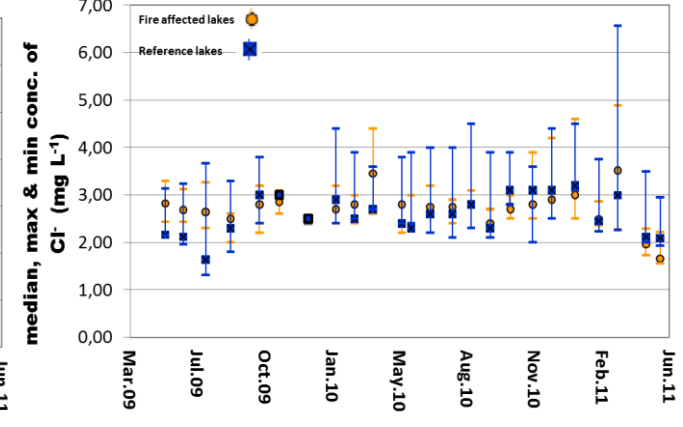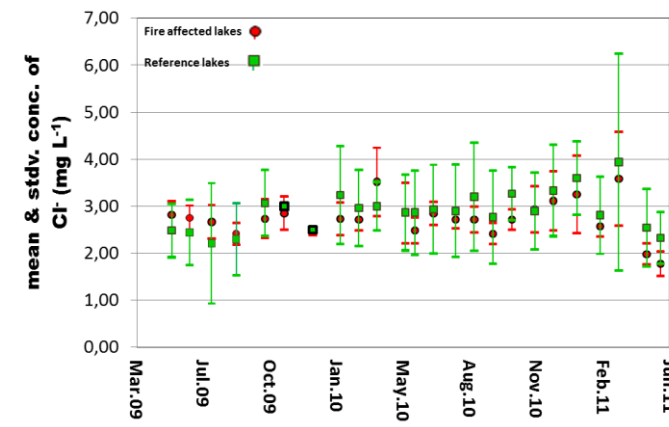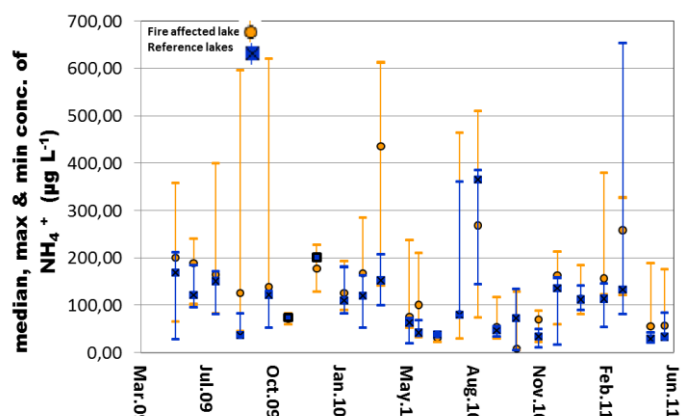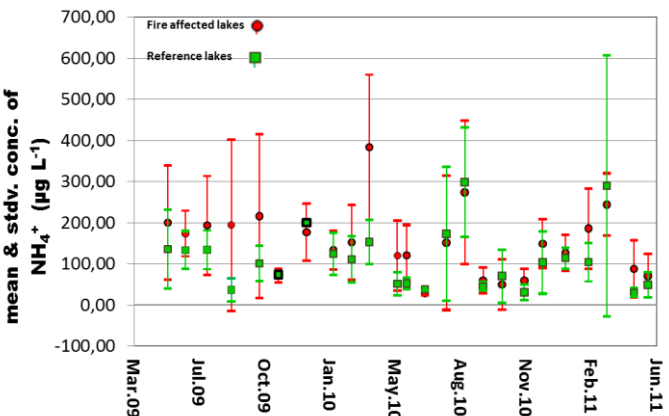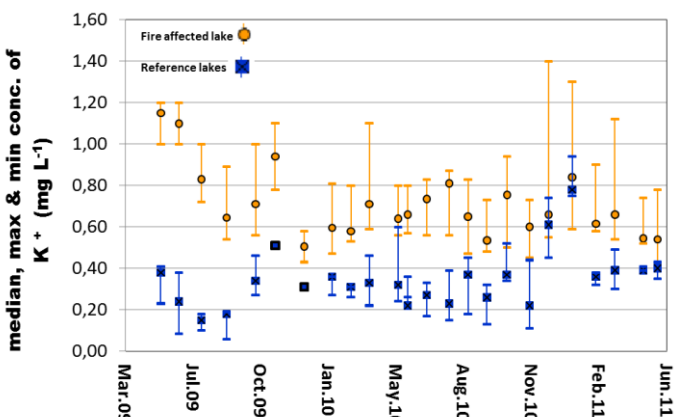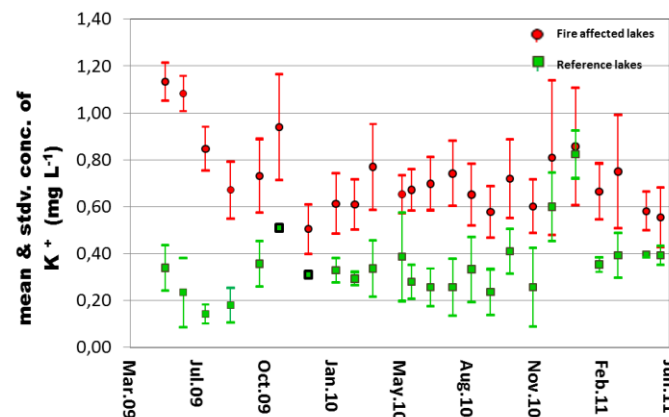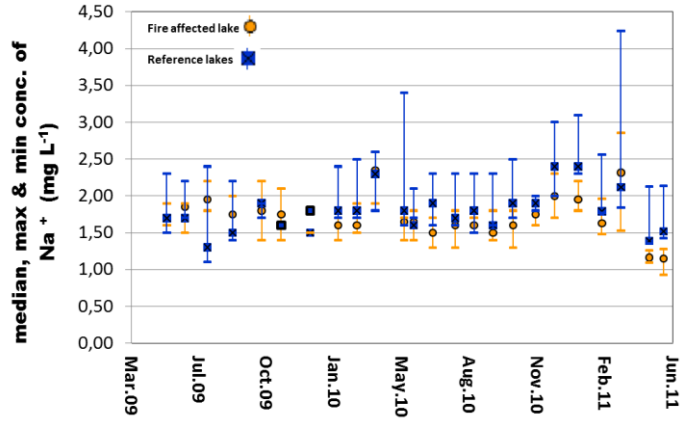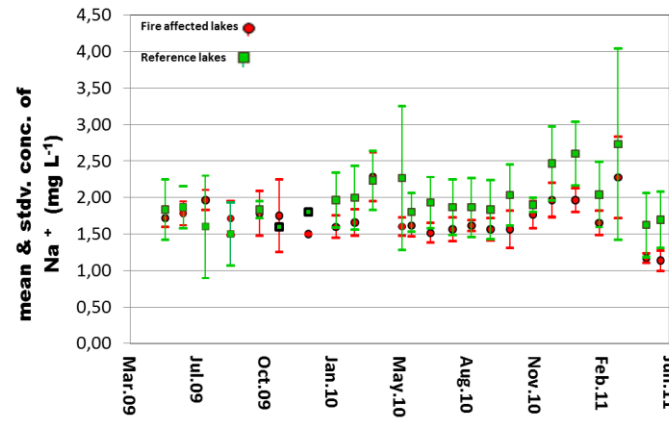