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Acidification effects in forest soils, soil water, groundwater and stream water following clear-cutting and above-ground biomass removal

-a literature review

Institutionen för vatten och miljö, SLU
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Preface

Over the years the Swedish Energy Agency has funded research and development in the field of energy, including the effects of intensified forestry and different mitigating measures e.g. ash return. In 2009 the project “Acidification of surface waters due to forest biofuel extraction - development of a classification tool” was granted fundings by the Swedish Energy Agency from the third research and framework development program 2007-2010 “Sustainable supply and upgrading of biomass feedstock for energy purposes”. Project leader is Associate Professor Stefan Löfgren at the Swedish University of Agricultural Sciences, department of Aquatic sciences and Assessment. Financial support is also given to post graduate studies whitin this project. The work is carried out in collaboration between the Swedish University of Agricultural Sciences and IVL Swedish Environmental Research Institute.

The aim of the project is to develop a tool for estimating the long-term sensitivity of surface waters to acidification from forest biomass extraction due to depletion of base cations. The EU Water Framework Directive, the energy policy, the environmental quality objective “only natural acidification” and the sector objective to reduce forestry-related acidification have all highlighted the potential of forest biomass extraction acidification effects on lakes and streams.

This literature review is part of the project and will provide state of the art information about the acidification effects following clear-cutting and biomass removal. When possible, the quantitative importance is discussed. Mechanisms responsible for the observed changes are identified and evaluated in terms of their importance from a whole-catchment perspective. The report is primarily directed towards the forestry and energy sectors.

Therese Zetterberg and Stefan Löfgren

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Summary

Today, Sweden is the leading user of renewable energy in the EU as a result of various energy policies as well as major indigenous sources of renewable energy such as biofuels. The use of renewable energies is intended to increase even further in the future while reducing the use of fossil fuels. Increased utilization of logging residues (above- and belowground) removes valuable nutrients and buffer capacity from the forests, which could prevent or delay a recovery from acidification in soils and surface waters. As of today, the relationship between whole-tree harvesting and long-term surface water acidification is unclear.

The objective of this literature review is to describe acidification effects in forest soils, soil water, groundwater and surface waters following clear-cutting and above-ground biomass removal by examining the empirical results from scientific studies presented in the literature. In addition, cutting effects on hydrology are discussed. The mechanisms responsible for the results are explained and discussed in terms of site-specific conditions (catchment characteristics and hydrology). When possible, treatment differences between whole-tree and stem-only harvest are discussed. The results are assessed in relation to previous reviews in order to improve our current understanding. The review describes research carried out mainly in Europe, the US and Canada but studies from New Zeeland and South Australia have been considered although climate, soil types and forest practices may differ. The review will provide increased understanding of the causes and effects of logging harvest residues in forested catchments and it is primarily directed towards the forestry and energy sectors.

Following the abrupt removal of trees in final cutting, the soil properties will be affected resulting in increased or decreased concentrations, pools and fluxes of most constituents. The quantitative importance and duration of these changes differ between sites. The treatment differences between whole-tree and stem-only harvest are difficult to quantify, which suggests that the effect of cutting per se is more important than differences in harvest intensity, especially during the clear-cut phase. Treatment effects can usually be detected in soils, soil water, groundwater and stream water. In general, short-term effects (equal to the clear-cut phase) are well studied, especially in forest soils and stream water since this is usually the time when the largest changes take place. Groundwater studies are largely lacking. Clear-cutting effects should also be viewed in a catchment perspective why soil studies should be combined with soil water, groundwater and stream water studies. This type of studies, preferably complemented with measurements of deposition, decomposition and mineralization, should therefore be prioritized in future research. In addition, long-term effects are rarely described due to the lack of long-term data. There are no studies describing treatment effects older than 31 years. Considering normal rotation times of forests (60-120 years) and that it may take decades for soil changes to become detectable, continued monitoring in already established research sites should be supported. This becomes even more apparent when considering the use of steady-state mass-balances and dynamic modelling for predicting long-term changes. According to the model simulations, clear-cutting, especially whole-tree harvest, have a long-term negative impact on soil nutrients which cannot always be compensated for by weathering processes and deposition. However, based on the empirical evidence assessed in this review, the short-term effect of clear-cutting final felling is relatively small even though there are sites where the effect of cutting has been considerable. The long-term effects are largely unknown. Thus, there is a discrepancy between the empirical evidence and model predictions.

1 Introduction

The energy use in Sweden has continued to increase since the Second World War owing to an import of fossil fuels and the development of nuclear power stations and hydropower production. However, the 1973's oil crises lead to awareness in finding renewable energy sources to reduce the fossil fuel dependency. Driving forces during the 1980's and 1990's has been acidification and climate change. As a result, the supply of electricity (from hydro- and nuclear power) has increased by 145 % and the supply of biofuels by 179 % since the 1970's (Swedish Energy Agency, 2008). The supply of oil products has decreased by 43 % but at the same time the demand for energy has increased by 7.7 % (from 375 to 404 TWh) and the total energy supply by 36.5 % (from 457 to 624 TWh).

Today, Sweden is the leading user of renewable energy in the EU as a result of various energy policies as well as major indigenous sources of renewable energy such as biofuels. In 2007, the total supply of biofuels, peat, waste etc. amounted to 120 TWh (or 19 %) used primarily in district heating plants, electricity production, forest product industry and for heating of residential buildings (Swedish Energy Agency, 2008). The supply of renewable energy (i.e. logging residue fuel) from the forest was 7.3 TWh in 2007 according to the Swedish Forest Agency (2008a). In the future, this amount could increase to 15 TWh using existing techniques, new methods and harvest of stumps could yield another 21-34 TWh (Swedish Forest Agency, 2008b).

The use of renewable energies is expected to increase even further in the future while reducing the use of fossil fuels. Nationally, the Swedish Parliament has adopted 16 national environmental quality objectives where the significant environmental problems should be solved by 2020 (governement bill 1997/98:145). One of these goals is "Reduced climate impact" whereby emissions of greenhouse gaseous should be reduced by 4 % compared with the concentrations in 1990. In Europe, the European Council has agreed upon an integrated energy and climate change strategy (COM, 2007) including an overall binding target of 20 % renewable energies by the year 2020 compared with 8.5 % in 2005. By the same year, the greenhouse gas emissions must be reduced by at least 20 % compared with 1990. However, the ability to cut emissions by replacing fossil fuels with renewables varies within the EU. Therefore, the Commission of the European Communities has proposed differentiated targets for each EU member state and the national overall target for Sweden in final consumption of renewable energy is set to 49 % compared with 39.8 % in 2005 (COM, 2008).

Increased utilization of logging residues (above- and belowground) removes valuable nutrients and buffer capacity from the forest which could prevent or delay a recovery from acidification in forest soils and surface waters (Swedish Environmental Protection Agency, 2007a). Hence, a new interim target for the acidifying effects of forestry has been proposed: "By 2015 the acidifying effects of forestry in acidified areas will not exceed what is compensated for by natural processes" concerning the national environmental objective "Natural Acidification Only" (Swedish Environmental Protection Agency, 2007a).

The negative impact of forestry is also governed by the Water Framework Directive which states that all of the European member states are committed to achieve good qualitative and quantitative status of all water bodies by 2015 (OJ L327, 2000). The goal is to “implement the necessary measures to prevent deterioration of the status of all bodies of surface water” and to “protect, enhance and restore all bodies of surface water” (op. cit.). Agricultural areas and forests are pointed out as two examples of land use patterns which could have a significant impact on surface waters, and if so, the Water Authorities have the means to take actions through the municipalities and the government authorities which carries out the necessary measures. In 2009, the Swedish Water Authorities presented for the first time plans for each of the five water districts. In the proposal from Västerhavets water district, forestry is discussed as a contributor to acidification of surface waters when base cations are removed from the forest through logging residue fuel (County Administration Board, 2009). Hence, if logging of residues has a negative impact on the pools of base cations in the soils, leading to surface water acidification, the loss must be compensated for by lime or wood ash. Alternatively, collecting logging residue fuels may be restricted in the most sensitive areas.

To summarize, the logging of forest residues will continue to increase according to binding energy policies. At the same time the water framework directive states that surface waters must be protected from acidification via the removal of base cations. As of today, the relationship between whole-tree harvesting and long-term surface water acidification is unclear.

The goal of this literature review is to:

- Describe short- and long-term acidification effects in forest soils, soil water groundwater and surface waters following conventional clear-cutting (CH) and clear-cutting together with above-ground biomass removal (WTH).
- Describe short- and long-term changes in water tables and runoff following CH and WTH.
- Describe short- and long-term treatment differences between CH and WTH when specified.
- Provide increased understanding of the causes and effects of logging harvest residues in forested catchments depending on catchment characteristics and hydrological flow paths.
- Assess the results in relation to previous reviews in order to improve our current understanding.

The review is mainly directed towards the forestry and energy sectors. It begins with an overview of the current acidification status in Sweden and a description of the potential effects of whole-tree harvesting. Next, acidification status is discussed followed by an examination of catchment characteristics and hydrology. This background information is necessary for understanding harvesting effects on forest soils and how it links to surface water quality. In the second part of this review, clear-cut effects with and without logging of residue fuel are reviewed beginning with treatment effects in forest soils, soil water, groundwater and surface waters. The results are assessed in relation to previous reviews ending with conclusions.

2 Background

2.1 Current acidification status in Sweden

Atmospheric deposition of sulfur (S) and nitrogen (N) during the last century has caused acidification of Swedish forest soils and surface waters, especially in the southwestern parts where the base cation soil storage has decreased considerably in the soils (Eriksson et al. 1992). For example, Falkengren-Grerup et al. (1987) observed a decrease in base saturation by 30-50 % and a decline in pH by more than one unit in deep soil horizons during 1949-1985. Changes in soil pH have also been reported by Tamm & Hallbäcken (1986; 1988). Documented acidification of surface water and lakes before 1980 is rare, but Renberg et al (1993) used paleolimnological data to link atmospheric deposition and acidification of clear water lakes from 1900 to present day.

The emissions of acidifying pollutants have been reduced and as a result the deposition of S and N has decreased (Figure 1). Today, the total S deposition is below 6 kg ha^{-1} yr^{-1}, except for the south-western parts (Figure 2). The total N deposition has also decreased but not to the same extent. The N deposition pattern, however, follows that of S. Areas most affected by S and N deposition coincide with areas receiving relatively high deposition of sea salts (NaCl) due to the close proximity to the sea. Sodium (Na$^+$) participates in exchange processes with hydrogen (H$^+$) and aluminium ions (Al^{n+}) while chloride (Cl$^-$) acts as a mobile anion and is believed to move through the soil relatively unaffected (Öberg, 1998). During heavy storm events, sea salt deposition can cause episodic acidification in streams and lakes (e.g. Wigington et al. 1996a; Wright et al. 1988) and in the most severe cases lead to fish death (Hindar et al. 1994).

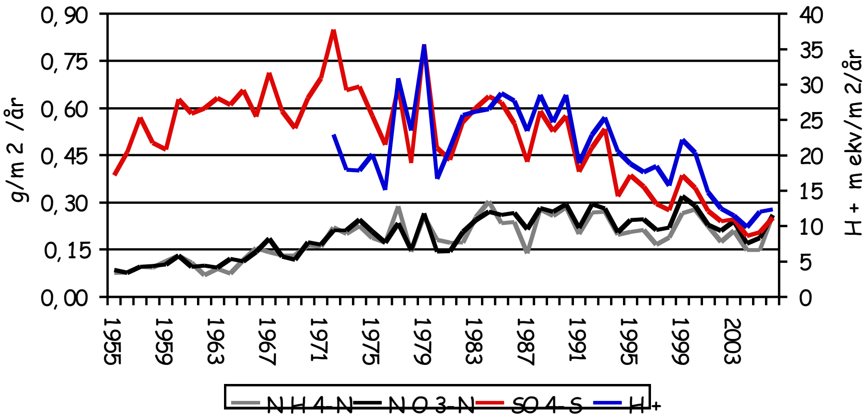


Figure 1. Wet deposition of NH$_4$-N, NO$_3$-N, SO$_4$-S and H$^+$ in the middle part of Sweden expressed on a yearly basis for the period 1955-2006. From MISU and IVL Swedish Environmental Research Institute.

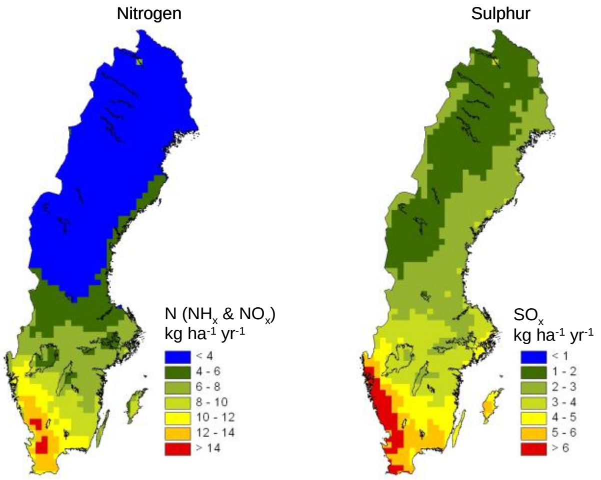


Figure 2. Total deposition of N and S in forests expressed as a mean during the period 2003-2005. The maps are based on MATCH ("Swedish model") with a resolution of 20*20 km. From IVL Swedish Environmental Institute.

2.1.1 Forest soils

Forest soil and surface water surveys in Sweden and elsewhere in Europe show clear signs of recovery from acidification as a result of the emission reductions (e.g. Fölster & Wilander, 2002; Wilander & Lundin 2000; Skjelkvåle et al., 2001ab; Stoddard et al., 1999). According to the Swedish Environmental Protection Agency (2007a) *"the trend towards increased acidification of forest soils"* has been reversed. The status and development of forest soil acidity is described using a combination of pH_{H2O}, effective base saturation (V_{eff}) and extractable aluminium (Al_{ex}) in the B-horizon which groups forest soils into one of the following classes: 1-2 = low, 3 = moderate, 4 = high, and 5 = very high acidity. During the period 1985-87 to 1999-03 the percentage of forest soils belonging to class 4 and 5 decreased from 48 % to 21 % (Figure 3) (Swedish Environmental Protection Agency, 2007b). The decrease is most pronounced in the southwestern parts of Sweden, which historically has received the highest deposition (Figure 4). Areas not as affected by anthropogenic deposition and/or with a good buffer capacity show a limited positive change (i.e. the northern parts of Sweden and calcium rich areas such as the island of Gotland).

