Effects of wildfire on water chemistry and perch (*Perca fluviatilis*) populations in some acidic lakes in southern Norway

Av Espen Lydersen, Rolf Høgberget, Marijanne Holtan, Mari Darrud and Clara Enedina Moreno

Espen Lydersen is professor in limnology at the Department of Environmental and Health Studies (INHM), University College of Southeast Norway (HSN), Campus Bø. *Rolf Høgberget* is scientist at the Norwegian Institute for Water Research, Grimstad. *Marijanne Holtan, Mari Darrud* are MSc-candidates, and *Clara Enedina Moreno*, PhD-candidate from IN HM, HSN.

Sammendrag

Effekter av skogbrann på vannkjemi og abbor (*Perca fluviatilis*) populasjoner ble undersøkt i 3 innsjøer i Aust-Agder, etter Frolandsbrannen i 2008. Under en betydelig nedbørsepisode, to måneder etter brannen, inntraff den mest alvorlige vannkjemiske episoden. I den mest påvirkede innsjøen, ble det da registrert en pH-verdi på 4,42, samtidig med at konsentrasjonen av giftige aluminiumsforbindelser (Al_i) var meget høy, 326 µg Al L^{-1}. Denne ekstreme vannkjemiske episoden, kombinert med betydelige forsuringseffekter første året etter brannen, hadde stor negativ innvirkning på abborbestanden i denne innsjøen. Ett år etter brannen var vannkjemien tilnærmet tilbake til forholdene før brannen. Dette medførte en betydelig reproduksjonssuksess for den gjenværende, gytemodne abboren våren 2009. Foryngingen (juveniliseringen) av abborbestanden etter brannen er trolig et forbigående fenomen, som etter noen år trolig ikke lenger vil kunne dokumenteres. I innsjøen som hadde den høyest pH og syrenøytraliseringskapasitet (ANC), var forsuringseffektene av brannen langt mindre, og ingen negative effekter på abborbestanden kunne påvises.

Summary

Effects of wildfire on perch (*Perca fluviatilis*) populations were studied in three, acidic boreal lakes in southern Norway. In two lakes, significant postfire acidification occurred due to pulses of sulfate mobilized in the soil. In one lake pH dropped to 4.42, and the concentrations of inorganic aluminum (Al_i) increased to 326 µg Al L^{-1} during a storm event, about two months after the fire. This extreme water chemical episode, combined with adverse acidification during the remaining first postfire year, almost eradicated the perch population. During the second postfire year, the water chemical conditions were almost back to prefire conditions and strong populations of young individuals were established. The juvenilization of the population after the first postfire year is most likely a temporary phenomenon. In the lake with the highest pH, the postfire acidification effect was too low to cause any effect on the perch population.

Introduction

Each year wildfires burn large areas of forest land around the world, often with significant impacts to life, natural resources, property and infrastructure (Smith et al., 2011). Future climate change scenarios for the interior of many continents suggest an increase of air temperature by 2 – 6°C by 2100 coupled with a 10 – 30% decrease of summer precipitation (IPCC, 2011; IPCC, 2014; Schär et al., 2004). Consequently, more severe and long lasting drought periods will occur. This climate change is likely to increase the number of days with severe burning condi tions, prolong the fire season, and increase light ning activity, all of which lead to probable increases in fire frequency and areas burned (Price and Rind, 1994; Gol-dammer and Price, 1998; Stocks et al., 1998; USDA, 2000; Flannigan and Wotton, 2001. Pechony and Schindell, 2010). Miranda (1994) suggests an increase in risk, severity, and frequ-ency of forest fires in Europe. The large question is where and how much various areas on earth will be affected by these potential changes.

Prolonged drought followed by a strong rain event, may lead to adverse changes in water chemis try. Already in 1922, Huitfeldt-Kaas (1922) documented mass death of salmon (*Salmo salar*) and brown trout (*Salmo trutta*) in Norway during such an event, and later many scientists have dealt with this phenomenon (Jeffries et al., 2003; Laudon et al., 2004; Aherne et al., 2006). Drought periods result in lower water table in wetlands and soils, leading to oxidation of previo usly stored, reduced sulfur (S), and sub-sequent efflux of oxidized S (SO_4^{2-}) upon re-wet-ting. In acid sensitive areas with high sulfur input and low acid neutralizing capacity, acute toxic levels of H^+ and cationic Al (Al^{n+}) may occur (Dickson, 1978; Cronan and Schofield, 1979). A wildfire will intensify the drought conditions as it create both aboveground and belowground heat pulses, but the impact of wildfire on soils depends on many factors as air temperature, humidity, wind condition, soil moisture burn conditions, fuel loading and fuel moisture (USDA, 2005).

Differences in watershed response between wildfires or within an individual wildfire, depend largely upon fire severity (intensity and fraction of catchment burnt), topography, lake morphometric conditions (residence time and catchment-to-lake area), acid neutralizing capacity of soil/soil water and the size of the hydrologic event or storm, and the timing relative to fire (Carignan et al., 2000; Smith et al., 2011; Lydersen et al., 2014).

The effects of wildfire on fish are mostly indirect in nature (USDA, 2005), even though some studies have documented direct mortality following wildfire (McMahon and Calesta, 1990; Minshall and Brook, 1991; Reiman et al., 1997). High severity fire and heavy fuel and slash accumulations in the riparian zone are the most common predisposing factors explaining direct fish mortality. The largest problems arise from the longer-term impact on habitat, due to changes in stream temperature, plant understory and overstory removal, ash-laden slurry flows, increases in flood peak flows, and sedimentation due to increased landscape erosion. While Rinne and Carter (2008) reported indirect, acute effects on fish, especially during storm events weeks to months after the fire was extinguished, Bozek and Young (1994) reported fish kills in a stream during a severe storm flow two years after the fire. While they reported severe fish kill due to asphyxiation/hypoxia caused by high concentration of suspended sediments during a post-fire storm, as sediments completely embedded the fish gills, Rinne and Carter (2008) concluded that drought conditions and stream intermittency combined with ash flows synergistically caused fish mortality. Other potential physic-chemical explanatory factors for the fish mortality were not incorporated or discussed in these studies. Most studies on the impacts of wildfire on fish populations deal with salmonids inhabiting running waters. Thus, as many scientific articles have documented significant postfire chemical and biological effects in lotic systems, the work by St-Onge and Magnan (2000) is the only wildfire study so far reported from lentic environments. By including lakes with different water chemistry, we hypothesized that different water chemical responses of wildfire will occur, with subsequent variations in effects on lentic biota. Thus, our population study on European perch (*Perca fluviatilis*) in three wildfire affected lakes in Southern Norway is an important contribution to the scientific literature within this field.