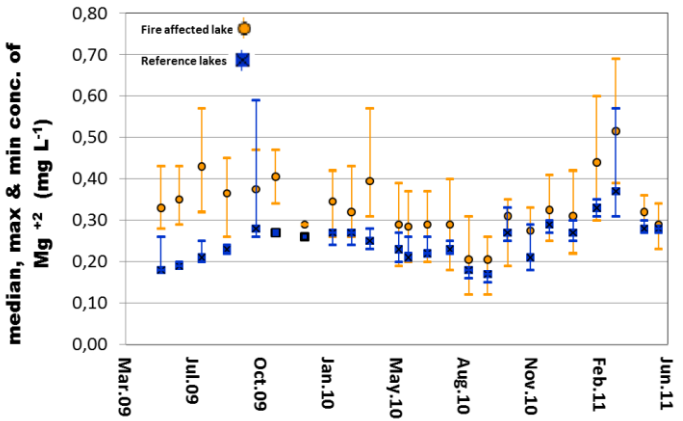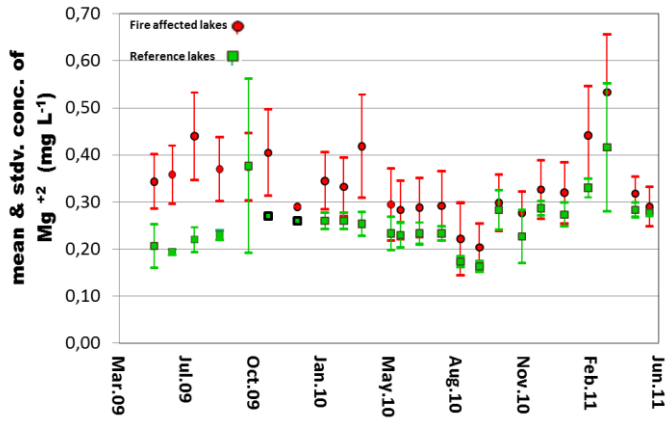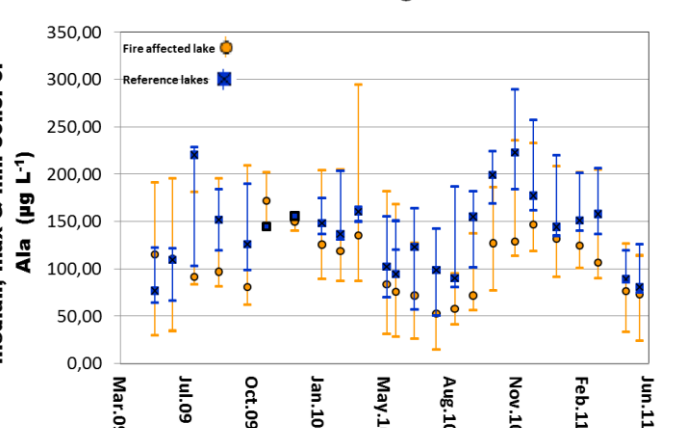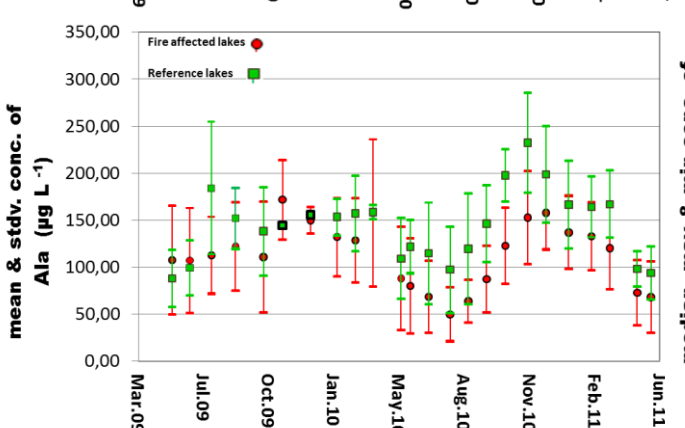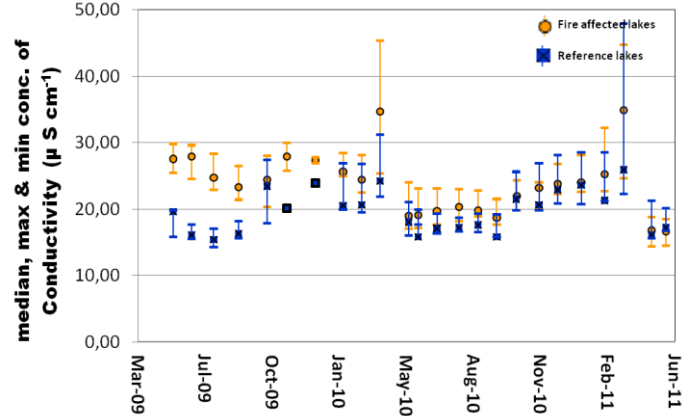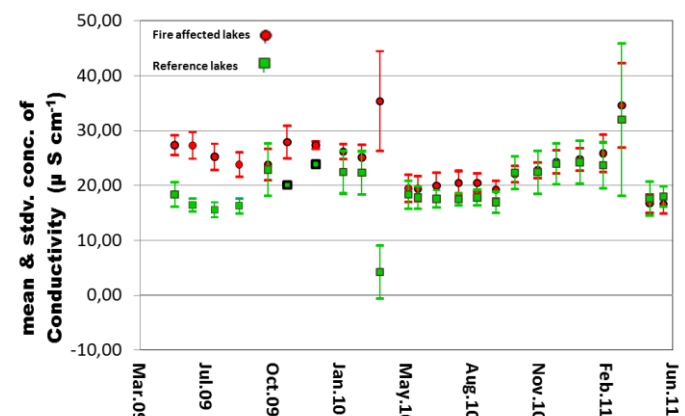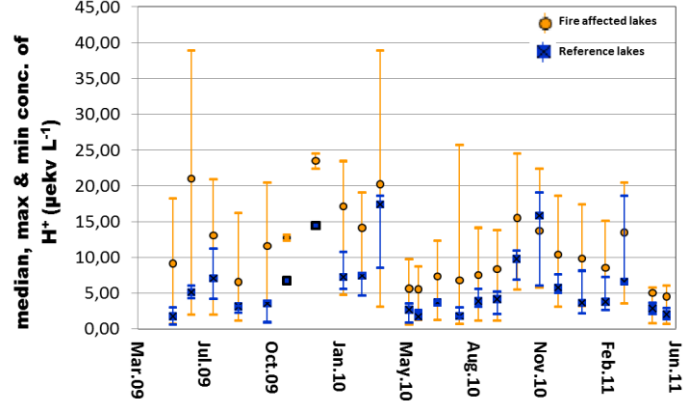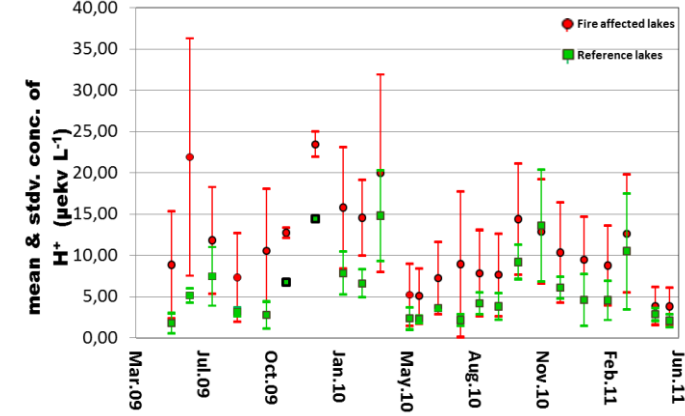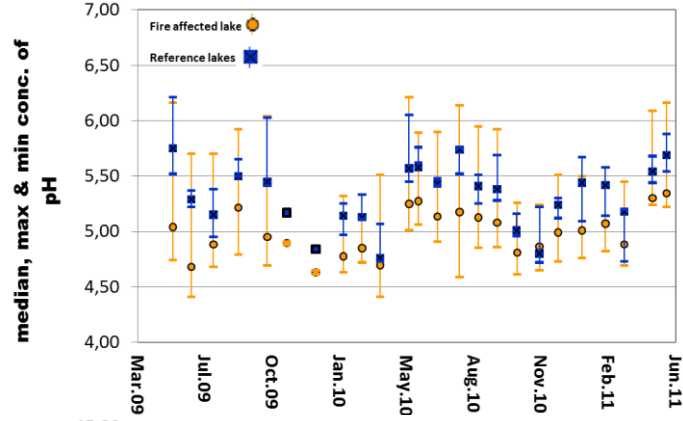