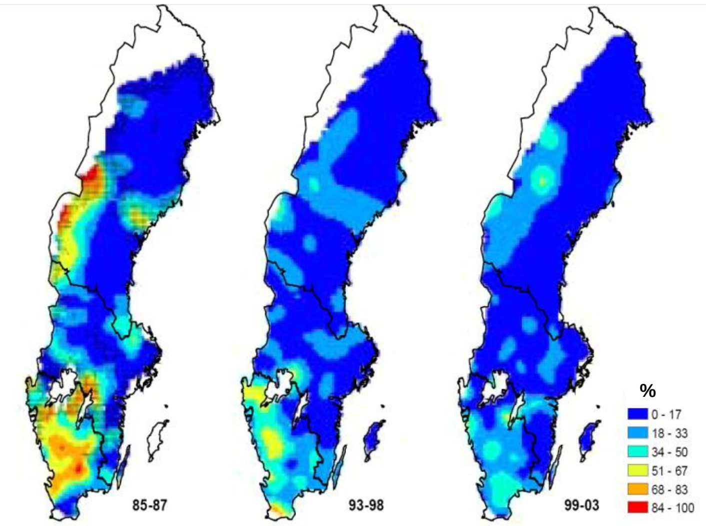


Figure 3. The status and development of forest soil acidification during the period 1985-87, 1993-98 and 1999-03. The figure shows the percentage of soils that belong to acidification class 4 (=high acidity) and 5 (=very high acidity) based on pH, base saturation and extractable aluminium in the B-horizon. From the Swedish Environmental Protection Agency (2007a).

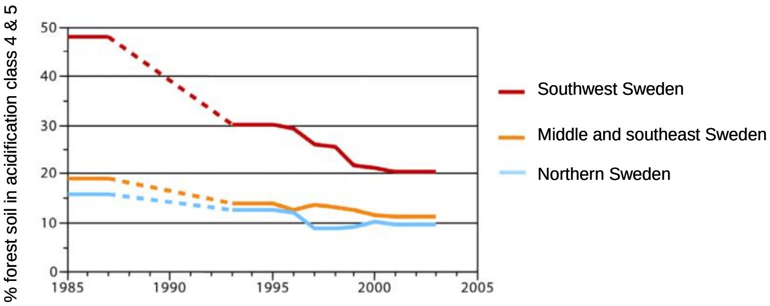


Figure 4. The percentage of forest soils in different parts of Sweden that belong to acidification class 4 (=high acidity) and 5 (=very high acidity) based on pH, base saturation and extractable aluminium in the B-horizon. From the Swedish Environmental Protection Agency (2007a).

2.1.2 Surface waters

The decrease in non-marine S concentrations has led to an increase in ANC and accompanying increases in pH and decreases in Al^{n+} (Evans et al. 2001). The increase in ANC has however been offset by a decrease in base cations, primarily Ca^{2+} (Löfgren et al. 2009a). Environmental monitoring data from 80 Swedish lakes show decreasing trends in sulfate (SO_4^{2-}) and base cations and increasing trends in ANC during 1990-2005 (Figure 5). This is accompanied by a small increase in pH, partly offset by the increasing TOC trend.

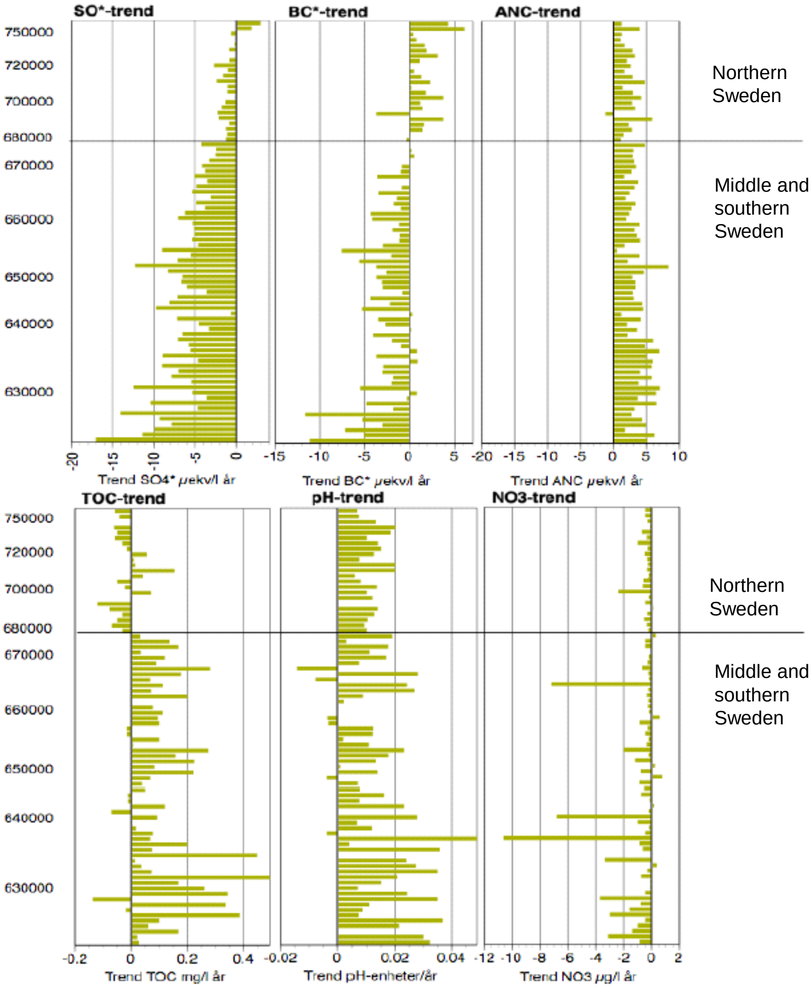


Figure 5. Average annual trends in surface water chemistry 1990-2005 based on data from 80 lakes. From the Environmental Protection Agency, 2007a.

2.2 Potential effects of whole-tree harvesting

Model simulations indicate that it may take decades or even centuries for some forest areas to fully recover from acidification unless further S and N emission reductions are undertaken (Sverdrup et al., 2005). According to the simulations, the acidification status is made worse by extraction of biofuels from forests. Harvesting of logging residues may lower the use of fossil fuels but in return it removes base cations from the forest, unless wood ash is brought back.

There have been several attempts to quantify the long-term sustainability of forest soils following different intensities of forest harvesting using simple input-output budgets (steady-state models). The results from these budget calculations are interpreted as a non-sustainable system if the output of nutrients exceeds the input to the system. For example, Olsson et al. (1993) and Olsson (1996a) demonstrated in a *regional* comparison that weathering alone can support the loss of Mg^{2+} and K^{+} via plant uptake but not Ca^{2+} when stem-only harvest is practiced. In contrast, whole-tree harvesting leads to losses of all three cations, especially in the southwestern parts of Sweden. Input of nutrients via atmospheric deposition and output via leaching losses were not included in these regional budget calculations. A *national* mass balance budget calculation, including atmospheric deposition and leaching losses, suggests that the average total base cation difference between supply and loss was nearly twice as high following whole-tree harvesting compared with stem-only (-0.62 keq ha^{-1} yr^{-1} and -0.33 keq ha^{-1} yr^{-1}, respectively) and thereby *"risk depletion of the soil in less than one to two rotation periods almost anywhere in Sweden"* (Sverdrup & Rosén, 1998).

Akselsson & Westling (2005) and Akselsson et al. (2007) further developed the national mass balance model by using the same approach as above. The results from these studies showed that weathering and deposition alone cannot compensate for losses of Ca^{2+} and Mg^{2+} for large parts of Sweden regardless of harvesting method (Figure 6 and Figure 7). The losses are less in Scots pine stands (*Pinus sylvestris*) compared with Norway spruce (*Picea abies*) stands. Potassium losses following stem-only harvesting in Scots pine stands are compensated for via weathering and deposition but not in Norway spruce stands following whole-tree harvesting (Figure 8).

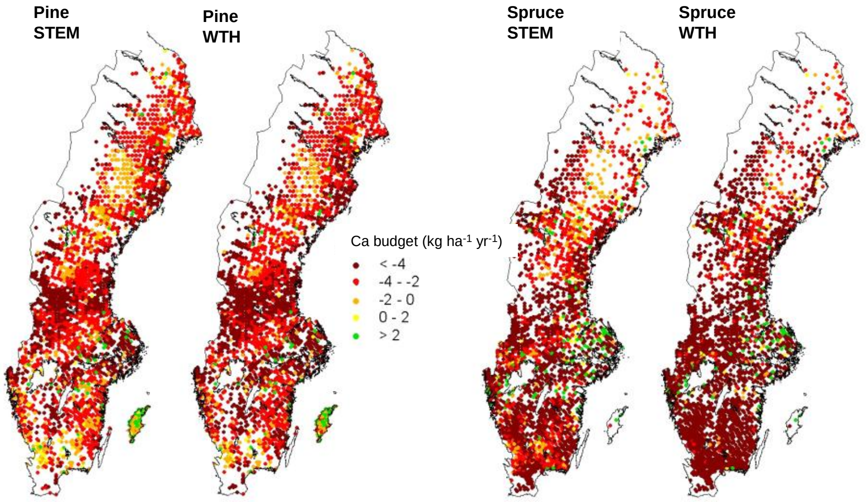


Figure 6. Mass balance budget calculations for Ca^{2+} (kg ha^{-1} yr^{-1}) in Scots pine and Norway spruce stands following stem-only (STEM) and whole tree (WTH) harvesting. From the Swedish Environmental Protection Agency, 2007b).

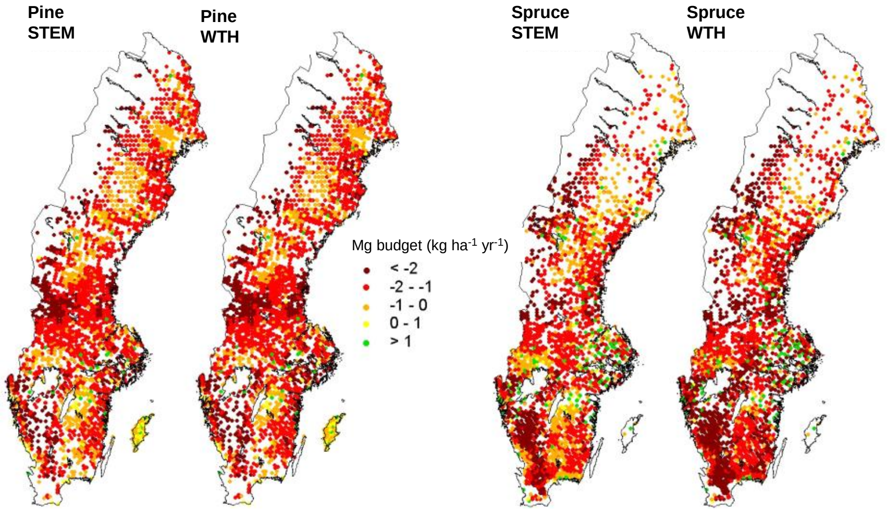


Figure 7. Mass balance budget calculations for Mg^{2+} (kg ha^{-1} yr^{-1}) in Scots pine and Norway spruce stands following stem-only (STEM) and whole tree (WTH) harvesting. From the Swedish Environmental Protection Agency, 2007b).

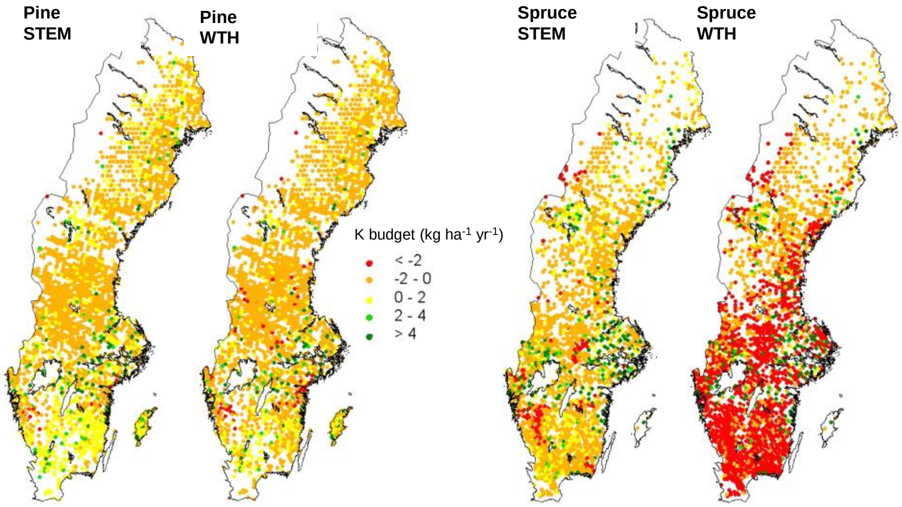


Figure 8. Mass balance budget calculations for K^{+} (kg ha^{-1} yr^{-1}) in Scots pine and Norway spruce stands following stem-only (STEM) and whole tree (WTH) harvesting. From the Swedish Environmental Protection Agency, 2007b).

The difference between tree species, regions and harvest intensities is further illustrated in Figure 9. Regardless of harvest method, stem-only or whole tree harvesting always leads to negative base cation balances. The losses are greater in Norway spruce forests compared with Scots pine, especially when whole-tree harvesting is practiced in southern part of Sweden (-0.26 keq ha^{-1} yr^{-1}).