Material and methods

Site description

The wildfire took place in the Aust-Agder county in June 2008 and lasted about a week. Almost 2600 ha of forest and wilderness was severely damaged. The wildfire area is characterized by slowly weatherable Precambrian rocks (granites and gneisses), with thin and patchy soil cover developed from glacial moraine. The vegetation is dominated by open forest of Norwegian pine (*Pinus sylvestris*) and birch (*Betula pubescens*), with stands of aspen (*Populus tremula*) on south facing hillsides, and Norwegian spruce (*Picea abies*) in more productive areas. The lakes are relatively small, table 1, acidic, boreal lakes, located between 173 and 255 m a.s.l. Average annual precipitation is about 1200 mm yr^{-1}.

While 100% of the catchments of Lake Hundsvatn and Lake Øyvatn were located within the wildfire area, ≈ 90% of the Lake Rasvassvatn catchment was located within the wildfire area.

Lake Øyvatn and Lake Hundsvatn have significantly higher catchment to lake area ratios (CA:LA = 15 and 18, respectively) compared with Lake Rasvassvatn, i.e. CA:LA = 1.4 (Table 1). Accordingly, the theoretical water residence times (τ) of Lake Øyvatn and Lake Hundsvatn are significantly lower (τ =0.06 and 0.21 years, respectively) compared with Lake Rasvassvatn (τ = 3.18 years). While Lake Øyvatn is too shallow (3.2 m) to be thermally stratified during the summer, both Lake Rasvassvatn and Lake Hundsvatn are deep enough to be thermally stratified, and thus dimictic lakes.

Lake Øyvatn is a headwater lake, while one and three small lakes are located upstream from Lake Rasvassvatn and Lake Hundsvatn, respectively. Based on gillnet fishing in all four upstream lakes, and reports from local landowners, we concluded that these upstream lakes did not host any fish.

Due to stream topographical reasons, migration of perch from downstream lakes is impossible to Lake Øyvatn, and nearly impossible to Lake Hundsvatn, while no such movement barriers exist between Lake Rasvassvatn and the nearest downstream lake. This downstream lake has also previously been limed, and several times, we have observed perch in the slow flowing stream between the two lakes.

Sampling and analysis

In 2008, water samples for chemical analysis, were taken in Lake Hundsvatn and Lake Rasvassvatn on June 25 (nine days after the fire), July 9,

Lake	Unit	Hundsvatn	Øyvatnet	Rasvassvatn
Latitude		N58 36.413	N58 36.675	N58 37.446
Longitude		E8 16.770	E8 19.037	E8 20.165
Altitute (min)	m a.s.l.	228	255	173
Altitude (max)	m a.s.l.	320	316	360
Annual precipitation	mm	1184	1183	1169
Lake area (LA)	km^2	0.15	0.072	0.89
Maximium depth	m	13	3.2	15
Lake volume	m^3	647833	76587	4573287
Catchment area (CA)	km^2	2.63	1.09	1.23
Residence time	yr	0.21	0.06	3.18
Catchment to lake area ratio (CA:LA)		18	15	1.4

Table 1. Location, lake and catchment characteristics of the studied lakes.

August 13 (during a strong rain event), and after lake turnover on October 30 (Høgberget, 2010), while Lake Øyvatn was only sampled once, in July 2008. From May 2009 to October 2012, monthly sampling was implemented in all lakes. At Lake Hundsvatn and Lake Rasvassvatn (dimictic lakes), both epilimnion (at 1 m) and hypolimnion (at 9 m in Lake Hundsvatn and at 14 m in Lake Rasvassvatn) were sampled. At Lake Øyvatn (shallow, monomictic lake), only samples from surface water (at 1 m) were taken.

Water samples were collected with a Limnos sampler (2.7 L, Limnos AB), transferred to pre-washed polyethylene bottles, and stored cold in dark until analyzed. Ion exchange fractionation of aluminum was conducted in the field according to Driscoll (1984). The water analyses were implemented according to standard procedures described in Lydersen et al. (2014). Conductivity, pH, alkalinity and Al-fractions were determined within one day after sampled, while the remaining analysis were all finished within the first week after sampling.

The perch were caught by Nordic gillnets, in July 2008, 3 weeks after the fire, and by Jensen gill nets late August 2012, more than 4 years after the fire. The Jensen series was used in 2012, due to lack of Nordic gillnets. A Nordic gillnet (Appelberg et al., 1995) is a 30 m long and 1.5 m high (45 m^2) net, consisting of 12, 2.5 m long panels, with various mesh sizes from 5 to 55 mm (NS-EN 14757: 2005). A Jensen series (Jensen, 1977) consist of eight, 25 m long and 1.5 m high mono mesh gillnets (37.5 m^2), varying in mesh from 21 mm to 45 mm.

As the lake area of Lake Hundsvatn and Lake Øyvatn are small (< 50 ha), only five Nordic gillnets were put out in these lakes in 2008, to avoid potential overexploitation of the populations, as recommended in the NS-EN 14757 standard. As local people claimed that Lake Rasvassvatn did not host any fish, ten Nordic gillnets were set in this lake to verify this statement.

The location of gillnet sites were randomly selected to minimize bias in locating sites and setting gear. The gill nets were set perpendicular to the shore, primarily within the depth interval 0-3 m. At minimum the fishing lasted for six hours (2012, day-fishing), maximum twelve hours in 2008 (night- fishing), in the various lakes. Catch per unit of effort (CPUE) was calculated as number of individuals (ind.) 100 m^{-2} (gill net area) day^{-1} (day = 12 hours).

The total length, from the nose to the end of the tail (± 1 mm) and weight (± 0.1 g) of all perch were registered and appropriate bone structures (otolith and operculum) sampled for age determination. Age determination was primarily conducted on otoliths. The calcified otolith structures were heated up with a butane lighter until it became light brown and, thereafter, transversally sectioned through the nucleus into two pieces prior to microscope investigation (Lea, 1910). Two independent readings were made by the same reader. When the results differed, another two readings were made, and in some cases, pictures were taken and sent to another expert for a second opinion evaluation. As age determination rely on counting numbers of winter zones in bone structures (otolith and/or operculum), the age notated 1+, 2+ etc. means 1 and 2 winter old individuals.

In 2008, 85 perch of a total catch of 151 were randomly selected for age determination in Lake Hundsvatn, while 37 out of 57 individuals were randomly picked for age determination in Lake Øyvatn. In 2012, all captured individuals were age determined, i.e. 78 in Lake Hundsvatn, 45 in Lake Øyvatn and 38 in Lake Rasvassvatn.

Besides perch, Lake Øyvatn and Lake Hundsvatn also host a "certain brow trout population", but our catches were too low for evaluation of wildfire effects on this species.