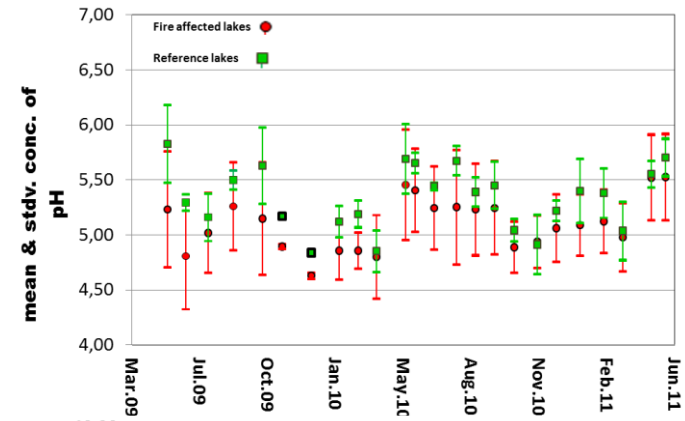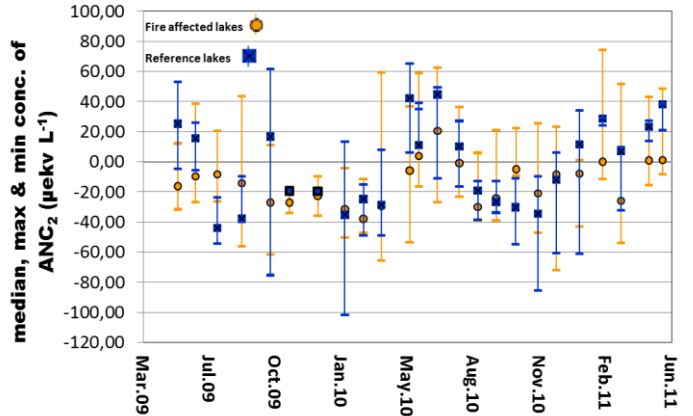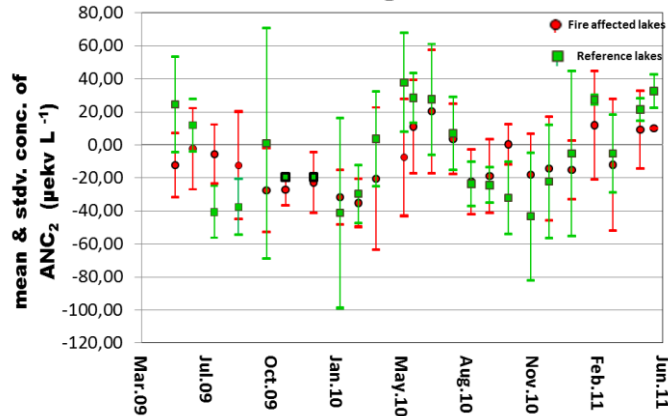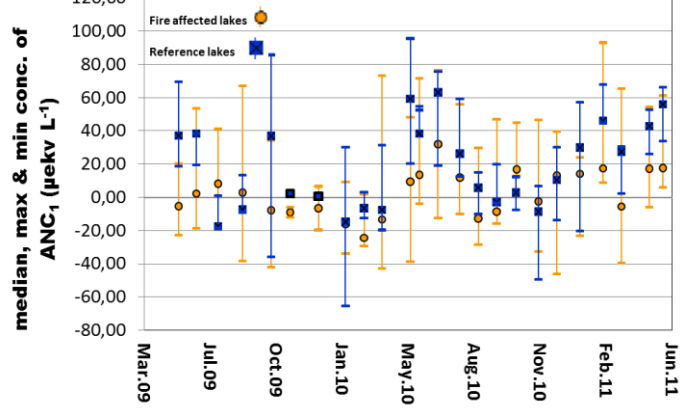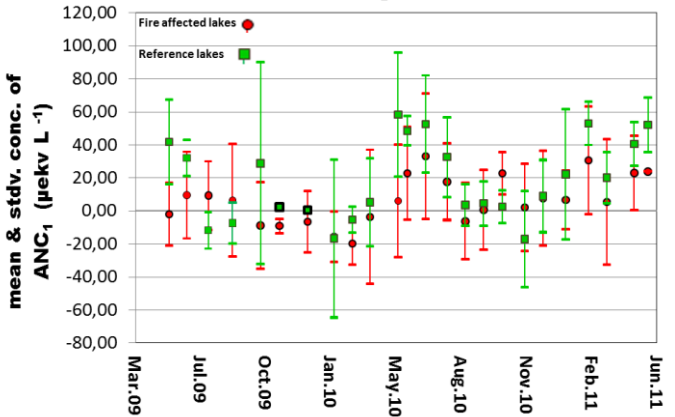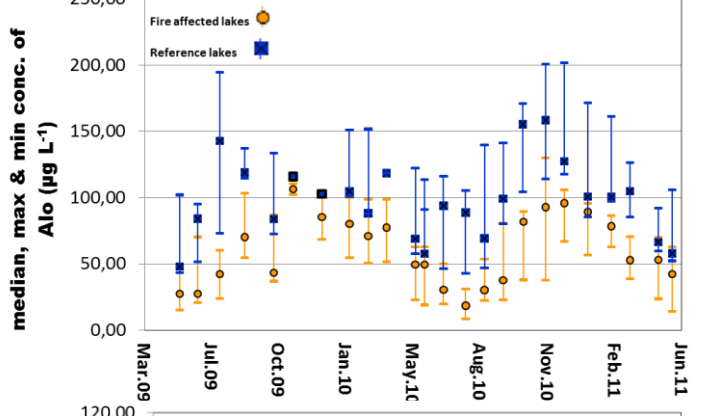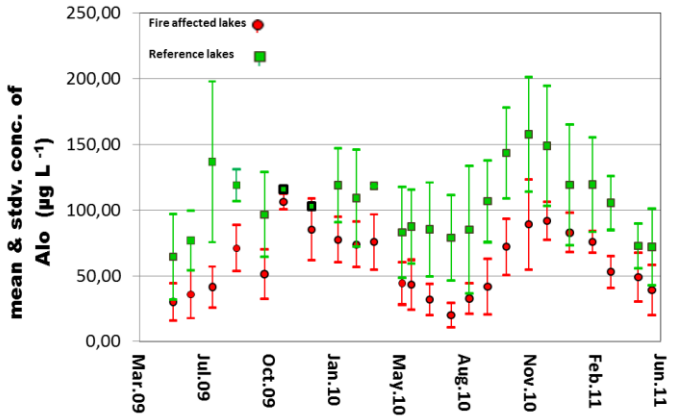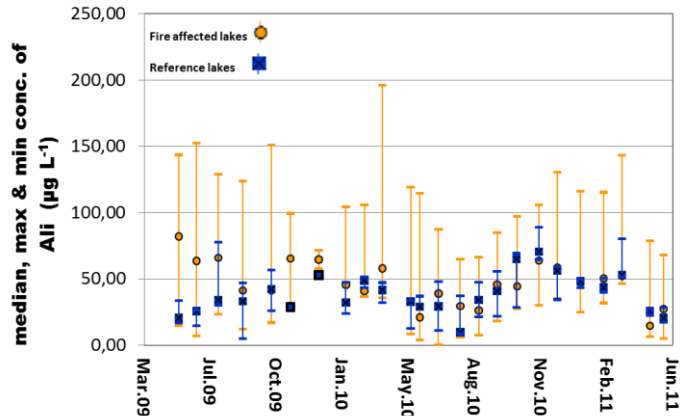