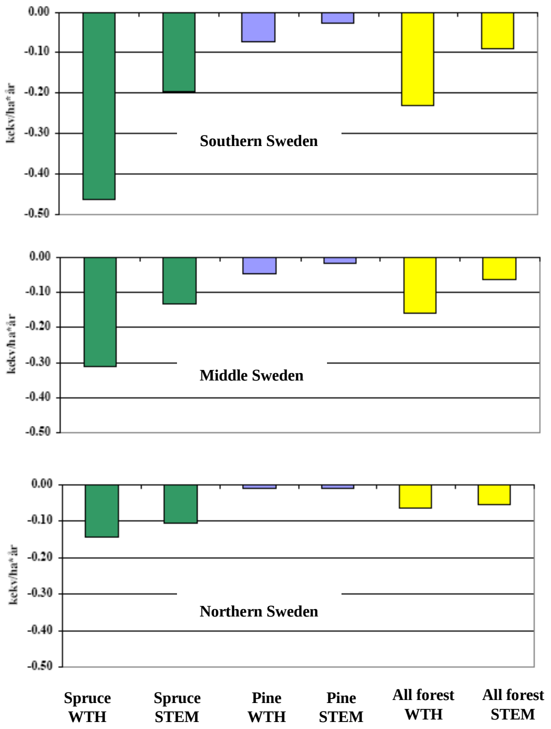


Figure 9. Base cation budgets (including Na) following whole-tree (WTH) or stem-only (STEM) harvesting in southern, middle and northern part of Sweden using potential harvesting volumes from SKA99. The WTH calculations are based on a 75 % removal of the branches and 56 % of the twigs. From the Swedish Environmental Protection Agency (2007b).

Similar results have been found in Finland and Germany using the same input-output approach for Ca^{2+}, Mg^{2+} and K^{+}. Depletion of base cations was simulated in 40-50 % of the area following whole-tree harvesting in southern, central and northeastern Finland (Joki-Heiskala et al. 2003). In comparison, when practicing stem-only harvest, 20 % of the area was affected in the southwestern, southeastern and northeastern Lapland. In northwest Germany, forest soils are already losing more Ca^{2+} and Mg^{2+} via leaching than is compensated for via deposition and weathering. In this system whole-tree harvesting would increase the deficit even further why Ca^{2+}-fertilization could become necessary (Rademacher et al., 2001).

In general, there are few studies describing nutrient removal via harvesting on a regional or national basis. One obvious reason for this is the lack of spatial data necessary for the calculations. Instead mass balances, based on the same methodology as above, are often used on a *local or plot scale*. For instance, Duchesne & Houle (2008) calculated base cation balances following harvest of a balsam fir forest in Quebec, Canada, using six different harvest intensity and weathering rate scenarios. The results showed that K^{+} was lost in five out of the six scenarios. Considering the large accumulation of K^{+} in tree biomass, whole-tree harvest could remove as much as 44% of easily accessible K^{+}. In comparison Ca^{2+} was only lost in two out of six scenarios while Mg^{2+} accumulated in all six scenarios. A similar study was carried out in the Baltic region and Nordic countries using six different harvest scenarios (Raulund-Rasmunssen et al., 2008). Following harvesting and removal of logging residues, the loss of base cations (Ca^{2+}, Mg^{2+} and K^{+}) were greater compared with weathering and deposition inputs on acids soils (pH=4.7-5.0 at 50 cm depth, characterized as sandy soils with poor mineralogy) (op. cit.). In southern Sweden, Sverdrup et al. (2006) predicted that stem-only harvest is sustainable or in excess for Ca^{2+}, Mg^{2+} and K^+ for different tree species on acid forest soils.

Mass balance budget models are robust and transparent allowing different scenarios to be easily applied. The magnitude of change can be used to assess the sustainability of a system and the rate of change (slow or rapid) in relative numbers before negative effects could occur. However, they do not consider feedback mechanisms such as increased weathering or less growth as a result of lower nutrient pools or describes the ecosystem effects. Also, the calculations are limited to the root depth, generally the upper 50 cm of the soil (from the soil surface) and do not link soil chemistry to effects in soil water, groundwater or surface waters. Considering the variability within a catchment, including hydrological pathways, extrapolation of chemical changes within forest soils in recharge areas to other parts of the ecosystem is not as straightforward and very uncertain.

In order to include feedback mechanisms and to provide information about temporally variable parameters, more complex dynamic modelling has been applied in addition to the mass balance models. Dynamic modelling is usually carried out for single sites since large amounts of data are required. Using the Model of Acidification of Ground water In Catchments (MAGIC), Aherne et al. (2008) modelled changes in lake chemistry following different deposition and harvest scenarios (stem-only and whole-tree harvest) in Finland. According to the results, whole-tree harvesting will lead to a long-term re-acidification of soils and lakes (with $ANC<0$) despite further emission reductions. In Canada, Watmough et al. (2003) used the steady-state water chemistry (SSWC) model to predict potential impact of harvest on lake chemistry using three different cutting scenarios (stem without bark, stem-only and whole-tree harvest). The results indicated that harvest “in south-central Ontario will potentially have a dramatic impact on lake chemistry”, especially in the whole-tree harvest scenario. By combining the results from mass balance models with dynamic modelling a more accurate picture of the acidification effects of biomass removal is often assumed. However, the model assumptions and data used greatly affect the simulation results.

3 Important conclusions from previous literature reviews

3.1 Swedish reviews

Pioneering bioenergy research in Sweden begun during the 1960’s, but it was not until the mid-70’s when actual research programs begun in light of the oil crisis 1973 (Anonymous, 1977). Most of the early research paid attention to whether or not harvest of logging residues could be carried out without causing nutrient deficiencies in soils and reduced tree growth, primarily as a result of the N removal. Later research included effects on soil, soil water, ground water and biodiversity. The most important conclusions from previous national literature reviews are given below. The results are based on effects following final cutting unless otherwise stated. The reviews are in Swedish and have not been scientifically published.

Sinclair et al. (1992) evaluated the effects on plant survival and tree growth based on results from Swedish field experiments. Among other things they concluded that:

- Removal of logging residues after final felling will increase plant survival but reduce the short-term (10-25 years depending on site quality) tree growth of Norway spruce (*Picea abies*), but not Scots pine (*Pinus sylvestris*).
- Nutrient content and humus volume will decrease, but the tree growth effect is transitional. Compared with the amount of organic matter produced during a rotation period, the removal during logging is small.
- The pH increase is smaller following whole-tree harvest compared with stem-only.

Based on these results, Sinclair et al. (1992) did not *"discourage whole-tree removal on Swedish mineral soil"*.

Rosén (1991) compiled results from both national and international studies on the ecological consequences of whole-tree harvesting. He concluded that:

- The loss of nutrients (N, Ca, Mg, K and P) following whole-tree harvesting is 1.5-5 times higher compared with stem-only, but relative small in relation to the soil total nutrient pools.
- The loss of Mg^{2+} and K^{+} (but not Ca^{2+}) ions can be compensated for via weathering following stem-only harvesting. Increased removal via whole-tree harvesting will increase the negative balances for Ca^{2+} and also lead to negative balances of Mg^{2+} and K^{+} for many forest sites, especially in the southern part of Sweden. This conclusion was primarily based on nutrient budgets and model calculations mentioned in chapter 2.2 and to a lesser extent on empirical data.
- The risk of a significant decrease of soil organic matter following whole-tree harvesting cannot be verified.
- It is more important to maintain high tree productivity (and litter production) than keep from harvesting logging residues. Thus, low productivity sites are most susceptible to a long-term decrease in humus.
- Mineralization of nutrients (N, P, K and Mg) and nitrification in the forest floor decreases after whole-tree harvesting.
- Whole-tree harvesting will lead to a short-term increase in soil temperature, but will have a negligible effect on soil moisture.
- In terms of soil water nutrient leaching, whole-tree harvesting only have a minor effect. However, leaching below piles of slash may be substantial.
- The survival and growth of seedlings will increase but the nutrient loss following whole-tree harvest may lead to a short-term reduction in forest growth (in Norway spruce but not Scots pine stands)
- The short-term effects are well described but the long-term effects are poorly understood because of the lack of data.

Egnell et al. (1998) have carried out one of the most comprehensive Swedish environmental impact assessments, including both national and international data. They concluded that:

- There is a temporary (10-20 years) increase in pH after clear-cutting, which is 0-0.4 units lower in the humus layer following whole-tree harvest compared with stem-only. There are no observed pH effects in the mineral soil.

- Soil base cation pools and base saturation decreases following whole-tree harvesting while the effect on N pools is not clear, although there are some reports of lower soil pools in the humus layer and mineral soil.
- Clear-cutting increases nutrient leaching. Compared with stem-only harvest, nutrient leaching is lower following whole-tree harvesting. The duration depends on how quickly the new forest stand establishes.
- There is no short-term (1-16 years) effect on soil C pools.
- Whole-tree harvesting increases soil temperatures during the growing season, while the effect on soil moisture is negligible except for the top soil layer.
- Whole-tree harvesting leads to increased plant survival and growth, while the effects on forest growth are negative for Norway spruce but not for Scots pine. There is no difference between high and low productivity sites. Leaving the twigs can counteract the negative effects.
- The long-term effects are unknown.

Based on these results, Egnell et al. (1998) concluded that whole-tree harvesting could be carried out in Sweden if the nutrient loss is compensated for via ash recycling. Also, the needles should be left on site in order to reduce the risk of negative effects, which is in accordance with the recommendations regarding forest fuel extraction issued by the Swedish Forest Agency (2008c).

Over the years, the Swedish Energy Agency has funded research regarding the effects of whole-tree harvesting and ash-recycling mainly in two research programs; sustainable production of bio-fuels from forest land (1997-99) and bio-fuels and the environment (2000-04). The results from these programs and other research were reviewed in 2006 and evaluated with respect to the national environmental quality objectives (Swedish Energy Agency, 2006). Just like the previous reviews, the Swedish Energy Agency (2006) review supported many of the conclusions previously drawn that harvest of logging residues can be utilized if the nutrient loss and acid neutralizing capacity is compensated for via ash-recycling. However, the long-term effects are still largely unknown. In terms of specific effects, it was concluded that:

- The numbers of new experimental field research sites since the last review by Egnell et al. (1998) are few. However, the time-series of already established field sites are longer allowing for more accurate conclusions to be drawn.
- Effects on soil chemistry in terms of increased exchangeable H^+ in the humus layer and decreased amounts of exchangeable Ca^{2+} and base saturation both in the humus layer and mineral soil can still be distinguished 30 years following whole-tree harvesting. These results are entirely based on a single Swedish report by Olsson et al. (2004). Similar effects have been seen following whole-tree harvest in thinnings.
- Effects on soil water at 50 cm depth show increased concentrations of H^+ and decreased concentrations of Ca^{2+} (and sometimes Mg^{2+} and K^+) following whole-tree harvesting in southern Sweden. Again, these conclusions are primarily based on one study ("hyggesåldersförsöket").
- Whole-tree harvesting reduces the concentrations of NO_3^- in the humus layer and sometimes in the soil water, but the effects are not unequivocal.
- The negative effects on tree productivity can largely be avoided if plant establishment is carried out one year earlier than normal.

- Whole-tree harvesting could reduce carbon sequestration.
- Whole-tree harvest without compensatory measures can only be carried out in restricted areas (mainly northern parts of Sweden or low- to medium productivity sites) in order to achieve the environmental quality goal “Natural acidification only”.
- There is a need to understand the discrepancy between theoretical and potentially occurring acidification effects as well as comparing empirical results with model calculations.

The objectives of these reviews have varied depending on the commissioning body and area of interest. As new causes of concern arose, the focus of research changed from tree productivity to soil acidification and eutrophication and more recently to climate change and greenhouse gas emissions. Effects on forest ecosystem are well-studied whereas effects on surface water have only briefly been addressed by interpolating soil water data or by applying experiences from stem-only harvest experiments. Today, the short-term effects in forest stands are well described but the effects on ground- and surface water are still largely lacking. Also, there is a need for long-term studies describing effects during one or more rotation periods.

3.2 International reviews

Internationally, the same environmental problems have been addressed in the scientific literature as in Sweden. For instant, increased nutrient removal via harvesting and risk of decreased soil fertility and tree productivity was addressed early in the UK by Rennie (1955) who warned that *“the nutrient-uptakes of timber producing forest of either coniferous or hardwood species are so large compared with the nutrient-contents of moor soils that further overall soil-degradation and, sooner or later, diminished site-productivity are inevitable”.* Other often cited studies include those of Weetman & Webber (1972), Boyle et al. (1973) and White (1974). In Canada, Kimmins (1977) wrote *“management of the site nutrient capital becomes increasingly important as the proportion of the capital that is removed increases, and as rates of removal approach or exceed natural rates of replacement”.* In a review by Adams et al. (2000), the effects of intensive forestry, increased harvesting, shorter rotations and acid deposition in eastern hardwood forests in the US were assessed with respect to long-term forest productivity. It was found that increased soil acidification and mobilization of base cations after cutting may affect in particular forest soils with low CEC and BS and forests with a large proportion of hickories (*Carya*), basswood (*Tilia americana*), oak (*Quercus*) and yellow-poplar (*Liriodendron tulipifera*) who store relatively large amounts of Ca^{2+} in their boles.

Kreutzweiser et al. (2008) reviewed logging impacts on the biogeochemistry of C, N, P and base cations in North America boreal forest soils as well as nutrient exports to streams and lakes. Stem-only or whole-tree harvest was the dominant clear-cutting methods. Some studies also included other silvicultural practices (e.g. site preparation and regeneration). It is also worth noticing that in this part of the world, fire is used as a principle method to aid regeneration. The studies included different kind of species (coniferous and broad-leaved) as well as different soils (podzols, cambisols etc.) and site quality classes (rich and poor sites). In general, it was found that soil effects are limited and site-specific. However, measurable effects in receiving waters frequently occur. In terms of specific treatment, the following effects were observed:

- The availability and mobility of base cations (Ca^{2+}, Mg^{2+} and K^+) are likely to increase following harvest resulting in increased export to receiving waters.