Calculations

Aluminum (Al) was fractionated according to the Barnes/Driscoll method (Barnes, 1975; Driscoll, 1984). Two Al-fractions were measured: total monomeric Al (Al_a) and organic monomeric Al (Al_o). Based on these fractions, inorganic monomeric Al (Al_i) was calculated as Al_a - Al_o.

Acid neutralizing capacity (ANC) was calculated on equivalent basis (eq L^{-1}) according to Reuss and Johnson (1986), i.e.:

[ANC]: $\Sigma[Ca^{2+}],[Mg^{2+}],[Na^{+}],[K^{+}] - \Sigma[SO_4^{2-}],[NO_3^{-}],[Cl^{-}])$

The concentrations of non-marine SO_4^{2-} (SO_4*) and non-marine base cations (Ca*, Mg*, Na* and K*) were calculated by subtracting a marine fraction estimated from the ion equivalent ratio to Cl^- in seawater (Weast, 1988):

$$SO_4^* = [SO_4^{2-}] - 0.103[Cl^-];\ Ca^* = [Ca^{2+}] - 0.037[Cl^-];\ Mg^* = [Mg^{2+}] - 0.195[Cl^-];\ Na^* = [Na^+] - 0.859\,[Cl^-];\ K^* = [K^+] - 0.0181[Cl^-]$$

As the macro-chemical effects in acid sensitive surface waters often deals with the relationship between strong acids (basically SO_4*) and catchment derived base cations (ΣBC* = Ca*+Mg*+Na*+K*), much of the macro-chemical interpretations in this paper are based on the relationship between these two non-marine variables.

To compare the condition factors of perch between lakes and years, we have used the Fulton condition factor (Fulton, 1904), $K_f = \frac{W*100}{(L)^3}$ where W is the weight in gram and L is the length in cm.

The Minitab 16 Statistical software program were used for statistical analyses (ANOVA, and two sample T-test), with significant level of $p \leq 0.05$.

Results

Water chemistry

The most adverse water chemical conditions were revealed in epilimnion of the thermally stratified Lake Hundsvatn, during a strong rain event in August 2008 (Table 2), almost 2 months after the fire (Lydersen et al. 2014). From a pH of 5.32 (H^+ = 4.8 μeq L^{-1}), 9 days after the fire was extinguished, pH dropped to 4.42 (H^+ = 38 μeq L^{-1}) during this event. Simultaneously, the concentration of inorganic, toxic aluminum (Al_i) increased

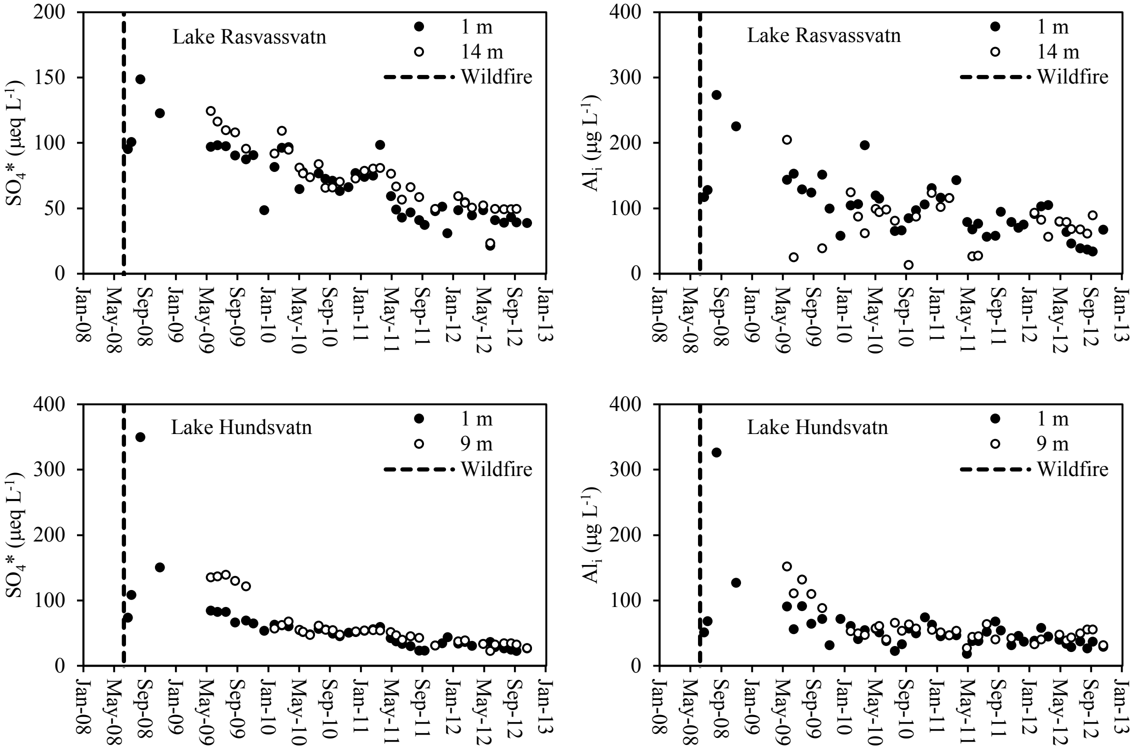

Figure 1. Concentration of non-marine sulfate (SO_4) and inorganic Al (Al_i) in epilimnion (at 1 m) and hypolimnion of Lake Hundsvatn and Lake Rasvassvatn, during the first four postfire years.*

from 51 µg Al L^{-1} to 326 µg Al L^{-1}. At October 30, 2008, after the first postfire autumn turnover, pH in the lake had increased to 4.68 (H^+ = 21 µeq L^{-1}) and Al_i declined to 127 Al µg L^{-1}, figure 1.

During the extreme event in August 2008, the H^+ and Al_i concentrations in Lake Rasvassvatn (17 µeq L^{-1}, pH = 4.8 and 273 µg Al L^{-1}) were significantly lower than in Lake Hundsvatn, but due to minor chemical improvements during the following 1.5 months, the H^+ concentration in Lake Rasvassvatn at October 30, was now only slightly lower (H^+ = 16 µeq L^{-1}) and the Al_i concentration significantly higher (225 µg Al L^{-1}) than in Lake Hundsvatn, table 2. The main reason for the far more adverse water chemical condition in Lake Hundsvatn compared with Lake Rasvassvatn, despite lower acid neutralizing capacity in Lake Rasvassvatn, is likely the far higher catchment to lake area ratio (CA:LA = 18) in this lake, table 1, compared with Lake Rasvassvatn (CA:LA = 1.4). This implies a much higher impact factor of wildfire in Lake Hundsvatn.