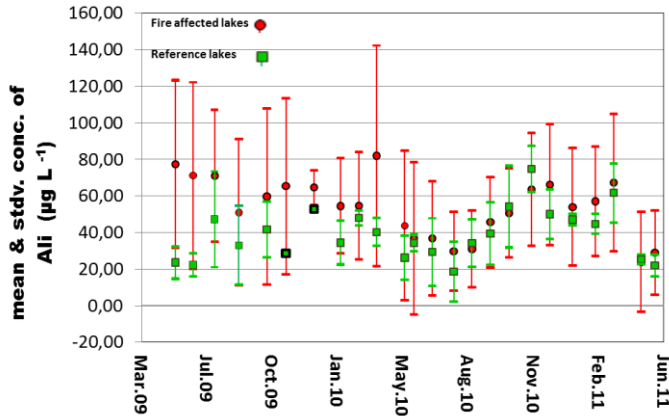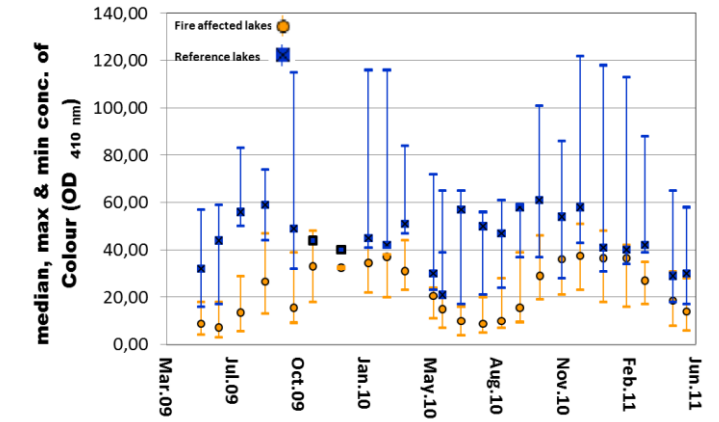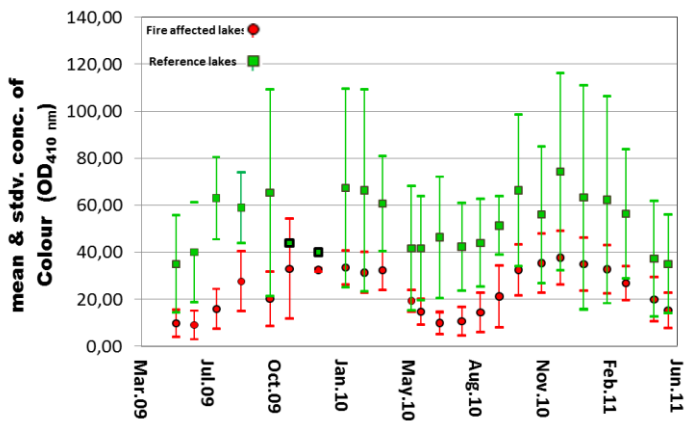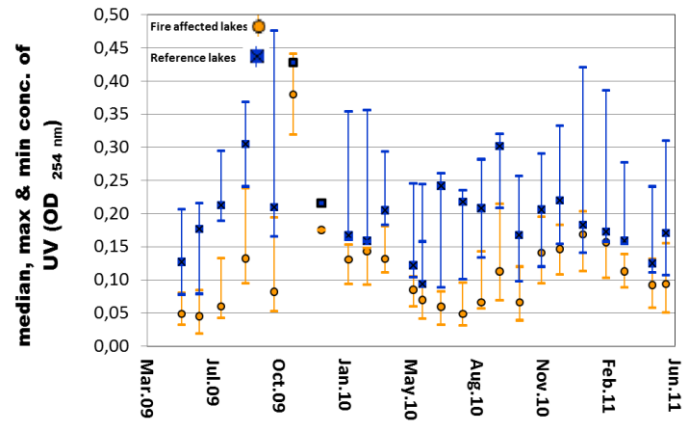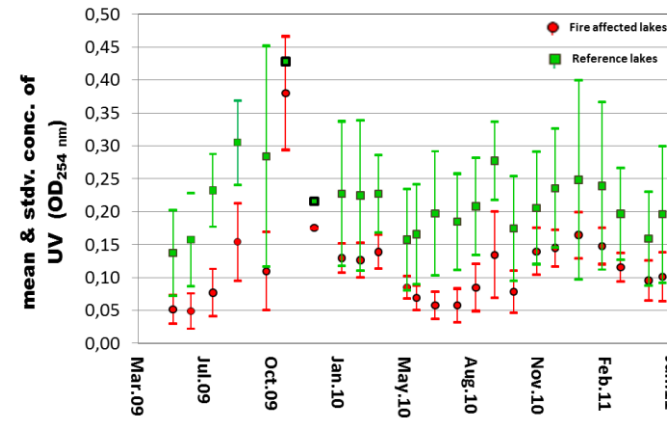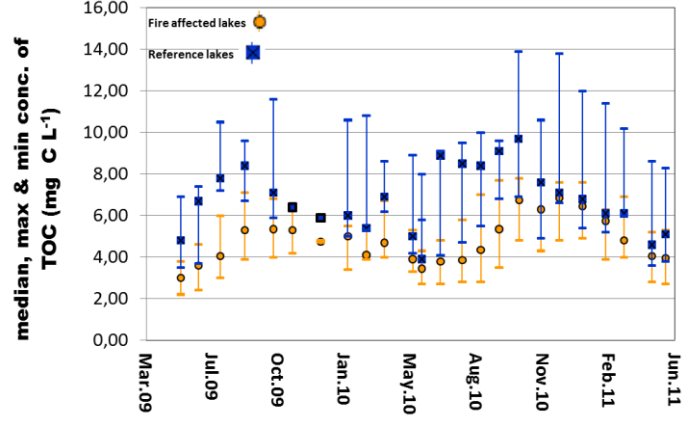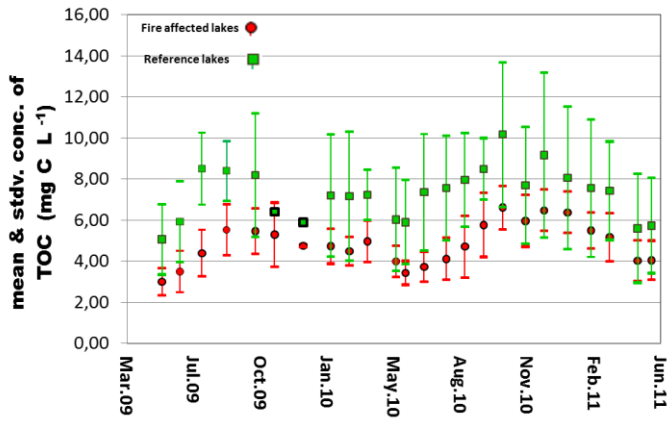
11.4 Appendix 4: Anderson Darling Normality test