- However, changes in soil content (both total and exchangeable) were not observed in 63-76 % of the cases for the forest floor and 71-100 % for the mineral soil.
- Out of the three cations, Ca^{2+} is most likely to be affected (59-69 % showed forest floor changes, both total and exchangeable). None of the studies showed any changes in the mineral soil. In case of a change, 66 % showed decreased concentration or storage while 33 % showed increases in the forest floor. Some studies suggested that whole-tree harvesting had greater effects compared with stem-only.
- The effect on soil C is limited with minimal impact on site productivity. Out of the 14 studies that were reviewed, 76 % showed neither change in concentration or in soil C pools (considering both the forest floor and mineral soil). In the remaining studies, both losses (attributed to decreased litter inputs) and gains were noted. In case of a change, the impact of stem-only harvest was less compared with whole-tree harvesting.
- Disrupting the carbon cycle usually leads to increased DOC-concentration both in soil water and surface waters attributed to leaching from slash, increased decomposition and a rise in groundwater tables.
- The effect on total soil N (pools and concentrations) is variable and difficult to predict and highly site-specific. With respect to the 13 studies reviewed, 73 % of the soils showed no change. Compared with the mineral soil, changes (both increases and decreases) are more likely to occur in the forest floor.
- Concentration increases in ammonium (NH_4^+) and nitrate (NO_3^-) are frequently noted usually attributed to decreased plant uptake and increased mineralization and nitrification. Increases have been noted both in soil water and streams.

While Kreutzweiser et al. (2008) offers useful insight into changes likely to occur following harvesting, the review does not quantify the changes in pools or concentrations. Also, the studies included in the review only report short-term changes, in most cases the first couple of years following cutting.

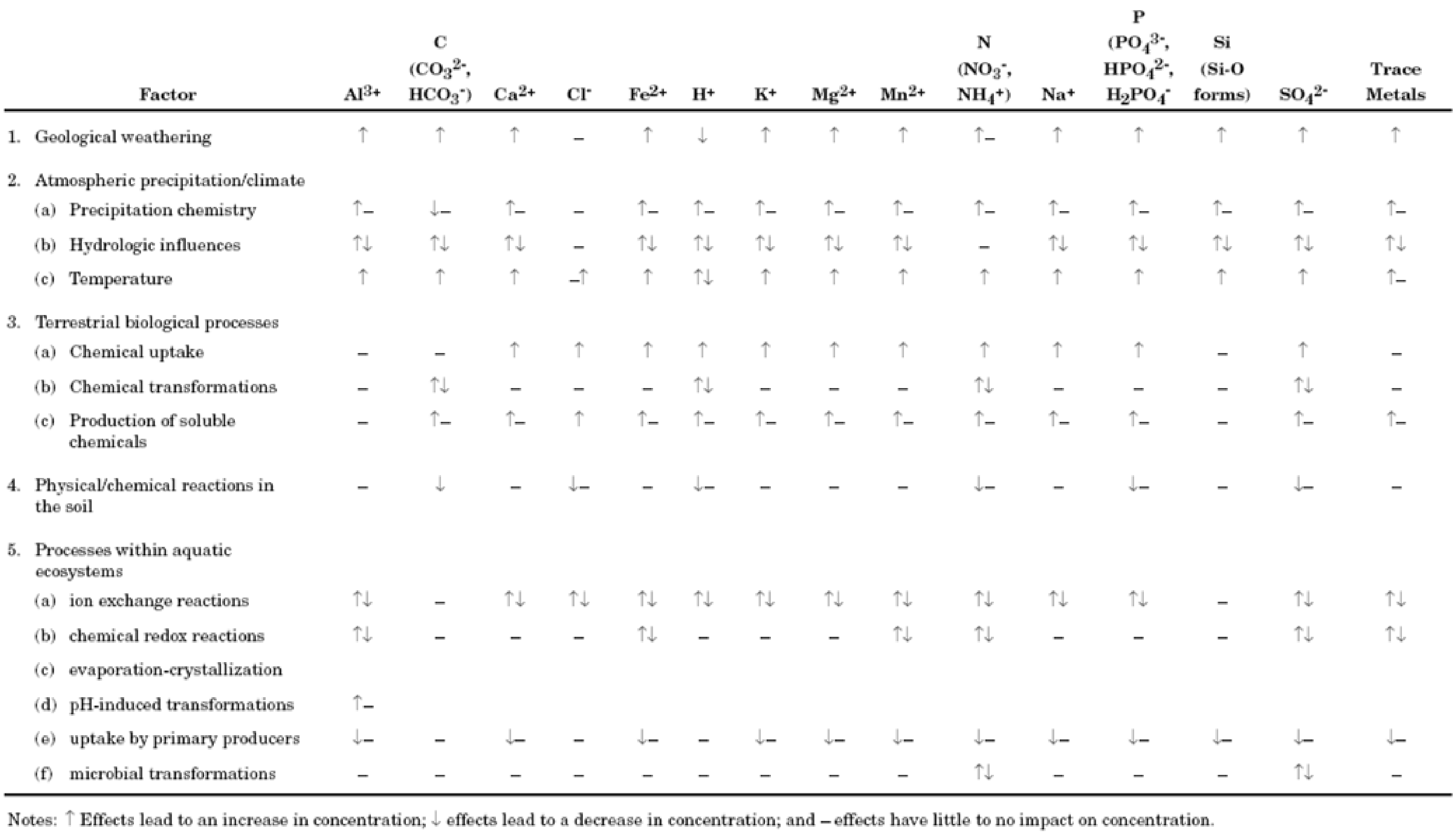
Feller (2005) summarized changes in stream water chemistry in the western US and Canada by considering harvesting effects on the five most important factors controlling stream water quality (1=weathering, 2=atmospheric deposition, 3=biological processes, 4=soil reactions and 5=within stream processes, Table 3). Catchment characteristics and hydrological flow paths are thus not considered in terms of stream water quality. Also, harvest intensity (stem-only or whole-tree) is not discussed in the review nor does it separate between short- and long-term effects. Also, site specific differences, such as tree species, soil type etc., are not specified. Still, the summary provides a good overview of cutting effects per se after final felling.

According to Feller (2005) harvesting leads to increased soil weathering rates thereby releasing inorganic ions, which can be leached to surrounding waters (Table 3, factor 1). Timber harvesting usually leads to increased acidity both in soil water and stream water in the order of magnitude of 0.2-0.5 pH-units as a result of nitrification and release of organic acids. This often results in increased export of base cations, aluminium and trace metals to receiving waters (Table 3, factor 2a). Initial increases in annual runoff are frequently reported followed by a decrease to pre-harvest levels or even below (Table 3, factor 2b. The effect on snow-pack is unclear. Removing the tree canopy leads to increased summer temperatures both on the soil surface as well as in the stream (Table 3, factor 2c).

In terms of terrestrial biological processes, Feller (2005) showed that lowered plant uptake could lead to increased concentration of inorganic nutrients in the stream (Table 3, factor 3a). Also, any processes affecting mycorrhizae negatively could lead to increases in NO_3^-. Nitrogen increases could also be the result of N-fixation whereas a decrease could be due to denitrification. Decreased activity of S-oxidizing bacteria or increased S-adsorption in soil is often the case resulting in decreased SO_4^{2-}-concentrations (Table 3, factor 3b). In addition, lower dry deposition, dilution, formation of sulfate-containing minerals via precipitation, increased decomposition and higher groundwater table leading to anoxic soil conditions contributes to lower SO_4^{2-}-concentrations. The production of soluble substances via increased decomposition of residual material left on ground would increase following harvesting (Table 3, factor 3c).

Effects on physical and chemical reactions in the soil (Table 3, factor 4) include increased anion exchange capacity of the soil (e.g. through increased production of protons via nitrification), which could lower the concentration of anions (e.g. SO_4^{2-}) in the stream and increased adsorption of P to Fe- and Al-hydroxides in the soil. Finally, effects on in-stream processes following harvest have not been reported but could lead to increased or decreased concentrations (Table 3, factor 5a-f). Feller (2005) concluded that at least four factors influence stream water concentration of inorganic ions and predicting stream-water response following harvesting is thus a challenge, especially considering the various catchment characteristics and harvesting methods.

Table 1. Initial stream water response following final cutting on the five most important factors determining aquatic chemistry. From Feller (2005).



Factor	Al^{3+}	C (CO_3^{2-}, HCO_3^-)	Ca^{2+}	Cl^-	Fe^{2+}	H^+	K^+	Mg^{2+}	Mn^{2+}	N (NO_3^-, NH_4^+)	Na^+	P (PO_4^{3-}, HPO_4^{2-}, $H_2PO_4^-$)	Si (Si-O forms)	SO_4^{2-}	Trace Metals
1. Geological weathering	↑	↑	↑	–	↑	↓	↑	↑	↑	↑–	↑	↑	↑	↑	↑
2. Atmospheric precipitation/climate															
(a) Precipitation chemistry	↑–	↓–	↑–	–	↑–	↑–	↑–	↑–	↑–	↑–	↑–	↑–	↑–	↑–	↑–
(b) Hydrologic influences	↑↓	↑↓	↑↓	–	↑↓	↑↓	↑↓	↑↓	↑↓	–	↑↓	↑↓	↑↓	↑↓	↑↓
(c) Temperature	↑	↑	↑	–↑	↑	↑↓	↑	↑	↑	↑	↑	↑	↑	↑	↑–
3. Terrestrial biological processes															
(a) Chemical uptake	–	–	↑	↑	↑	↑	↑	↑	↑	↑	↑	↑	–	↑	–
(b) Chemical transformations	–	↑↓	–	–	–	↑↓	–	–	–	↑↓	–	–	–	↑↓	–
(c) Production of soluble chemicals	–	↑–	↑–	↑	↑–	↑–	↑–	↑–	↑–	↑–	↑–	↑–	–	↑–	↑–
4. Physical/chemical reactions in the soil	–	↓	–	↓–	–	↓–	–	–	–	↓–	–	↓–	–	↓–	–
5. Processes within aquatic ecosystems															
(a) ion exchange reactions	↑↓	–	↑↓	↑↓	↑↓	↑↓	↑↓	↑↓	↑↓	↑↓	↑↓	↑↓	–	↑↓	↑↓
(b) chemical redox reactions	↑↓	–	–	–	↑↓	–	–	–	↑↓	↑↓	–	–	–	↑↓	↑↓
(c) evaporation-crystallization															
(d) pH-induced transformations	↑–														
(e) uptake by primary producers	↓–	–	↓–	–	↓–	–	↓–	↓–	↓–	↓–	↓–	↓–	↓–	↓–	↓–
(f) microbial transformations	–	–	–	–	–	–	–	–	–	↑↓	–	–	–	↑↓	–

Notes: ↑ Effects lead to an increase in concentration; ↓ effects lead to a decrease in concentration; and – effects have little to no impact on concentration.

Martin et al. (1984) reported short-term (2-3 years) changes in stream water chemistry between clear-cuts and unharvested controls in New England, USA following final felling. They used a synoptic approach covering a wide range of forest types and soils but did not discuss harvest intensities. In general, only minor changes in major anion and cation concentrations occurred as a result of harvesting except for one area in New Hampshire where NO_3^- and Ca^{2+} concentrations were 4 respectively 2 times higher compared with a control forest. The largest stream water differences were found between areas that were completely clear-cut relative to uncut controls compared with only partially clear-cut areas. In another synoptic study by Neal et al. (1998b), the effects of conifer harvesting on stream water were examined. Using data from 67 sites (51 clear-cuts and 16 mature stands) across Wales they showed that the response following harvesting was very variable and site specific (catchment characteristics and water flow paths), but local short-term effects in terms of acidification (increased Al and decreased ANC) and increased NO_3^- concentrations can occur. However, due to the variable response, they concluded that the idea of "*there being 'representative catchments' with predictable runoff chemistry for particular combinations of geology, soil and land use may at best prove elusive and may not even exist*".

Many reviews have focused on NO_3^- leaching to groundwater, stream water and the coastal marine environment due to an increasing concern of the impact of forest operations (in particular clear-cutting) on drinking water quality and eutrophication (e.g. Vitousek et al., 1979; Vitousek & Melillo, 1979; Binkley & Brown, 1993; Brown & Binkley, 1994; Gundersen et al., 2006). In addition, production of NO_3^- via nitrification and subsequent leaching leads to temporary acidification effects in soils and water. For example, during nitrification, hydrogen ions (H^+) are produced which in turn lower soil pH. Via aluminium buffering some of the released H^+ are neutralized, but in return, inorganic aluminium is produced. An increase in soil water anion concentration will be accompanied by an equal amount of base cations according to the electron neutrality principle, and vice versa (chapter 4). Thus, an increase in soil water anion concentration may lead to increased leaching to groundwater and surface waters, reinforced by increased groundwater tables and stream runoff as evapotranspiration temporally ceases.

Vitousek et al. (1979) described the mechanisms responsible for increased NO_3^- losses following disturbances such as clear-cutting for 19 sites in the US. They identified nine different processes in the N cycle, which could help to delay or prevent a stream water response (Figure 10). Accumulation of NH_4^+ in the soil could be prevented or delayed via N immobilization, NH_4^+ fixation, ammonia volatilization or plant uptake (1). A lag in nitrification and denitrification could delay or prevent an accumulation of NO_3^- (2) while factors such as lack of water, NO_3^- sorption and denitrification in deep soil horizons could affect the mobility of NO_3^- (3). The relative importance of these processes may vary but the net effect is often insufficient to prevent increased stream water concentrations of NO_3^-, perhaps with the exception of plant uptake as the stand grows older.

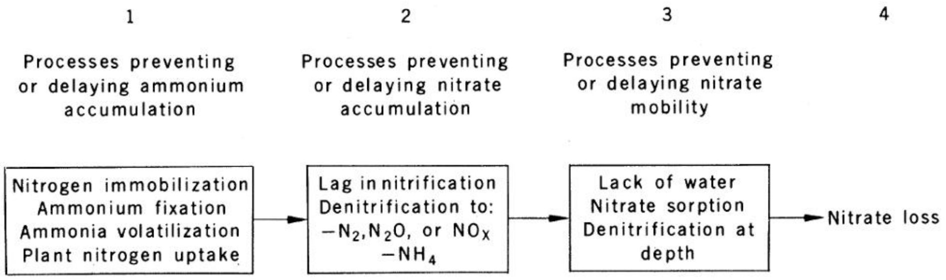


Figure 10. Processes delaying or preventing stream water NO_3^- response following clear-cutting. From Vitousek et al. (1979).