In Lake Øyvatn, we have no chemical data from the extreme water chemical episode in August 2008, but the July 2008 data shows almost identical SO_4* concentrations in all three investigated lakes, table 2. However, as the acid neutralizing capacity (ANC) is far higher in Lake Øyvatn compared with the other two lakes, figure 2, the postfire water chemical effects were far less in this lake.

Even though the largest water chemical improvements occurred from August 2008 peak, until spring turnover 2009, less pronounced, but significant, water chemical improvements were recorded the three following postfire years, figure 1 and 2. Both base cations (BC*) and non-marine sulfate (SO4*) continued to decrease, somewhat larger decreases regarding SO4*

	SO_4* (µeq L^{-1})						H^+ (µeq L^{-1})					
Lake	Jun-08	Jul-08	Aug-08	Oct-08	May-09	May12	Jun-08	Jul-08	Aug-08	Oct-08	May-09	May-12
Lake Hundsvatn (epi)	73	108	350	150	84	36	4.8	7.4	38	21	8.5	4.5
Lake Hundsvatn (hypo)					135	23					15	6.5
Lake Rasvassvatn (epi)	95	101	149	123	97	21	8.9	9.5	17	16	10	3.2
Lake Rasvassvatn (hypo)					124	23					10	4.9
Lake Øyvatn (epi)		105			76	44		1.2			0.7	1.3
	Al_i (µg Al L^{-1})						TOC (mg C L^{-1})					
Lake Hundsvatn (epi)	51	68	326	127	91	33	4.9	4.8	5.0	4.8	3.8	5.0
Lake Hundsvatn (hypo)					152	39					3.7	4.4
Lake Rasvassvatn (epi)	117	128	273	225	144	64	3.0	2.6	2.6	3.5	2.2	3.8
Lake Rasvassvatn (hypo)					205	79					3.9	4.7
Lake Øyvatn (epi)		3			15	9		4.9			3.3	4.9

Table 2. Key water chemical parameters in the lakes, during the first post-fire months, and from May 2012, almost four years after the wildfire. Epi = epilimnion at 1 m. Hypo = hypolimnion, i.e. at 9 m in Lake Hundsvatn and at 14 m in Lake Rasvassvatn. Lake Øyvatn is only 3.2 deep and therefore not thermally stratified.

compared with BC*. Despite so, no significant changes in ANC were revealed during this period (Lydersen et al., 2014). Thus, ANC was almost back to postfire levels after spring turnover 2009, one year after the fire. pH and Ali were almost back to postfire levels after autumn turnover 2009, about 1.5 year after the fire.

No hypolimnion data exist for 2008 from the two thermally stratified lakes. Our first hypolimnion data from May 19, 2009 (after spring turnover), showed SO_4* and Al_i concentrations in hypolimnion almost equal to the post-turnover conditions present in the two lakes at October 30, 2008, figure 1. The delay by one turnover period regarding water chemical improvements in hypolimnion versus epilimnion during the three first postfire turnovers means that when the extreme water chemical conditions arose in epilimnion during the August 2008 peak, prefire water conditions were still present in the hypolimnion of the stratified lakes. Thus, fish may have escaped the most adverse conditions by occupying the hypolimnion, but after the autumn turnover in 2008, this opportunity was lost. Accordingly, during the two following turnovers, the best water chemical conditions were present in epilimnion as the chemical improvements in draining waters after wildfire first occur in the top-most layer of thermally stratified lakes. After the autumn turnover 2009, the water chemistry was almost similar in the hypolimnion and epilimnion in the two dimictic lakes, figure 2.

Fish

In 2008, the catch per unit of effort (CPUE) was 67 ind. 100 m^{-2} day^{-1} in Lake Hundsvatn and 16 ind. 100 m^{-2} day^{-1} in Lake Øyvatn. In Lake Rasvassvatn, only two perch were caught in 2008, corresponding to a CPUE of 0.4 ind. 100 m^{-2} day^{-1}.

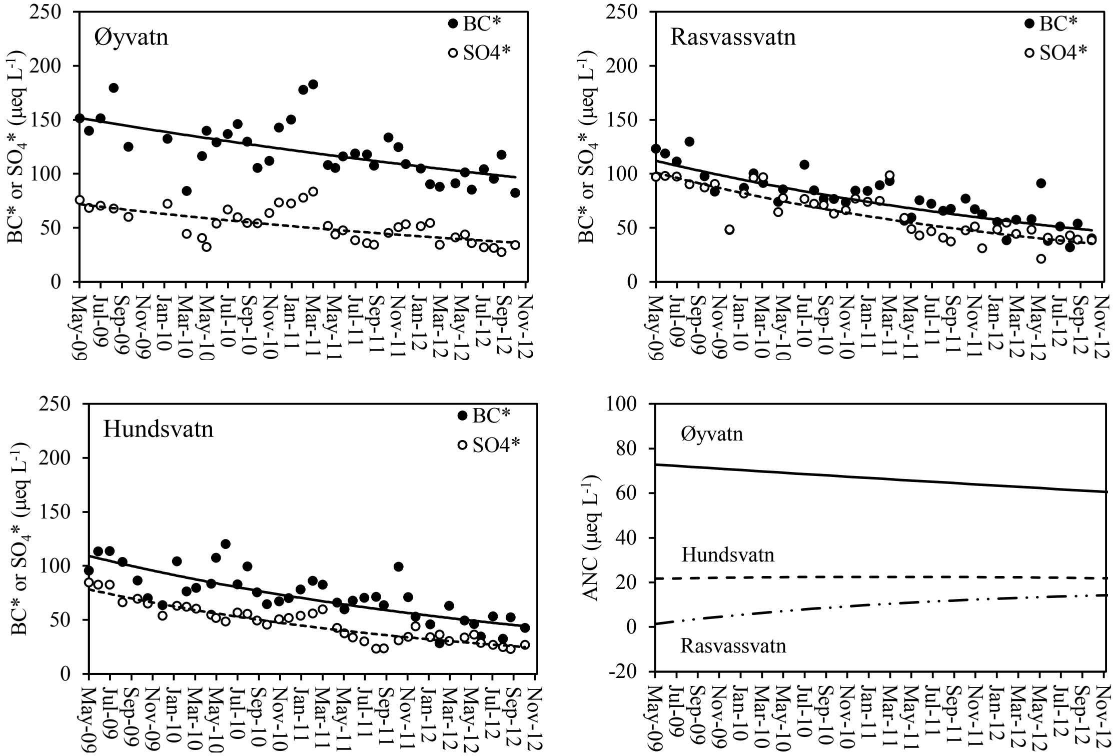

Figure 2. Time trends in non-marine base cations (BC), non-marine sulfate (SO_4*) and acid neutralizing capacity (ANC) in epilimnion of the three wildfire affected lakes, from the first postfire year and the 3 following years.*