Table 20: AD-test, inorganic parameters.

Lake	SO_4^{2-} mg L^{-1}	Cl^- mg L^{-1}	NO_3^- µg L^{-1}	Ca^{2+} mg L^{-1}	Mg^{2+} mg L^{-1}	Na^+ mg L^{-1}	K^+ mg L^{-1}	NH_4^+ µg L^{-1}
Heitjenn	Yes	No	No	Yes	Yes	No	No	No
Grunnetjenn	Yes	No	No	No	Yes	No	No	No
Hundsvatn 1m	Yes	No	No	Yes	Yes	Yes	No	No
Hundsvatn 9m	No	No	Yes	No	No	No	No	No
Fisketjenn	Yes	No	No	Yes	No	No	No	No
Øyvatn	Yes	No	No	No	No	Yes	No	No
Rasvassvatn 1m	Yes	No	No	No	Yes	Yes	Yes	No
Rasvassvatn 14m	Yes	No	No	No	Yes	No	No	No
Jordtjenn 1m Ref..	Yes	Yes	No	No	Yes	Yes	No	No
Jordtjenn 7m Ref.	Yes	No	No	Yes	Yes	No	No	No
Svattjenn Ref.	No	Yes	Yes	Yes	No	Yes	Yes	No
Melestjenn Ref.	Yes	No	No	No	No	No	No	No