Out of these nine processes, N immobilization, plant uptake, lags in nitrification and a lack of water for mobility are the most important (Vitousek et al., 1979). From Figure 10 it should also be clear that even though the abiotic conditions (increased light, higher temperatures etc.) may favour mineralization and nitrification, but the stream water signal might still be absent due to the processes described. Based on the 19 study sites, Vitousek et al. (1979) could identify the following response types (with a dominance of type 1 and 2) ultimately resulting in NO_3^- losses of >100µeq per litre:

- no change in NH_4- or NO_3-concentration (type 1)
- significant increase in NH_4 in the soil but not NO_3 in the soil or soil solution (type 2)
- increase in NO_3 in the soil but not the soil water (type 3)

In another study by Vitousek & Melillo (1979) increased NO_3^- concentrations varying between 0.02 and 35 mg per liter (differences between clear-cut and reference) were observed during the first year after cutting for a range of states, quality sites and disturbances, mainly in the US. Elevated concentrations of P and N in streams as a result of different management practices, including timber harvesting in the US (and in some cases Canada) were reviewed by Brown & Binkley (1994) who stated that *"although forest practices may elevate the concentration of many chemicals in stream water, only the concentrations of phosphate and nitrate are of significant concern in forestry",* clearly demonstrating the environmental concern of that time (eutrophication and exceedance of drinking water standards). Based on the results from 18 studies, short-term increases (<0.5 mg per litre or 36 µeq per litre) of NO_3^- were observed in the 31 studies included with one important exception; results from the Hubbard Brook Experimental Forest where herbicides were used to retard revegetation. Also, elevated concentrations of NO_3^- were found in some areas including alder-rich sites and areas with high deposition of N (present and/or historically). The effects are normally short-termed and decreases with time as the new forest stands grow older.

Following the work of Vitousek & Melillo (1979), Gundersen et al. (2006) reviewed the leaching of NO_3^- from temperate forests (both coniferous and deciduous) as a result of forest management and air pollution. Based on 29 studies in Europe and the US, it was found that NO_3^- leaching in ground water, seepage water and stream water usually increase within 2-3 years followed by a decline after 3-5 years. Mechanisms, explaining these results include reduced uptake, accelerated decomposition and revegetation rates. The highest concentrations (5 mg per litre as a mean over the region) were found in areas receiving the highest atmospheric N-input (Figure 11). However, compared with pre-harvest concentrations, N-saturated areas did not show higher concentrations compared with non-saturated areas. In contrary to what is often believed, Gundersen et al. (2006) did not see a clear difference in harvest intensity (stem-only vs. whole-tree) in terms of NO_3^- leaching. Instead, the increase in NO_3^- leaching following clear-cutting is a result of pre-existing conditions.

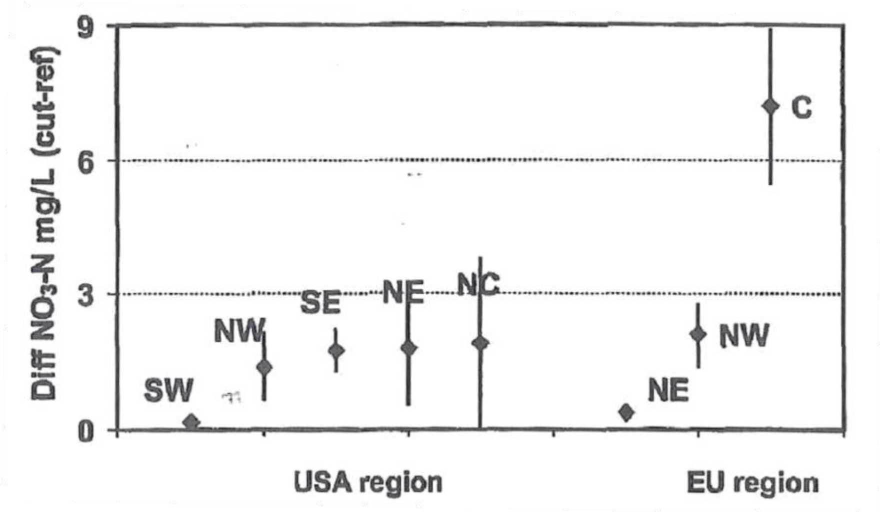


Figure 11. Nitrogen leaching in Europe and the US following clear-cutting expressed as the difference between a clear-cut and a reference area within 1-2 years after harvesting. SW=southwest, NW=northwest, SE=southeast, NE=northeast, NC=north central, C=central. From Gundersen et al. (2006).

Harvest also has an impact on the carbon cycling in forest soils. Understanding short- and long-term changes in soil C (and N) are important not only in terms of forest productivity and forest ecosystem services (carbon sequestration), but also as a reduction in organic matter content and cation exchange capacity, it may decrease the buffering capacity of the soil and subsequently increase the sensitivity to acid inputs. Hope et al. (1994) reviewed processes regulating fluxes of organic carbon and how they are affected by land use. Based on studies from New Zeeland, Great Britain and the US, it was found that the carbon effects of harvest are much more variable compared with other nutrients. Hope et al. (1994) concluded that the differences in the export of DOC sometimes depend on hydrology, season and antecedent weather conditions and not the actual cutting. On the other hand, cutting per se may lead to changes in hydrology, which can increase the export via increased runoff or change the hydrological flow paths to more/less organic horizons leading to dilution/concentration. In addition, changing hydrological flow paths may alter oxidation/reduction processes in soils leading to accelerated/reduced decomposition rates. In a more recent review by Kreutzweiser et al. (2008), it was found that the export of DOC to streams usually increase after logging leading to increased lake concentrations.

Johnson & Curtis (2001) performed a meta-analysis on the literature describing forest management effects on soil N and C. In general, the effect was small but significant differences occurred between species and harvest intensity; following stem-only harvesting, soil C and N increased (only in coniferous forest) with 18 % in the A-horizon compared with whole-tree harvesting that caused decreases (-6 %). The O-horizon was not included in the analysis. In addition, Sanchez et al. (2006) used data from the Long-Term Soil Productivity (LTSP) study to evaluate the effect of organic matter removal (bole, whole-tree and whole-tree plus forest floor) on soil N and C pools and compaction after treatment for 19 sites across the US and Canada. Five years after treatment, no effects on the pools could be seen, not even in the most extreme treatment (whole-tree plus forest floor). Effects on survival and growth were examined by Fleming et al. (2006) in relation to organic matter removal, soil compaction and vegetation control using the same LSTP sites. In general, whole-tree harvesting usually led to a small increase in seedling survival, which was in part counteracted by decreased growth.

Following clear-cutting, evapotranspiration temporally ceases which often results in higher water tables and increased stream runoff. In turn, these changes can cause dilution and greater base cation leaching losses from the system. Also, decomposition and redox processes might be affected. Runoff changes following clear-cutting have been well documented since the 1960’s and most often described as annual changes and within-year variation (seasons or months). In addition, the characteristics of storm flow events have been described (e.g. Guillemette et al. 2005). For example, in an early review by Hibbert (1967) annual increases between 34-450 mm were reported. In general, runoff decreased with the reestablishment of the new forest but in one case increases in runoff was still evident after 20 years. It was also found that stream flow increase was proportional to the reduction in forest cover. In the upper region (450 mm per year), expected increase in runoff was 4.5 mm per 1% reduction in forest cover. In another review by Bosch & Hewlett (1982) annual increases in runoff were found in all but one case (a burned Eucalyptus stand in Australia). In general, every 10% reduction in vegetation cover resulted in a 40 mm increase in annual runoff in pine and eucalyptus forests compared with 25 and 10 mm in deciduous hardwood and shrubs, respectively. A maximum of 660 mm per year was found for one site situated in North Carolina, USA. It was also shown that a reduction in vegetation cover below 15-20% would not lead to a stream flow response. During the first years following clear-cutting, the annual runoff also increased linearly with precipitation (Figure 12). Bosch & Hewlett (1982) concluded *“Decreases in water yield following afforestation seem to be proportional to the growth rate of the stand while gains in water yield after clear-felling diminish in proportion to the rate of recovery of the vegetation”.* The reviews of Hibbert (1967) and Bosch & Hewlett (1982) do not include any European studies.

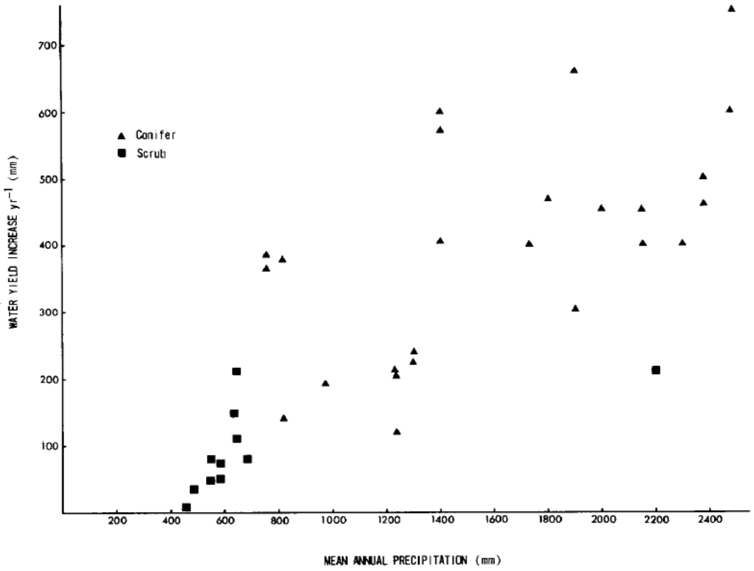


Figure 12. During the first year following clear-cutting, the increase in annual runoff is a function of the annual precipitation. From Bosch & Hewlett (1982).

In a more recent review by Brown et al. (2005) it was concluded that increases in runoff are usually observed only during the first five years. However, when the long-term changes were compared it was found that it might take decades before returning to pre-harvest conditions. Studying the within-year variation, it was found that the largest increase occurs during wet periods and vice versa. In areas with warm summers and cold winters with snow, the largest increase occurred during the summer months (i.e. during the vegetation period at base flow).

The reviews cited here are limited to describing short-term effects (often <5 years) and seldom separate differences between whole-tree and stem-only harvesting, species and site quality. Also, studies from the Nordic countries are seldom included. The main findings of these reviews show that the stream water response following final cutting varies across sites, regions and countries as a result of different catchment characteristics, species, and atmospheric deposition. The response also varies depending on harvest intensity (stem-only, whole-tree and partially cut), harvested area, use of buffer strips and silvicultural practices (site preparation, ditching etc.). Most reviews have focused on stream water response and seldom summarize soil chemical changes (except for soil N and C). Furthermore, most reviews have focused on the US and Canada and for that reason often use the same references. Concentration increases, decreases or lack of response cannot be explained by one single factor or mechanisms. In general, a number of mechanisms either reinforce or counterbalance the response. Given the large variability, describing the effects following final felling cannot be done using a universal model.

4 Some fundamentals regarding acidification

4.1 Acidification of soils

The acidity of soils depends on the relative amount of exchangeable cations (Ca^{2+}, Mg^{2+}, Na^{+} and K^{+}), H^{+} and Al^{n+}-species. In alkaline and neutral soils the amount of exchangeable cations dominates whereas H^{+} and Al^{n+}-species dominate in acid organic respectively mineral soils (Reuss & Johnson, 1986). Processes changing the total amount of cation exchange sites (e.g. via increased accumulation of organic matter or clay formation) or the concentration of base cations (e.g. via leaching or harvest removal) would alter soil acidity by changing the cations exchange capacity (CEC) and the relation between exchangeable acidity (EA) and base cations, affecting the base saturation (BS). Increased weathering or addition of lime and wood ash would make the soil more basic.

For example, atmospheric deposition of S and N introduce additional protons into the forest ecosystem, which can either participate in soil exchange processes or in weathering reactions whereby aluminium ions are released. Aluminium, in turn, replaces base cations on the exchange complex, which are then leached from the system accompanied by anions SO_4^-, NO_3^-, Cl^- and organic anions according to the electron neutrality principle. The resulting change is an increase in EA (H^{+}+Al^{n+}) and a decrease in BS (Ca^{2+}, Mg^{2+}, Na^{+} and K^{+}). Biomass removal in forestry, either in pre-commercial thinning, thinning or final cutting, also removes base cations from the system, which could lead to soil acidification. Obviously the response varies depending on factors such as the amount of nutrients removed and the pre-existing soil properties.

Useful parameters for describing forest soil acidity include pH, base saturation, exchangeable aluminium, cation exchange capacity, and exchangeable acidity. Currently, the Swedish Environmental Agency (2007a) uses a combination of the first three parameters to assess the acidifications status of soils.

4.2 Acidification of soil water, groundwater, stream water and lakes

pH is a direct way of measuring the acidity of waters whereas alkalinity and acid neutralizing capacity (ANC) measures the ability to neutralize acids (or the resilience against acidification) by considering different buffering systems. There are a number of soil reactions, which can lower or increase the pH (Table 2). For instance, plant respiration, dissociation of organic acids, vegetation uptake of cations, sulfide oxidation, and nitrification are examples of natural acidifying processes. In addition to these processes, anthropogenic acid deposition has caused soil and surface water acidification (chapter 2.1). Weathering of silicates, carbonates, aluminium hydroxides and iron hydroxides, cation exchange processes, SO_4^{2-} adsorption and vegetation uptake of anions are examples of buffering mechanisms preventing pH decreases.

The H^+-concentration is affected by almost every biogeochemical reaction in the soil making it one of the most complex cycles to study (de Vries & Breeuwsma, 1987), which in turn makes it even more difficult to predict a certain response following harvest. However, some patterns are generally agreed upon. Trees naturally acidify the soil via nutrient uptake and via bacterial nitrification of NH_4^+ to NO_3^-. For example, Hallbäcken & Tamm (1986) found an age-related decline in soil water pH by almost one unit in the A-horizon of a spruce stand during a period of 55 years. Buffering processes, which neutralize the build-up acidity, include decomposition and weathering. If undisturbed, these processes will compensate for each other in the long-term. However, following clear-cutting in final felling the stems and sometimes the finer parts of the tree (i.e. logging residues) are removed, which decreases the ability to neutralize H^+ via decomposition.