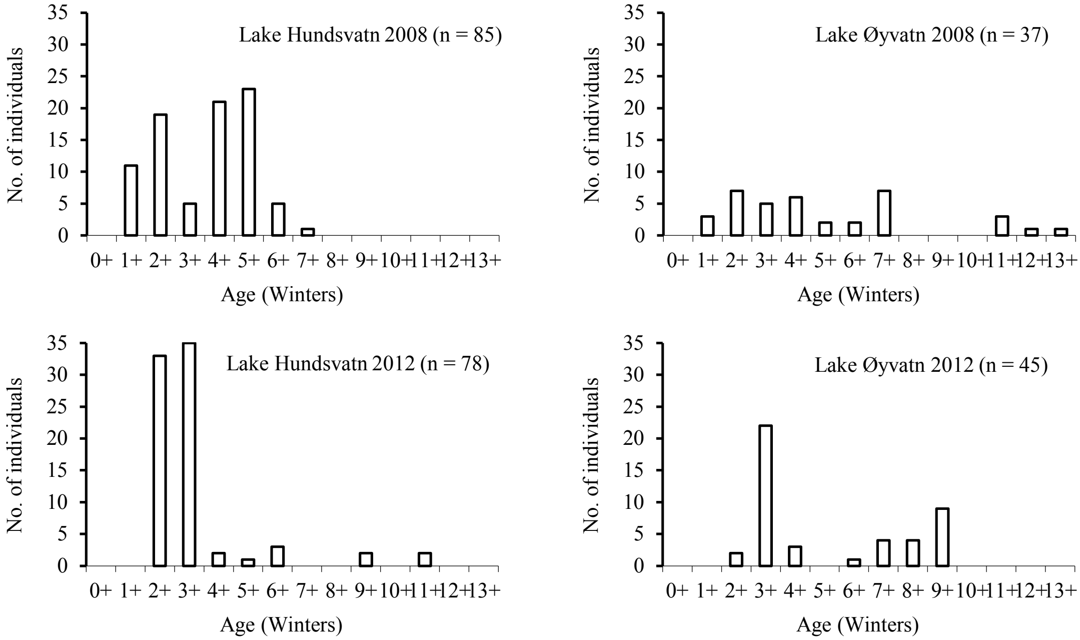

Figure 3. Age distribution of perch in Lake Hundsvatn and Lake Øyvatn at the time of wildfire (2008), and > 4 years after the fire in 2012.

Lake	Øyvatn	Rasvassvatn	Hundsvatn	Øyvatn	Rasvassvatn	Hundsvatn
Year class	2+	2+	2+	3+	3+	3+
N	2	7	33	22	8	35
Length	19.9 ± 1.8	17.0 ± 0.5	16.2 ± 0.6	20.2 ± 2.6	19.2 ± 1.5	17.2 ± 1.1
Weight	94 ± 18	48 ± 6	41 ± 5	102 ± 44	70 ± 20	49 ± 9
K_f	1.19 ± 0.10	0.97 ± 0.04	0.98 ± 0.04	1.15 ± 0.11	0.96 ± 0.06	0.96 ± 0.05

Table 3. Means (± SD) of length (cm), weight (g) and Kf for two year classes of perch caught in the lakes during autumn 2012, > 4 years after the wildfire.

In 2012, CPUE was 26 ind. 100 m^{-2} day^{-1} in Lake Hundsvatn and 13 ind.100 m^{-2} day^{-1} in Lake Øyvatn. In Lake Rasvassvatn, the CPUE in 2012 was 68 ind. 100 m^{-2} day^{-1}, and all individuals were caught in the 21 mm and 26 mm nets.

While four relatively strong year classes (1+, 2+, 4+ and 5+) were present in Lake Hundsvatn in 2008, 2+ and 3+ old perch totally predominated the 2012 catch, figure 3. A relatively strong population of 3+ old perch was present in Lake Øyvatn in 2012, but the changes in year class distribution between 2008 and 2012 was rather small in this lake compared with Lake Hundsvatn.

Due to lack of gillnets with mesh sizes less than 21 mm during gillnet fishing in 2012, and the fact that we often observed several 0+ and 1+ individuals in the lakes in very shore-near, vegetation rich shallow areas (0-50 cm) of the lakes, we were not able to evaluate these year classes by gillnet fishing.

Based on the gill net fishing in 2012, with good catches in all 3 lakes, perch from Lake Øyvatn was significantly older (4.2 ± 2.1 years) than in the two other lakes (3.4 ± 1.0 years in Lake Rasvassvatn and 3.1 ± 1.9 years in Lake Hundsvatn). Comparison of 2+ and 3+ old individuals between lakes, showed significantly larger and higher condition factor (K_f) in perch from Lake Øyvatn compared with the two other lakes, table 3. Also, significantly larger perch were found in Lake Øyvatn compared with Lake Hundsvatn in 2008, comparing the same year classes.

Two sample T-tests revealed significant higher mean weight ($p = 0.026$) and condition factor, K_f ($p < 0.001$), in 2+ old perch from Lake Hundsvatn in 2008 (weight = 46 ± 7 g; K_f = 1.14 ± 0.11; n = 19) compared with 2012 (weight = 41 ± 5 g; K_f = 0.98 ± 0.04; n = 33). Very similar results were revealed for 3+ old individuals, as both length ($p = 0.018$) and weight ($p = 0.025$) were significantly higher in 2008 (length =18.3 ± 0.8 cm; weight: 65 ± 9 g; n = 5) compared with 2012 (length =17.2 ± 1.1 cm; weight: 49 ± 9 g; n= 35). As for the 2+ old perch, the condition factor (K_f) in 2012 was also higher in 2008 (1.04 ± 0.08) compared with 2012 (0.96 ± 0.05), but not significant ($p = 0.092$). In Lake Øyvatn, no significant differences in length, weight or K_f were found within the same year classes in 2008 compared with 2012.