Table 21: AD-test for H^+, conductivity, Al_i, ANC and organic parameters.

| Lake	H^+ µekv L^{-1}	Cond. µS/cm	Ali µg L^{-1}	Alo µg L^{-1}	Ala µg L^{-1}	TOC mg C L^{-1}	UV OD_{410nm}	Colour OD_{254nm}	ANC1 µekv L^{-1}	ANC2 µekv L^{-1}
Heitjenn	Yes	No	Yes	Yes	Yes	Yes	Yes	No	Yes	Yes
Grunnetjenn	No	No	No	No	Yes	Yes	Yes	Yes	Yes	Yes
Hundsvatn 1m	Yes	Yes	Yes	Yes	Yes	Yes	No	Yes	Yes	Yes
Hundsvatn 9m	No	No	No	Yes	No	Yes	Yes	Yes	Yes	Yes
Fisketjenn	No	Yes	Yes	No	Yes	Yes	No	No	Yes	Yes
Øyvatn	No	Yes	Yes	Yes	No	Yes	Yes	Yes	Yes	Yes
Rasvassvatn 1m	No	No	Yes	No	No	Yes	No	Yes	Yes	Yes
Rasvassvatn 14m	No	No	Yes	Yes	Yes	Yes	No	No	Yes	Yes
Jordtjenn 1m Ref.	Yes	No	Yes	Yes	Yes	Yes	No	No	Yes	Yes
Jordtjenn 7m Ref.	No	Yes	Yes	Yes	Yes	No	No	Yes	Yes	Yes
Svattjenn Ref.	No	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes
Melestjenn Ref.	No	No	Yes	Yes	Yes	Yes	Yes	No	Yes	Yes