Table 2. Sources and sinks of H+ in soils. From de Vries & Breeuwsma (1987).

Sources of H^+	**Sinks of H^+**
Uptake of cations	Uptake of anions
Mineralization of anions	Mineralization of cations
Oxidation reactions	Reduction reactions
Dissociation of weak acids (CO2, organic acids)	Association of weak acids (CO2, organic acids)
Weathering, desorption of anions, precipitation, adsorption of cations	Weathering, desorption of cations, precipitation, adsorption of anions

Alkalinity takes into account one of the most important buffer systems in nature, namely the carbonate system whereby carbon dioxide (CO_2) dissolves in water (H_20) and forms carbonic acid (H_2CO_3) which, depending on pH, dissolves into bicarbonate (HCO_3^-) and carbonate (CO_3^{2-}) releasing H^+:

$$CO_2 + H_2 0 \leftrightarrow H_2CO_3 \leftrightarrow H^+ + HCO_3^- \leftrightarrow 2H^+ + CO_3^{2-} \qquad \textit{(Equation 1)}$$

Alkalinity is defined as the sum of bicarbonate, carbonate and hydroxyl ion (OH^-) minus the H^+ concentration (Stumm & Morgan, 1981):

$$Alkalinity = HCO_3^- + CO_3^{2-} + OH^- - H^+ \qquad \textit{(Equation }\mathbf{2}\textit{)}$$

ANC includes buffer systems other than the carbonate system and can be defined both operationally and conceptually (op. cit.). Operationally, ANC is defined as the equivalent sum of all the bases, which can be titrated with a strong acid to a pre-determined equivalence point (i.e. how much acid must be added to change the pH from one value to a pre-determined reference value). Therefore, aquatic systems with a high concentration of bases have higher buffer capacity (ANC) against acidification compared with base-poor systems. Or, expressed in another way, the addition of an acid may cause a small or large pH-change depending on the buffer capacity (ANC) of a particular system. Conceptually, ANC measures the concentration of all the ions containing protons in deficiency minus the concentration of all the ions containing protons in excess (i.e. the net deficiency of protons).

ANC is derived using a charge balance model based on the electron neutrality principle. Hence, the number of negative charges must be balanced by an equal amount of positive charges per unit volume (op. cit.). For example, in an aqueous solution containing a strong base (NaOH) and a strong acid (HA), the base will react with the acid according to the reaction:

$$NaOH + HA \rightarrow Na^+ + OH^- + H^+ + A^- \qquad \textit{(Equation 3)}$$

Following the electron neutrality principle this reaction rearranges to:

$$Na^+ + H^+ = A^- + OH^- \qquad \textit{(Equation 4)}$$

or:

$$\sum nC^{n+} = \sum mC^{m-} \qquad \textit{(Equation 5)}$$

Thus, the difference between strong acids and strong bases (pH-independent definition) defines ANC:

$$ANC = \sum nC^{n+} - \sum mC^{m-} \qquad \textit{(Equation 6)}$$

ANC is often expressed in equivalents per litre rather than moles per litre because of the presence of mono-, di- and trivalent ions in the solution. Alternatively, ANC can also be defined as the difference between weak acids and weak bases (pH-dependent definition). However, traditionally, the pH-independent definition is used since such parameters are much easier to correctly analyze.

The dominant cations and anions in natural waters (Warfvinge, 1999) are listed in Table 3. They are grouped according to their autoprotolysis in water into weak bases and weak acids respective strong bases and strong acids. Weak acids only partially dissociate in water and do not release all of the H^+, while weak bases do not completely protonate. Both dissociation and protonation are pH-dependent.

Table 3. The chemical composition of natural waters consists of both strong and weak acids and bases. R denotes organic acids. From Warfvinge (1999).

	Base	Cation	Acid	Anion
STRONG	$Ca(OH)_2$	Ca^{2+}	H_2SO_4	SO_4^{2-}
	$Mg(OH)_2$	Mg^{2+}	HCl	Cl^-
	KOH	K^+	HNO_3	NO_3^-
	$NaOH$	Na^+		
	NH_3	NH_4^+		
WEAK	H_2O	H^+	H_2O	OH^-
	$Al(OH)_3$	Al^{3+}, $Al(OH)^{2+}$, $Al(OH)_2^+$	H_2CO_3	HCO_3^-
				CO_3^{2-}
	R^-	H^+	$R\text{-}H$	R^-

By applying the pH-independent definition (equation 6), ANC (moles per litre) can be defined as:

$$ANC = 2Ca^{2+} + 2Mg^{2+} + K^+ + Na^+ + NH_4^+ - 2SO_4^{2-} - Cl^- - NO_3^- \qquad \textit{(Equation 7)}$$

Or, using the pH-dependent definition;

$$ANC = OH^- + HCO_3^- + 2CO_3^{2-} + R^- - H^+ - (Al - species) \qquad \textit{(Equation 8)}$$

The charge balance for the total system is:

$$\begin{aligned} &2Ca^{2+} + 2Mg^{2+} + K^+ + Na^+ + NH_4^+ + H^+ + (Al - species) \\ &= 2SO_4^{2-} + Cl^- + NO_3^- + OH^- + HCO_3^- + 2CO_3^{2-} + R^- \end{aligned} \qquad \textit{(Equation 9)}$$

which rearranges to:

$$\begin{aligned} &2Ca^{2+} + 2Mg^{2+} + K^+ + Na^+ + NH_4^+ - 2SO_4^{2-} - Cl^- - NO_3^- \\ &= H^+ + (Al - species) - OH^- - HCO_3^- - 2CO_3^{2-} - R^- \end{aligned} \qquad \textit{(Equation 10)}$$

Thus, electron neutrality of the total aquatic system can only be achieved if the concentration of strong acids is balanced by the concentration of weak bases and the concentration of strong bases by weak acids. As can be seen from equation 10, and based on the previous discussion, it is more convenient to calculate ANC from the pH-independent using the easily measured and conservative major anions and cations (Equation 7).

Organic acidity (R^-) can be simulated as a function of total organic carbon and alkalinity (Köhler et al. 2000). Thus, there will be a relationship between DOC, pH and ANC. The relationship between pH and ANC is illustrated in Figure 13 at different concentrations of CO_2 (P_{CO2}) and DOC. As can be seen, for a given value of ANC, there may be a large variation in pH depending on P_{CO2} and DOC. For example, at ANC=50 $mmol_c$ per m^3 the pH-value is approximately 6.5 at $pCO_2=2*3.5*10^{-4}$ atm compared with 5.5 at $P_{CO2}=20*3.5*10^{-4}$ atm. Also, at ANC=50 $mmol_c$ per m^3 the pH is approximately 6.5 in waters with a DOC-concentration of 2 mg per litre compared with circa 4.5 at DOC-concentrations of 20 mg per litre. Thus, increases in P_{CO2} and DOC-concentration lower the pH-value (Warfvinge, 1999). This is important to keep in mind since the concentration of DOC varies with soil depth as well as between soils and streams. So does P_{CO2} in soil water, ground water and open waters. Biological processes in the soil (decomposition, respiration etc.) increases P_{CO2} which degasses when the soil solution becomes surface water. The resulting pH-change can be more than 1.5 units.

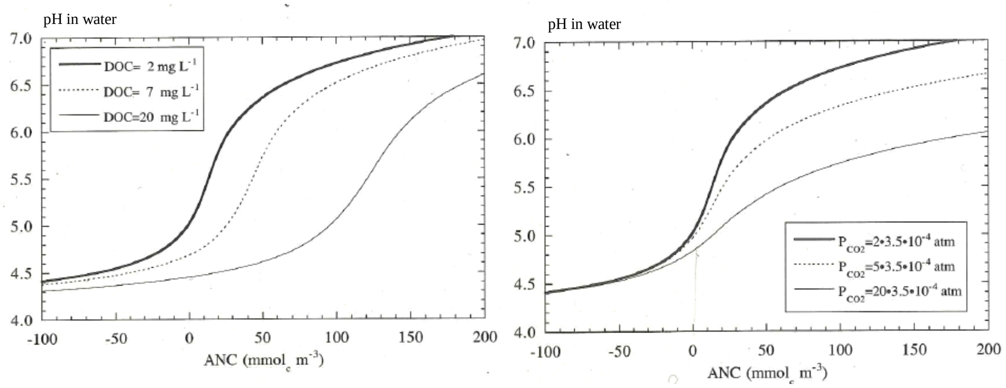


Figure 13. Titration curves describing the relationship between pH and ANC during different partial pressures of carbon dioxide (right) and concentrations of DOC (left) according to the pH-dependent definition of ANC (equation 7). From Warfvinge (1999).

A loss of base cations via harvesting, especially during whole-tree harvesting, has a potential to cause acidification by decreased base cation leaching thereby decreasing the ANC (equation 7), which in turn lowers the pH (Figure 13). Decreases in pH increase the solubility of aluminium, especially below pH≤5.5 (Figure 14).

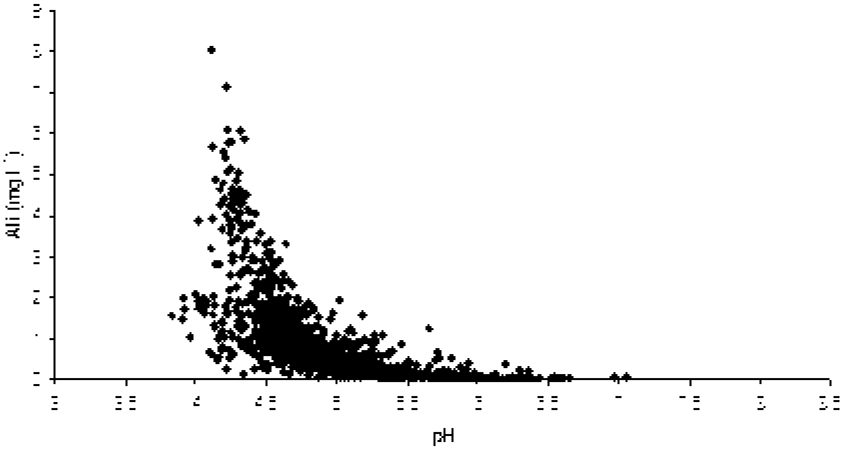


Figure 14. The relationship between pH and aluminium based on soil water data from 68 stations from the Swedish Throughfall Monitoring Network. The measurements are carried out 3 times a year; during spring, summer and autumn at a soil depth of 50 cm. From IVL Swedish Environmental Institute.

5 Catchment characteristics and hydrology

Just like soils, stream characteristics differ depending on the mosaic of biogeochemical environments and groundwater flow paths within a catchment. Understanding the temporal and spatial variation in stream water chemistry at a certain point is a challenge as it summarizes all of the physical, chemical and biological processes that take place upstream in the catchment as well as the reactions taking place within the stream.

Before discussing clear-cutting effects it is important to understand how soil water, groundwater and stream water concentrations and fluxes vary within a catchment depending on the distribution of recharge-, discharge- and peat land areas (catchment characteristics), groundwater flow paths (transport of solutes) and residence time (contact time with solids). For instance, the chemical composition of surface water may vary depending on flow situation (base flow or high flow), hydraulic connectivity between subareas within the catchment, presence of riparian zones etc. Changes in stream water chemistry will follow when either the hydrological flow paths or biogeochemical processes in the catchment are affected (Kreutzweiser et al., 2008).

An example of how stream water chemistry changes during a storm flow event is given in Figure 15. Depending on flow paths and catchment characteristics the concentration of DOC in the stream increases (green line, forest dominated stream water) or decreases (red line, mire dominated stream water) during increased runoff (blue line) as the groundwater moves through DOC-rich or DOC-poor soil layers.

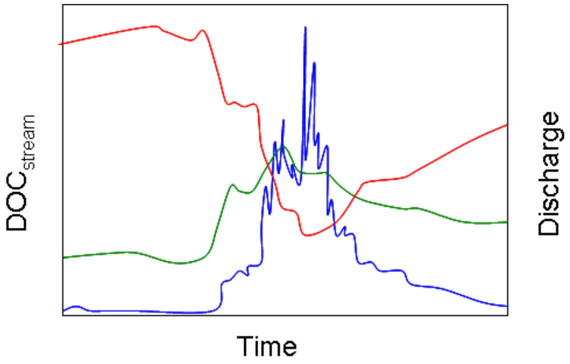


Figure 15. During a storm flow event the groundwater table increases which generates increased runoff (blue line). If the groundwater moves through DOC-poor soil layers, the DOC concentration in the stream will decrease (red line) On the other hand, if the groundwater moves through DOC-rich soil layers the concentration will increase (green line). Redrawn from Kevin Bishop, SLU.

5.1 Stream flow generation

Water enters the catchment as precipitation (rain and/or snow) and is to a large extent stored in the unsaturated zone of the soil as soil water. Through gravitation, the water moves vertically deeper down the soil towards the saturated groundwater zone. As a result, the groundwater table moves closer to the soil surface and the groundwater flows laterally to lower lying areas (Lundin, 1990).

The input and output water balance of a catchment is described by equation 11:

$$P = E + Q + \Delta S \qquad \textbf{(}\textit{\textbf{Equation 11)}}$$

where input of water via precipitation (P) is the sum of evapotranspiration (E), runoff (Q) and changes in soil water storage (S). From this it can be seen that runoff and soil storage will increase if evapotranspiration decreases given that precipitation is constant. Also, increases in runoff must be preceded by an increase in groundwater storage since stream water is dominated by groundwater. At an annual time scale, evapotranspiration is the main process leading to changes in soil water storage in the catchment as a result of vegetation changes (Zhang et al., 2001).

In the Nordic countries where the climate is characterized by warm summers and cold winters most of the precipitation falls as snow during winter and as rain during summer. Hence, runoff in the winter is low followed by a large increase during the snowmelt (spring flood) period (Lundin, 1999). For example, as much as 50% of the annual runoff takes place during snowmelt (e.g. Rosén, 1984). Runoff usually decreases in the growing season as a result of evapotranspiration followed by an increase during autumn storms. Also, the temperature decreases during autumn, which reduces evapotranspiration.