Discussion

The significant increase in 2+ and 3+ old individuals and the low proportion of older year classes in Lake Hundsvatn in 2012 compared with 2008, was a very strong indication of a juvenilzation of the population. This is further strengthened by the fact that the gillnet series used in 2012 (Jensen series), with single nets, varying in mesh from 21 to 45 mm, should significantly overestimate the proportion of larger/older individuals compared with smaller/younger individuals in relation to the Nordic gillnet series used in 2008 (Appelberg et al. 1995), where 58% of the gillnet-area have mesh sizes < 20 mm, a mesh interval not present in the Jensen series. Despite this fact, 88% of the catch in Lake Hundsvatn in 2012 consisted of 2+ and 3+ old individuals, compared with only 28 % in 2008. Comparison of similar year classes in Lake Hundsvatn in 2008 with 2012, also revealed lower average length and weight at age, as well as condition factor (K_f) in 2012 compared with 2008. These observations, in addition to the strong increase in the perch population in Lake Rasvassvatn in 2012 compared with 2008, and the marginal changes in the perch population structure and morphometry in Lake Øyvatn during the same period, might be explained by the difference in the postfire ANC conditions between these lakes, as the postfire effects on nutrients (Tot-N, Tot-P, NO_3^-, NH_4^+), in and between the lakes, were relatively small (Lydersen et al., 2014).

The main physiological effects on fish exposed to acidic aluminum (Al) rich water are disturbance of a) respiratory gas transfer and b) ionic regulation (Neville, 1985; Neville and Campbell, 1988). At pH above 6.0 and with Al present in inorganic forms, the main cause of fish death is hypoxia (Neville, 1985), mostly caused by the high concentration of polymerized Al on gill surfaces. No significant ion loss is observed (Cameron, 1976; McDonald, 1983; McDonald et al., 1983). At pH between 4.0 and 4.5, electrolyte loss seems to be the main cause of death (Neville, 1985; Cameron, 1976; McDonald, 1983), due to the high H^+ concentration. At intermediate pH values, between 4.5 and 6.0, a combination of the two mechanisms occur (Neville, 1985). Even though Al likely is the most investigated metal regarding chemical forms and toxicity to fish, it is very difficult to assess critical concentrations of inorganic, cationic Al (the primary toxic fraction). This is because critical concentrations differs a lot due to variations in physico-chemical factors as pH, water temperature, ionic strength etc., and due to large variation in biological response between fish species, life stages and fish strains (Rosseland and Skogheim, 1984; Lydersen et al., 2002). Based on the literature review by Lydersen et al. (2002), it is reasonable to assume critical concentrations of inorganic Al (Al_i) within the range 20-80 mg Al L^{-1}, where sublethal effects occur at the lowest concentrations. Empirical studies of ANC and fish status in Norwegian lakes (Lydersen et al., 2004), shows a 95% probability for no population damage of perch at an ANC of 14 µeq L^{-1}. Accordingly, the extreme low pH and ANC and the very high concentrations of Al_i measured in

Lake Hundsvatn (pH: 4.42; ANC: -80 µeq L^{-1}; Al_i: 326 mg Al L^{-1}) and Lake Rasvassvatn (pH: 4.76; ANC: -39 µeq L^{-1}; Al_i: 273 mg Al L^{-1}) in August 2008, most likely have been lethal for perch. However, no significant fish death was reported. There might be several reasons for this:

- Lack of documentation of fish death as very few people visited this barren, sooty wildfire area, with locked road barriers all over the area.
- The extreme water chemical episode revealed in epilimnion August 2008 was too short to have substantial lethal impact on fish.
- When the extreme acidification occurred, the two most acid sensitive lakes were both thermally stratified. Thus, fish might have migrated down into the marginal wildfire impacted hypolimnion water.

In Lake Øyvatn, the wildfire-induced acidification was too marginal to cause any lethal effects on fish, as the original pH and ANC levels in this lake were significantly higher than in the two other lakes.

Whereas significant numbers of perch may have fled the extreme water chemical conditions in the epilimnion of Lake Hundsvatn in August 2008, by moving down into the hypolimnion water, this was not possible after the autumn 2008 turnover, as the total water column now was completely mixed. The pH, Al_i and ANC between autumn 2008 turnover and spring 2009 turnover, were still at levels where negative effects on perch should be expected. This is well supported by our fish data, which strongly indicates a significant dieback of perch during the first postfire year, as all age classes present in 2008 were almost absent in the catch from 2012, figure 3. However, the surviving fish produced large numbers of the 2009 (3+) and 2010 (2+) year classes. As these year classes were strong, a very strong intraspecific competition, especially for food, should be expected. This might explain the decrease in size (length, weight) and condition factor (K_f) found in the two first year classes born after the first postfire year, compared with corresponding individuals present before the wildfire. In Norwegian lakes, both Linløkken et al. (1991) and Forseth et al. (1997) have documented that both chronic and acute acidic episodes may imply higher mortality rates among older perch, which may cause fewer year classes present in the population, shorter intervals between year classes of perch and a temporary juvenilization of the stock.

The significant increase in CPUE of perch in Lake Rasvassvatn from 2008 to 2012, strongly indicates increased abundance of perch in this lake, 4 years after the fire. This is most likely a response to the improved water chemical conditions. In 2012, during the period May - October, median pH value in Lake Rasvassvatn was 5.33, median Al_i concentration 52 mg Al L^{-1}, and median ANC value 12 µeq L^{-1}. Empirical data on water chemistry and fish in Norwegian lakes (Lydersen et al., 2002; Lydersen et al., 2004), suggest that the water chemical condition in Lake Rasvassvatn in 2012 was at a level where establishment of a self-recruiting perch population should be possible. The decline in acid rain in southern Norway during the last decades (Aas et al., 2013) has resulted in a general improvement in water chemistry in lakes (Garmo et al., 2013). The relatively strong increase in ANC in Lake Rasvassvatn, the last postfire years (Figure 3), indicates that a long-term effect of the wildfire might have contributed to a somewhat faster water chemical recovery in this lake. We do not know if the perch in Lake Rasvassvatn were born in the lake or immigrated from the neighbor lake, but as the catch covered five year classes (1+ to 5+), with predominance of 4+ old individuals, we assume that most of the perch caught in 2012 were immigrants. This is also supported by stomach analyses, which revealed a total dominance of the large planktonic crustacean *Bythotrephes longimanus*. The species is quite plastic and can tolerate a wide range of salinities, pH, temperatures, and conductivities, but prefer shallow, oligotrophic, fishless ponds, pools, and lakes (Grigorovich et al., 1998; Pangle and Peacor, 2009). Thus, likely an establishment of a permanent perch population will almost eliminate the *B. longimanus* population by predation.

Our fish data is different from what reported by St-Onge and Magnan (2000), who reported significantly lower proportion of small yellow perch (*Perca flavescens*) in wildfire impacted stratified headwater lakes in Canada. As wildfire may cause very different physical and chemical response in lakes, due to differences in many physico-chemical watershed and lake factors (Lydersen et al., 2014), also large variations in biological responses to aquatic organisms should be expected. As biogeochemical documentation of wildfire effects on freshwaters are manifold and very limited regarding biological effects in lakes, hopefully more knowledge will appear within this field in the years to come.