Table 22: AD-test in each month for inorganic parameters.

month	SO_4^{2-} mg L^{-1}	Cl^- mg L^{-1}	NO_3^- µg L^{-1}	Ca^{2+} mg L^{-1}	Mg^{2+} mg L^{-1}	Na^+ mg L^{-1}	K^+ mg L^{-1}	NH_4^+ µg L^{-1}
January	Yes	No	Yes	Yes	Yes	Yes	Yes	No
February	Yes	No	Yes	Yes	Yes	No	Yes	No
March	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes
April	Yes	No	Yes	Yes	Yes	No	Yes	No
May	No	Yes	No	Yes	Yes	Yes	No	No
June	No	Yes	Yes	Yes	Yes	Yes	Yes	No
July	Yes	Yes	No	Yes	Yes	Yes	No	No
August	Yes	No	No	Yes	Yes	Yes	Yes	No
September	Yes	No	Yes	Yes	Yes	No	Yes	Yes
October	Yes	Yes	No	Yes	Yes	Yes	Yes	No
November	Yes	Yes	Yes	No	Yes	Yes	Yes	Yes
December	Yes	No	Yes	Yes	No	Yes	No	Yes

Table 23: AD-test in each month for H$^+$, conductivity, Al$_i$, ANC and organic parameters.

month	H^+ µekv L^{-1}	Cond. µS/cm	Ali µg L^{-1}	Alo µg L^{-1}	Ala µg L^{-1}	TOC mg C L^{-1}	UV OD$_{410nm}$	Colour OD$_{254nm}$	ANC1 µekv L^{-1}	ANC2 µekv L^{-1}
January	Yes	Yes	No	No	Yes	No	No	No	Yes	Yes
February	Yes	Yes	No	No	No	No	No	No	Yes	Yes
March	Yes	No	No	Yes	Yes	Yes	No	No	Yes	Yes
April	Yes	Yes	No	Yes	Yes	No	No	No	Yes	Yes
May	No	Yes	No	No	Yes	No	No	No	Yes	Yes
June	No	Yes	Yes	No	Yes	No	No	No	Yes	Yes
July	Yes	Yes	Yes	No	Yes	No	No	No	Yes	Yes
August	No	Yes	No	Yes	No	Yes	Yes	Yes	Yes	Yes
September	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes
October	Yes	Yes	No	Yes	Yes	No	No	No	Yes	Yes
November	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes
December	Yes	Yes	No	No	No	No	Yes	No	Yes	Yes

Table 24: AD-test for average concentrations in burned lakes (B) and reference lakes (R)

parameter	H^+ µekv L^{-1}	Cond. µS/cm	Ali µg L^{-1}	Alo µg L^{-1}	Ala µg L^{-1}	TOC mg C L^{-1}	UV OD$_{410nm}$	Colour OD$_{254nm}$	ANC1 µekv L^{-1}	ANC2 µekv L^{-1}
Average B-Lakes	Yes	Yes	Yes	Yes	Yes	Yes	No	No	Yes	Yes
Average R-Lakes	No	No	Yes	Yes	Yes	Yes	No	No	Yes	No

parameter	SO_4^{2-} mg L^{-1}	Cl^- mg L^{-1}	NO_3^- µg L^{-1}	Ca^{2+} mg L^{-1}	Mg^{2+} mg L^{-1}	Na^+ mg L^{-1}	K^+ mg L$^-$	NH_4^+ µg L^{-1}
Average B-Lakes	Yes	No	No	Yes	Yes	No	No	Yes
Average R-Lakes	No	Yes	No	Yes	Yes	No	No	No

11.5 Appendix 5: Paired T –test

Table 25: Paired T-test for H⁺, conductivity, Alᵢ, ANC and organic parameters.