In general, the groundwater table lies within a couple of meters from the surface in rough terrain but gets closer to the soil surface as the terrain levels out. Wetlands, streams and lakes form where groundwater reaches the soil surface. The movement of groundwater depends on topography, which creates a mosaic in the catchment including areas where groundwater reaches the soil surface (discharge areas) and areas where groundwater is replenished (recharge areas). Discharge areas may be completely saturated with water (groundwater table=soil surface) or unsaturated if the hydraulic conductivity is high enough (groundwater table < soil surface). Some of the discharge areas are permanent (i.e. lakes, rivers and wetlands) even under dry periods of the year, whereas others only form during the wetter part of the year. In either case, discharge areas, including riparian zones, comprise a relatively small part of a catchment (Figure 16).

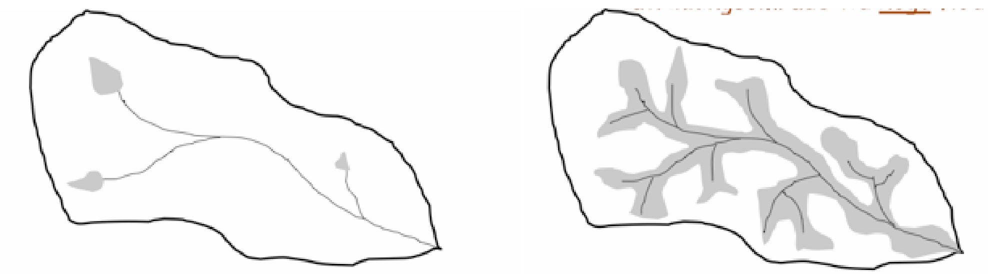


Figure 16. The topography in a catchment determines to what extent recharge and discharge areas occur. Some discharge areas are permanent (left) whereas others only form during periods of high groundwater levels. From Löfgren et al., 2008.

The water flow rate in a soil, or the hydraulic conductivity, is dependent on the soil texture. The flow rates are faster in coarse sediments than they are in fine clays. The hydraulic conductivity in sorted sediments is correlated to the grain size and porosity. In unsorted soils such as Nordic till, the hydraulic conductivity generally decreases with soil depth (Figure 17, lower) explained by differences in weathering, biological activity (earthworms, burrowing animals and penetrating roots), climate (freezing/thawing and drying/wetting) and compaction (Rodhe, 2003). During rainfall events the groundwater table rises to layer with higher hydraulic conductivity leading to increased transmissivity (Lundin, 1982; Lundin, 1990; Rodhe, 1987; Bishop et al. 1990). The pores become saturated with water and the connectivity to the stream increases compared with periods of low flow (Figure 17, upper and Figure 16).

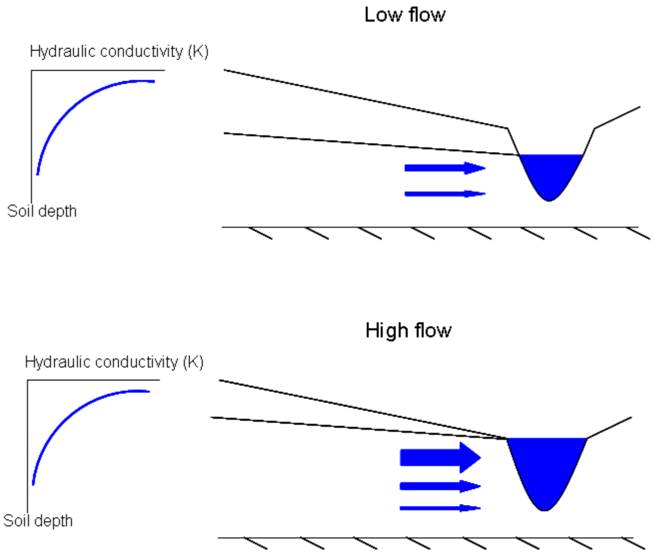


Figure 17. The hydraulic conductivity (K) increases with decreased soil depth. As a result, the groundwater contribution to the stream greatly increases during periods of high flows. Redrawn from Rodhe, 2003.

During periods of low flow, most of the stream water is dominated by groundwater (Figure 18). After a rainfall event, the water first entering the stream is usually generated from old groundwater (also called “pre-event water”) stored in the soil (Rodhe, 1987). Precipitation that cannot infiltrate the soil may also enter the stream as “event water” via overland flow on saturated discharge areas (including the stream itself). Thus, the largest volumes of groundwater in the stream are generated during short periods of time when the groundwater moves through saturated subsurface layers.

In areas where the soil becomes frozen during winter and depending on the soil water content at the time of freezing, a large fraction of the precipitation may still infiltrate the soil. The spring-flood in the northern part of Sweden has been shown, using isotope hydrograph studies, to be dominated by old pre-event water and not new melt water (Laudon et al. 2002).

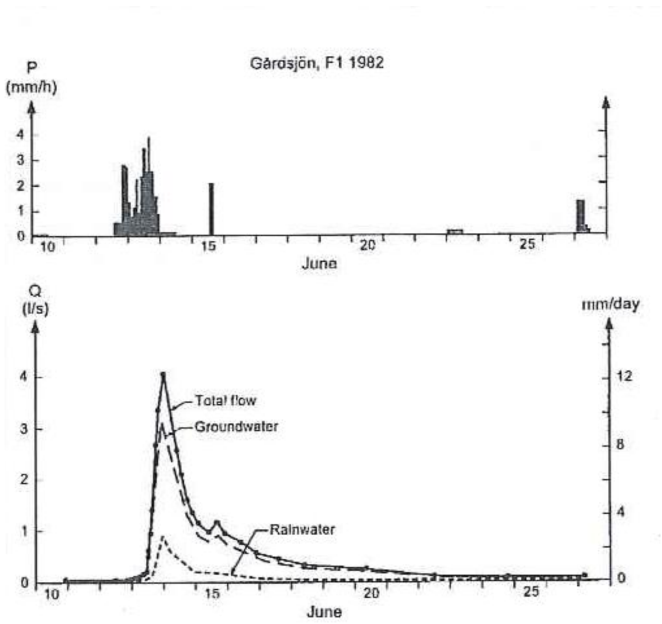


Figure 18. Most of the water reaching the stream after a rainfall event is old groundwater ("pre-event water") identified by isotope ^{18}O-analysis. Only a smaller portion of the rainwater ("event water") enters the stream. From Rodhe, 1987.

Despite much research, connecting subsurface storm flow to the heterogeneous nature and complex responses of a hillslope after a storm event is still a challenge. Nevertheless, there seem to be at least two criteria which must be fulfilled 1) a minimum input of rain/snowmelt necessary to generate a flow response (Tromp-van Meerveld et al. 2006a) and 2) connectivity between different areas of varying hydrological saturation within the catchment to activate a lateral flow of groundwater (Tromp-van Meerveld et al. 2006b). This is referred to as "a threshold response" to a rainfall or snowmelt event and the "spilling and filling" of saturated areas in the hillslope.

5.2 Dominating soil types in Sweden

Depending on soil forming factors (i.e. parent material, topography, climate and organisms) different soil types have developed through time (Jenny, 1941). Soil acidification resulting from deposition of anthropogenic pollutants may also have changed the chemical properties of soils and hence the soil forming process.

According to the Swedish National Inventory of Forests and based on the World Reference Base for Soil Resources (IUSS Working Group WRB, 2007), eight major soil types can be found in Sweden; Podzols (45 %), Arenosols (20 %), Leptosols (11 %), Cambisols (10 %), Histosols (8 %) and Regosols (6 %) distributed across the country (Figure 19). In addition, Umbrisols and Gleysols are found in much smaller areas not shown on the map.

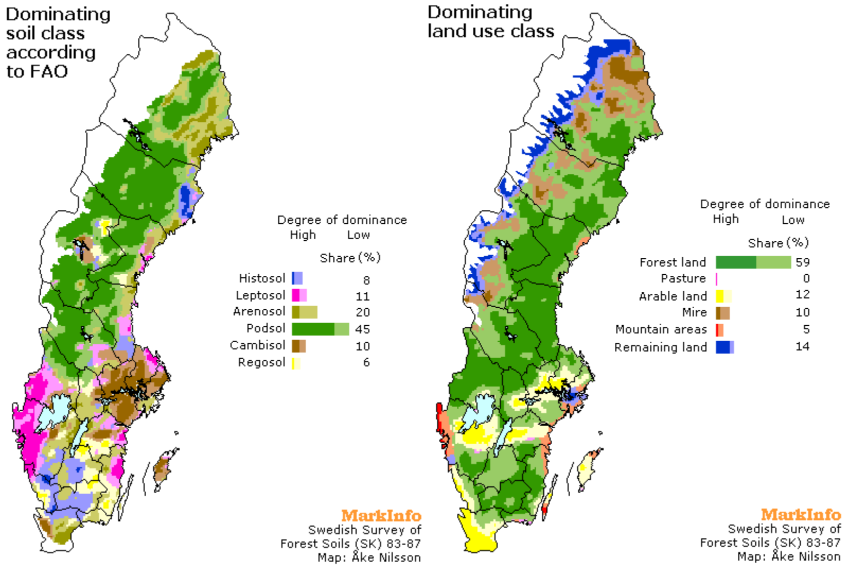


Figure 19. Dominating soil types in Sweden (left) according to the Food and Agricultural Organization (FAO) and the World Reference Base for Soil Resources (WRB) and land use classes (right). The colour saturation indicates areas where the soil type is the most dominant. From Markinfo (www-markinfo.slu.se/).

Depending on the dominant soil-forming factor, soils can be identified/grouped according to the following principles: Histosols are organic soils, which separates them from all other mineral soils. Next are podzols where iron and/or aluminium are important in the soil-forming process. Leptosols belong to a group of soils with limited rooting and very shallow and extremely gravelly soils, whereas Gleysols are examples of soils affected by water (groundwater). Finally, Arenosols, Cambisols, Regosols and Umbrisols are young soils with very little profile development. Arenosols are sandy soils, Cambisols are moderately developed soils, Regosols have no significant profile development and Umbrisols have very dark and acidic topsoil.

5.3 Recharge areas

Recharge areas, where groundwaters are replenished, are typically in upland forest areas. Podzols are the dominating forest soil type in Sweden (Figure 19). They are acid with an organic A-horizon, an ash-grey E-horizon and a reddish-brown to black B-horizon and an unaffected C-horizon (Driessen & Dudal, 1991). Most podzols in Sweden has developed on glacial till consisting of gneisses and granites. The mineralogy of podzols is variable, but they always have a high content of quartz, which give them a sandy texture in the E-horizon. The clay content is usually low. Water moves freely but may be restricted in the B-horizon and below in the presence of hardpans, fragipans and/or an indurated layer. Decomposition of the organic debris is slow.

Variations in soil chemistry at three typical podzolic sites in northern Sweden and Finland are shown in Figure 20. In general, pH (0.01 M $CaCl_2$) varies between 2.8 to 5 and increases with depth as a result of weathering processes deeper down in the soil (Figure 20). The concentration of exchangeable Ca^{2+} (and Mg^{2+}, K^+ and Na^+ not shown in figure) is higher in the upper part of the soil and decreases with depth. The highest titrable exchangeable acidity (pH=8.2) is found in the upper part of the soil where exchangeable Al^{n+} and H^+ are high. In contrast, the highest amounts of oxalate-extractable aluminium (Al (ox)) and diothinite-extractable iron (Fe (dit)) are found in the B-horizon (Figure 20). As percolating water moves downwards (alternatively if groundwater tables rises) the soil solution changes. The highest cation exchange capacity (CEC) is normally found in the upper part of the mineral soil and forest floor, coinciding with the greatest amounts of organic matter.

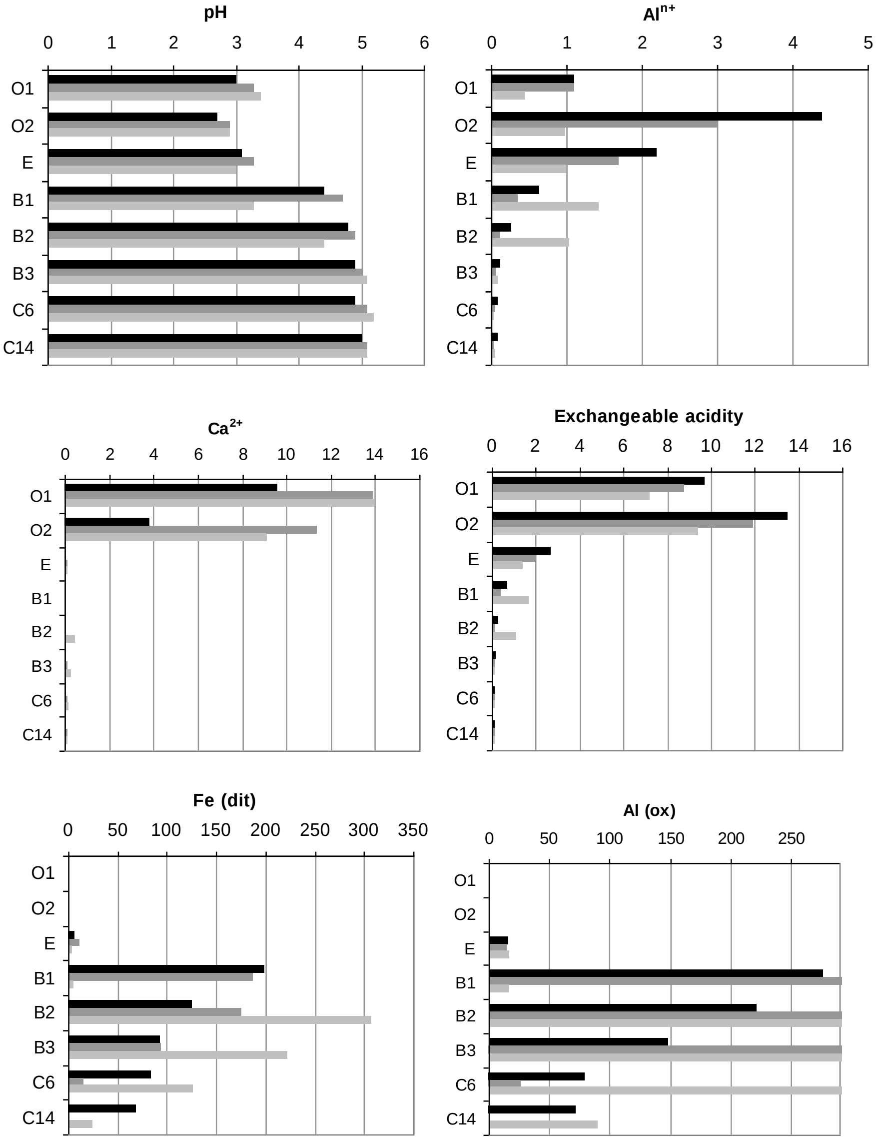


Figure 20. Variation in soil pH (0.01 M $CaCl_2$), exchangeable Ca^{2+} , Al^{n+} and Fe^{n+}(0.1 M $BaCl_2$, $cmol_c$ per kg) and extractable Fe (diothinite) and Al (oxalate) ex with depth at three podzolic forest sites in Sweden (n=2) and Finland (n=1). From Ilvesniemi et al. (2000).