Conclusions

Climate change with higher temperatures and less precipitation in summer, imply more severe drought and subsequent higher risks of wildfires in large areas of the world. Thus, heavy rainfall following such events, may likely cause more frequent and more extreme water chemical episodes in such areas in the years to come. However, as the amount of acid rain, primarily sulfuric acid, has declined significantly during the latest decades in many acid sensitive areas of the world (Monteith et al., 2007), the water chemical conditions during heavy rain episodes following drought, are much less severe today compared with similar events during the most impacted acid rain period, 1970-1990 (Lydersen et al., 2014). Thus, much more extreme water chemical conditions, with subsequent adverse effects on fish, will normally occur during the first postfire storm events compared with corresponding post-draught events only. Most vulnerable lakes are those with short residence time and low acid neutralizing capacity.

Acknowledgement

This study is part of a forest fire project (Pyrowater) funded by the Norwegian Research Council (NRC) during the period 2009-2012. We greatly appreciate the Norwegian Water Research Institute (NIVA), for funding and initiating water chemical investigations and gillnet fishing during the very essential, initial post-fire period. We also want to acknowledge the technical staff at Telemark University College, Bjørn Steen and Karin Brekke Li, who have conducted almost all the water chemical analysis and Dick Wright (NIVA) for his fruitful scientific and linguistic contribution during the preparation of this manuscript.

References

Aherne, J., Larssen, T., Cosby B.J., Dillon, P.J. (2006). Climate variability and forecasting surface water recovery from acidification: Modelling drought-induced sulfate release from wetlands. The Science of Total Environment 365(1-3): 186-99.

Appelberg, M., Berger, H.-M., Hesthagen, T., Kleiven, E., Kurkilahti, M., Raitaniemi, J., Rask, M. (1995). Development and intercalibration of methods in Nordic freshwater monitoring. Water Air & Soil Pollution 85: 401-406.

Barnes, R.B. (1975). Determination of species forms of aluminum in natural water. Chemical Geology 15(3): 177-191.

Bozek, M.A. & Young, M.K. (1994). Fish mortality resulting from delayed effects of fire in the greater Yellowstone ecosystem. Great Basin Naturalist 54(1): 91–95.

Cameron, J.N. (1976). Branchial ion uptake in arctic grayling: resting values and effects of acid-base disturbance. Journal Experimental Biology 64: 711-725.

Carignan, R., D'Arcy, P., Lamontagne, S. (2000). Comparative impacts of fire and forest harvesting on water quality in Boreal Shield lakes. Canadian Journal of Fisheries & Aquatic Sciences 57: 105-117.

Cronan, C.S., Schofield, C.L. (1979). Aluminum leaching response to acid precipitation: Effects on high elevation watersheds in the Northeast. Science 204: 304-306.

Dickson, W. (1978). Some effects of the acidification of Swedish lakes. Verhandlungen des Internationalen Verein Limnologie 20: 851-856.

Driscoll, C.T. (1984). A procedure for the fractionation of aqueous aluminum in dilute acidic waters. International Journal of Environmental Analytical Chemistry 16(4): 267-283.

Flannigan, M.D. & Wotton, B, M. (2001). Climate, weather, and area burned. In: Johnson, E.A., Miyanishi, K., editors. Forest Fires – Behavior and Ecological Effects, Academic Press, San Diego, 351–373.

Forseth, T., Halvorsen, G.A., Ugedal, O., Fleming, I., Schatau, A.K.L., Nøst, T., Hartvigsen, R., Raddum, G., Mooij, W., Kleiven, E. (1997). Biological status in limed lakes. Norwegian Institute for Nature Research (NINA). Commission Report 1997-508, 52 pp (in Norwegian, English summary).

Fulton, T.W. (1904). The rate of growth of fishes. 22nd Annual Report of the Fishery Board for Scotland 3: 141-241.

Garmo, Ø. Skancke, L.B., Høgåsen, T. (2012). Monitoring long-range transboundary air pollution – Water chemical effects 2012. Report 6515-2013 (TA-3033/2013), 70 pp (in Norwegian, English summary).

Grigorovich, I.A., Pashkova, O.V., Gromova, Y.F., van Overdijk, C.D.A. (1998). *Bythotrephes longimanus* in the Common wealth of the independent states: variability, distribution, and ecology. Hydrobiologia 379: 183-198.

Goldammer, J.G., Price, C. (1998) Potential impacts of climate change on fire regimes in the tropics based on MAGICC and a GISS GCM-derived lightning model. Climate Change 39: 273-296.

Huitfeldt-Kaas, H. (1922) Causes to mass death of salmon and brown trout in the River Frafjordelva. River Helleelva and River Dirdalselva in Ryfylke, autumn 1920. Norges Jaeger og Fiskerforenings tidsskrift 51: 37-44 (in Norwegian).

Høgberget, R. (2010). The wildfire in Mykland 2008. Results after 2 years monitoring of chemical effects in lakes. Norwegian Institute for Water Research. Report-5979, 44 pp (in Norwegian, English summary).

IPCC (2001). Climate change 2001: Impacts, adaptation and vulnerability, Chapter 5: Ecosystems and their goods and services. In: McCarthy, J.J., Canziani, O.F., Leary, N.A., Dokken, D.J., White, K.S., editors. Contribution of Working Group II to the Third Assessment Report of the Intergovernmental Panel on Climate Change, Cambridge University Press, Cambridge, NY.

IPCC (2014). Climate change 2014: Impacts, adaptation, and vulnerability. Part A: Global and Sectorial Aspects. In: Field, C.B., Barros, V.R., Dokken, D.J., Mach, K.J.J., Mastrandrea, M.D., Bilir, T.E., Chatterjee, M., Ebi, K.L., Estrada, Y.O., Genova, R.C., Girma, B., Kissel, E.S., Levy, A.N., MacCracken, S., Mastrandrea, P.R., White, L.L., editors. Contribution of Working Group II to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, NY.

Jensen, K.W. (1977). On the dynamics and exploitation of the population of brown trout, *Salmo trutta* L., in Lake Heimdalsvatn, southern Norway. Report from the Institute of Freshwater Research, Drottningholm 56: 18-69.

Jeffries, D.S., Clair, T.A., Couture, S., Dillon, P.J., Dupont, J., Keller, W., McNicol, D.K., Turner, M.A., Vet, R., Weeber, R. (2003). Assessing the recovery of lakes in southeastern Canada from the effects of acidic deposition. Ambio 32(3): 176-182.

Laudon, H., Dillon, P.J., Eimers, M.C., Semkin, R.G., Jeffries, D.S. (2004). Climate-induced episodic acidification of streams in central Ontario. Environmental Science & Technology 38(22): 6009-6015.