Lake	n	H⁺ μekv L⁻¹ mean dif.	Cond. μS/cm mean dif.	Ali μg L⁻¹ mean dif.	Alo μg L⁻¹ mean dif.	Ala μg L⁻¹ mean dif.	TOC mg C L⁻¹ mean dif.	UV OD₄₁₀nm mean dif.	Colour OD₂₅₄nm mean dif.	ANC1 μekv L⁻¹ mean dif.	ANC2 μekv L⁻¹ mean dif.
Heitjenn	9	-3,26	-5,53*	-21,2	-0,48	-21,64*	0,54	-0,005	-2,19	32,15***	30,3***
Grunnetjenn	9	-3,52	-4,14*	-17,55*	-3,91	-21,5	0,38	-0,003	0,33	27,19***	25,91***
Hundsvatn 1m	11	-5,39**	-3,7**	-18,38	-4,77	-23,1	0,68	-0,035	1,18	15,67	13,35
Hundsvatn 9m	9	-1,68	-11,21**	-29,4	6,33	-23	0,71***	0,03***	9,11**	35,59***	33,17***
Fisketjenn	9	-7,76*	-2,87	-12,34	-0,76	-13,10	0,47	-0,001	1,23	13,33	11,76
Øyvatn	8	0,28	-0,2	-0,85	-1,4	-2,2	0,41	-0,01	1	3,6	2,2
Rasvassvatn 1m	11	-4,88*	-3,86***	-24	-6,61	-30,6	0,15	-0,03	-1,38	10,8	10,3
Rasvassvatn 14m	9	0,39	-5,13*	-10.9	-27	-38	0,51	-0,04	-15,93	35,1*	33,3*
Jordtjenn 1m Ref..	11	-1,72	0,36	0,43	-14,8	-15,3	-0,35	-0,06	-8*	18,29	19,46
Jordtjenn 7m Ref.	8	-0,25	-0,59	-1,12	9,26	8,18	0,33	-0,01	0	19,1	17,99
Svartjenn Ref.	9	-1,76	0,2	14,56*	5,4	20	0,77	-0,01	-1	10,7	8,1
Melestjenn Ref.	9	0,25	2,99	3,83	4,01	7,84	0,92*	-0,02	-3,44	17,9	14,7

Tabell 26: Paired T-test for inorganic parameters.

Lake	n	SO₄²⁻ mg L⁻¹ mean dif.	n	Cl⁻ mg L⁻¹ mean dif.	n	NO₃⁻ μg L⁻¹ mean dif.	n	Ca²⁺ mg L⁻¹ mean dif.	n	Mg²⁺ mg L⁻¹ mean dif.	n	Na⁺ mg L⁻¹ mean dif.	n	K⁺ mg L⁻¹ mean dif.	n	NH₄⁺ μg L⁻¹ mean dif.
Heitjenn	8	-1,86***	9	-0,03	9	-35,5	9	-0,09	9	-0,04	9	-0,12	9	-0,11	9	-155,7*
Grunnetjenn	9	-1,29***	9	-0,14	9	-75,2	9	0,15	9	-0,06	9	-0,2	9	-0,12	9	33,6
Hundsvatn 1m	11	-0,94***	11	-0,1	11	-4,2	11	0,01	11	-0,04	11	-0,04	11	-0,09	10	-44,2
Hundsvatn 9m	9	-2,96***	9	-1,19**	9	-12	9	-0,28*	9	-0,15	9	-0,54***	9	-0,4*	9	-94,5
Fisketjenn	9	-1,14***	9	0,09	9	11	9	-0,07	9	-0,01	9	-0,07	9	-0,04	8	-83,7*
Øyvatn	8	-0,87***	8	0,29	8	16,8	8	0,02	8	-0,38	8	0,50	8	-0,05	8	-17,4
Rasvassvatn 1m	11	-1,1**	11	-0,003	11	-34,9	11	-0,09	11	-0,04	11	-0,14	11	-0,04	10	-27,9
Rasvassvatn 14m	9	-2,39***	9	-0,58*	9	-31,1	9	-0,22**	9	-0,09	9	-0,27	9	-0,14	8	-10,1
Jordtjenn 1m Ref..	11	-0,66*	11	0,35*	11	-20	11	0,01	11	0,02	11	0,16	11	0,15*	11	-32,3
Jordtjenn 7m Ref.	8	-1,25**	8	0,06	8	-21,9	8	-0,03	8	0,01	8	-0,15	8	0,01	8	-100,3**
Svattjenn Ref.	9	-0,76*	9	0,12	9	6,9	9	-0,14	9	0,00	9	0,09	9	0,07	9	6,6
Melestjenn Ref.	9	-1,16*	9	0,64	9	35,3	9	0,00	9	0,05	9	0,18	9	0,09	9	66