Lea, E. (1910). On the methods used in herring investigations. Publs. circonst. Cons. Perm. Int. Explor. Mer. 53: 7-174.

Linløkken, A., Kleiven, E., Matzow, D. (1991). Population structure, growth and fecundity of perch (*Perca fluviatilis* L.) in an acidified river system in Southern Norway. Hydrobiologia 220: 179-188.

Lydersen, E., Høgberget, R., Moreno, C.E., Garmo, Ø.A., Hagen, P.C. (2014). The effects of wildfire on the water chemistry of dilute, acidic lakes in southern Norway. Biogeochemistry 119: 109-124.

Lydersen, E., Larssen, T., Fjeld, E. (2004). The influence of total organic carbon (TOC) on the relationship between acid neutralizing capacity (ANC) and fish status in Norwegian lakes. The Science of Total Environment 326(1-3): 63-69.

Lydersen, E., Löfgren, S., Arnesen, R.T. (2002). Metals in Scandinavian surface waters: Effects of acidification, liming, and potential reacidification. Critical Reviews in Environmental Science & Technology 32: 73-295.

Mcdonald, D.G. (1983). The interaction of environmental calcium and low pH on the physiology of the rainbow trout, *Salmo gairdneri*.1. Branchial and renal net ion and H^+ fluxes. Journal of Experimental Biology 102: 123-140.

Mcdonald, D.G., Walker, R.L., Wilkes, P.R.H. (1983). The interaction of environmental calcium and low pH on the physiology of the rainbow trout, *Salmo gairdneri*. 2. Branchial ionoregulatory mechanisms. Journal of Experimental Biology 102: 141-155.

McMahon, T.E. & de Calesta, D.S. (1990). Effects of fire on fish and wildlife: In: Walstad, J.D., Radosevich, S.R., Sandberg, D.V, editors. Natural and prescribed fire in Pacific Northwest forests 1990; Corvallis, Oregon State University Press, 233-250.

Minshall, G.W. & Brock, J.T. (1991). Observed and anticipated effects of forest fire on Yellowstone stream ecosystems. In: Leiter RB, Boyce MS, New Haven CT, editors. The greater Yellowstone ecosystem: redefining American's wilderness heritage 1991; Yale University Press, 123-135.

Miranda, A.I. (1994). Forest fire emissions in Portugal: a contribution to global warming? Environmental Pollution 83: 121-123.

Monteith, D.T., Stoddard, J.L., Evans, C.D., de Wit, H.A., Forsius, M., Høgåsen, T., Wilander, A., Skjelkvåle, B.L., Jeffries, D.S., Vuorenmaa, J., Keller, B., Kopácek, J., Vesely, J. (2007). Dissolved organic carbon trends resulting from changes in atmospheric deposition chemistry. Nature 450: 537-540.

Neville, C.M. (1985). Physiological response of juvenile rainbow trout, *Salmo gairdneri*, to acid and aluminum – Prediction of field responses from laboratory data. Canadian Journal of Fisheries & Aquatic Sciences 42: 2004-2019.

Neville, C.M., Campbell, P.G.C. (1988). Possible mechanisms of aluminum toxicity in a dilute, acidic environment to fingerlings and older life stages of salmonids. Water Air & Soil Pollution 42: 311-327.

Pangle, K.L., Peacor, S.D. (2009). Light-dependent predation by the invertebrate planktivore *Bythotrephes longimanus*. Canadian Journal of Fisheries & Aquatic Sciences 66: 1748-1757.

Pechony, O., Shindell, D. (2010). Driving forces of global wildfires over the past millennium and the forthcoming century. Proceedings of the National Academy of Sciences of the United States of America 107(45): 19167-19170.

Price, C., Rind, D. (1994). Possible implications of global climate change on global lightning distributions and frequencies. Journal of Geophysical Research 99: 108–123.

Reiman, B.E., Lee, D., Chandler, G., Myers, D. (1997). Does wildlife threaten extinction for salmonids: responses of redband trout and bull trout following recent large fires on the Boise National Forest. Proceedings – Fire Effects on Rare and Endangered Species and Habitats Conference, Nov 13-16, 1995, Coeur d' Alene, Idaho, IAWF, 47–57.

Reuss, J.O., Johnson, D.W. (1986). Acid deposition and the acidification of soils and waters. Ecological Studies (Springer) 59: 62-65.

Rinne, J.N., Carter, C.D. (2008). Short-term effects of wildfires on fishes in the southwestern United States, 2002: management implications. In Proceedings of the 2002 Fire Conference: Managing fire and fuels in the remaining wildlands and open spaces of the Southwestern United States. Gen. Tech. Rep. PSW-GTR-189. Albany, CA, U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station, 167-174.

Rosseland, B.O., Skogheim, O.K. (1984). A comparative study of salmonid fish species in acid aluminum-rich water. II. Physiological stress and mortality of one- and two-year-old fish. Annual Report from the Institute of Freshwater Research. Drottningholm, Report No. 61: 186-194.

Schär, C., Vidale, P.L., Lüthi, D., Frei, C., Häberli, C., Liniger, M.A., Appenzeller, C. (2004). The role of increasing temperature variability in European summer heatwaves. Nature 427: 332-336.

Smith, H.G., Sheridan, G.J., Lane, P.N.J., Nyman, P., Haydon, S. (2011). Wildfire effects on water quality in forest catchments: a review with implications for water supply. Journal of Hydrology 396(1-2): 170-192.

Stocks, B.J., Fosberg, M.A., Lynham, T.J., Mearns, L., Wotton, B.M., Yang, Q., Jin, J.Z., Lawrence, K., Hartley, G.R., Mason, J.A., McKenney, D.W. (1998). Climate change and forest fire potential in Russian and Canadian boreal forests. Climate Change 38: 1-13.

St-Onge I. & Magnan, P. (2000). Impact of logging and natural fires on fish communities of Laurentian Shield Lakes. Canadian Journal of Fisheries & Aquatic Sciences 57(suppl. 2): 165-174.

USDA (2000). Wildland fire in ecosystems. Effects of fire on flora. US Department of Agriculture Forest Service. Gen. Techn. Rep. RMRS-GTR-42, Vol. 2.

USDA (2002). Wildland fire in ecosystems. Effects of fire on soil and water. US Department of Agriculture Forest Service. Gen. Tech. Rep. RMRS-GTR-42, Vol. 4.

Weast, R.C. (1988). CRC Handbook of Chemistry and Physics, 1st Student edition. CRC Press, Inc. Boca Raton, Florida, p F-100.

Aas, W., Solberg, S., Manø, S., Yttri, K.E. (2013). Monitoring of long range transported air pollutants. Annual Report for 2012. SPFO no. 1148, 211 pp. (in Norwegian, English summary).