Podzols develop through the process of podzolization and different theories have been put forward to explain the genesis of a podzol (reviewed by Lundström et al. 2000a). Independent of the theories, podzols have a zonal appearance including one upper layer from which materiel is moved from (eluviated) and a lower layer where material is accumulated or immobilized (illuviated). This requires a downward movement of water. The major podzolization theories were reviewed by Lundström et al. (2000a) including *eluviation* processes such as the formation of organic acids metal complexes and weathering of silicates forming Al-Si hydroxides in a colloidal suspension of solid particles (sols) and *illuviation* processes such as adsorption, precipitation and microbial degradation. A summary of the processes is given below.

Organic compounds in the soil solution can originate from leaching, decomposition and root exudates. Organic compounds include humic, fulvic and low molecular weight organic acids. The organic acids dissociate in water and bind with polyvalent ions such as Al^{3+} and Fe^{3+} on their carboxyl (R-COOH) and phenol (R-OH) groups (Figure 21). The unsaturated organic metal complex is leached from the upper layers of the soil together with organic substances and dissolved ions. This is the first step of the podzolization process whereby soluble organic metal complexes are transported (eluviated) to greater depths leaving a greyish E-horizon.

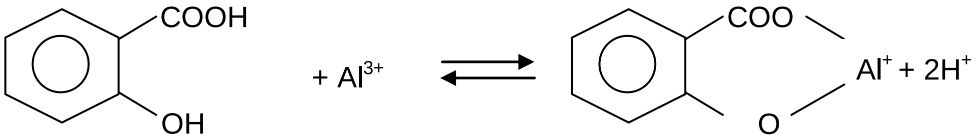


Figure 21. The formation of an organic metal complex.

The second step involves the accumulation (illuviation) of the metal complexes in the spodic B-horizon. Precipitation of the complexes can be brought about by a number of factors.

One important mechanism has to do with Al and Fe solubility. As the metal/carbon (Me/C) complex moves down with the percolating water, more and more of the reaction sites will be occupied by Al^{3+} and Fe^{3+} and as a result the solubility of the complex decreases. At some point, enough reaction sites will be occupied and the complex will precipitate. Alternatively, the complex can be adsorbed to other minerals e.g. Al- and Fe-oxides. Other factors such as the concentration of Al^{3+} and Fe^{3+} in the soil solution, changes in acidity, ionic strength and desiccation are also important. For example, saturation of the organic metal complex can be caused by evaporation (desiccation) or if complexes with a low Me/C-ration absorbs to complexes with a high Me/C-ratio already precipitated. The distance by which the complex moves depends on the number of acid groups per mass unit C and their acid strength and complexing ability (Lundström et al. 2000a). Normally, the complex moves a couple of decimetres in the soil before precipitation. The process by which the organic matter moves and precipitates is mainly attributed to fulvic and humic acids (Lundström et al. 2000a).

Another mechanism, which could explain the formation of a B-horizon, is through microbial action. It has been seen that the organic part of an organic metal complex is subject to microbial degradation by fungi and bacteria. In the process, Al^{3+} and Fe^{3+} are released leaving them free to precipitate as oxides or hydroxides. This process is mainly attributed to easily degradable low molecular weight organic acids.

Organic metal complexes may not always be necessary in the podzolization process. Instead mineral weathering of aluminosilicates (i.e. feldspar and plagioclases) in the upper soil horizon produce Al^{3+} and silicic acid (H_4SiO_4) which react with each other to form a soluble inorganic positively charged hydroxy-aluminium silicate complex (also known as a proto-imogolite complex) in the E-horizon. This product is stable at low pH (<5). The complex is transported down in the soil and as the pH rises it becomes more and more insoluble and accumulates in the B-horizon. Eventually Al^{3+} turns into imogolite and/or allophane. Fe^{3+} is supposed to react in a similar way as Al^{3+} forming the oxide ferrihydrite. The presence of organic matter in the B-horizon is then explained by the downward transport of soluble negatively charged organic matter, which precipitates on the imogolite, allophane and/or ferrihydrite surfaces. Lundström et al. (2000b) showed that microbial degradation of the organic part of the organic metal complex (see above) releasing Al^{3+} and Fe^{3+}, which in turn forms imogolite-type material (Al) and ferrihydrite (Fe), is the most probable explanation for the immobilization of Al and Fe in the B-horizon.

5.4 Discharge areas

Peatlands, wetlands, mires, bogs and riparian areas are examples of ecosystems found in discharge areas. The general characteristics of these systems are that they are completely or partially saturated with water, leading to reduced conditions and a build-up of organic materials. These areas act as sinks and sources for many elements as well as serving other important ecological functions. The hydrology in these systems is the most characteristic feature. Bishop et al. (2004) has pointed out the importance of discharge areas, in particular the riparian zone, for affecting stream water chemistry.

Common soil types found in discharge areas are Histosols and Gleysols. Histosols are identified as soils with at least 40 cm or more of organic materials extending down from the soil surface (Driessen & Dudal, 1991). The organic material consists mainly of partially decomposed plant material (i.e. sphagnum and other mosses), which may or may not be mixed with mineral soil. They form in lowland areas in the boreal regions with a high groundwater level, which leads to oxygen-depleted conditions and a slow decomposition. The soils are permanently wet with the groundwater level close to the soil surface. The bulk density is low. Most Histosols are acid with a very low inorganic mineral content. Humic substances form in the organic soil, which bind with metal ions (i.e. micro- and macronutrients) and are leached out of the system. The amount of nutrients is always higher in the uppermost surface layers and is a reflection of the nutrient cycling of the vegetation. Traditionally, Histosols are nutrient-poor and not suitable for farming or forestry.

Gleysols are mineral soils with a high groundwater level in some period of the year or throughout the year. As a result, they are found in lowland areas, in depressions or at the foot of slopes. Gleysols have a grey, blue or olive coloured matrix brought about by the reduction of Fe^{3+} to Fe^{2+}, which is easily leached from the system or forms FeS (which gives the soil its colour) (Driessen & Dudal, 1991). In systems with a fluctuating groundwater level, Fe^{2+} can be oxidized back to Fe^{3+}, which precipitates as red mottles in former root channels and cracks in the soil. Organic matter content, cation exchange capacity and base saturation are all high in Gleysols.

The chemical composition of water moving out from a discharge area is different from the water moving into the area. In general, the water becomes enriched with DOC and the Al chemistry is largely affected partly due to pH increases (Löfgren and Cory, 2010).

5.5 Catchment level

The way in which groundwater is mobilized to the stream and how the chemical characteristics changes have been the subject of much research. It is fair to say that stream water chemistry does not equal soil and soil water chemistry. As described above, old “pre-event” water reaches the stream shortly after a rainfall input but the response of the chemical constituent varies. Some constituents increase (e.g. DOC), some decrease (e.g. Ca^{2+}) and some remain fairly constant (e.g. Cl^{-}). The chemical signal of different constitutes in the stream varies strongly depending on the groundwater level and thereby the rate of flow. At the same time most of the water found in a stream after a storm-event is old pre-event water (e.g. Kirchner, 2003).

Bishop et al. (2004) and Seibert et al. (2009), give a plausible explanation, stressing the importance of the riparian zone (the area that connects the mineral soil to the stream) for stream water chemistry. The RIM-model, which was developed based on hillslope studies in the northern part of Sweden, describes how the riparian zone control stream water chemistry, especially the short-term dynamics, as the groundwater passes the riparian soil solution (Figure 22). The model explains how variations in stream water chemistry depend on the variation of solutes and water with depth in the riparian zone. It is a simple model using a flow-concentration model based on soil-water chemistry and lateral water flux. Löfgren & Cory (2010) showed the importance of the near stream zone for stream water concentrations of Al.

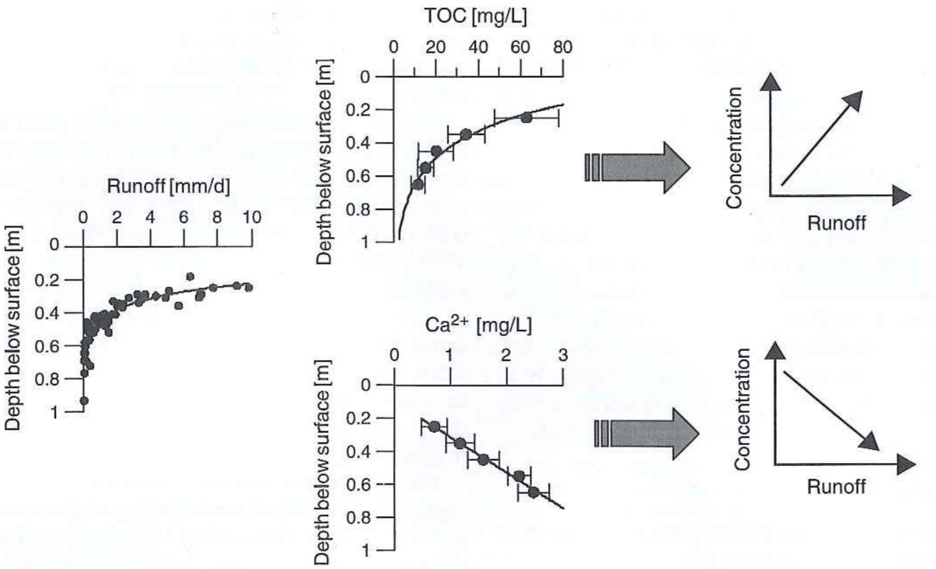


Figure 22. The riparian convolution model where water moves in a lateral manner through the riparian soils. The hydraulic conductivity in the soil increases with decreased depth (left). During a rainfall event, the groundwater table rises to soil horizons with higher concentrations of TOC and lower concentrations of Ca^{2+} (middle), which gives an increase of TOC in runoff and a dilution of Ca^{2+} (right). Figure from Bishop et al. 2004.

Hillslopes are fundamental landscape units in a catchment. Most catchments also have a riparian zone, which transforms the water coming from the hillslope before it enters the stream. Separating and quantifying sources of solutes coming from a hillslope during different flow situations still remain a problem.

Thus, in successfully describing headwater stream chemistry using dynamic models one must first identify the spatial variation of terrestrial landscape units (e.g. forest soil, wetlands, peat lands and riparian zones). Secondly, differences in soil types and vertical and lateral chemistry should be recognized in order to understand how rainfall turns into soil water, groundwater and ultimately surface water. Finally, interactions between land cover and hydrological connectivity (vertical and horizontally) must be defined in order to correctly describe the volume and chemical composition of water depending on residence time of water and temporal variation. Considering the large variability within catchments, lumping and other simplifications are necessary. Recent research have shown the importance of the riparian zone in determining stream water chemistry (Bishop et al., 2004; Burt & Pinay, 2005; Seibert et al., 2009; Löfgren & Cory, 2010) why special attention should be given these areas in modelling efforts.

6 Material and methods

The main focus of this literature report is to describe clear-cutting effects on soil, soil water, groundwater and stream water following final felling and biomass removal. In addition, clear-cutting effects on hydrology are discussed. The mechanisms responsible for the results are explained and discussed in terms of site-specific conditions (catchment characteristics and hydrology). When possible, treatment differences between whole-tree and stem-only harvest are discussed.

The review describes research carried out mainly in Europe, the US and Canada but studies from New Zeeland and southern Australia have been considered although climate, soil types and forest practices may differ. Unfortunately, older studies (mainly Swedish) have not been published in the scientific literature but have been included as they are regarded too valuable to be omitted (e.g. Wiklander, 1974, Wiklander 1983; Tamm & Wiklander, 1972; Nykvist 1974a, b; Nykvist 1999; Örlander et al., 1997; Westling et al., 2004; Haveraaen, 1981 and Zetterberg et al., 2008). A majority of the studies describe short-term (<5 years) clear-cutting effects since this is usually the period during which quantitatively important changes occur. Long-term effects (>15 years) are scarce and there are no studies describing treatment effects later than 31 years (Zetterberg et al., 1998). The Swedish database Silvaboreal for forest research (www.silvaboreal.com), the Northern European database NOLTFOX for long-term field trials (http://noltfox.metla.fi/) and the LTSP study (http://www.for.gov.bc.ca/hre/ltsp/ or http://www.ncrs.fs.fed.us/ltsp/network/) has provided useful information in the search for studies along with traditional literature search via the databases Web of Knowledge and Google.

There are many articles describing short-term effects of clear-cutting (especially on NO_3^- concentrations) already well described in the literature. Therefore, only some of them have been included in this report. Instead, focus has been on articles describing effects on concentrations, pools and fluxes of base cations. In the literature, the terminology for logging residue fuels (hereby referred as logging residues) has varied through time (e.g. slash, brash, logging debris etc.) but with the same meaning, namely the above ground material left on ground following harvesting. Stump removal, which has gained a renewed interest recently, is also a logging residue fuel but a distinction is usually made between above- and belowground residues when referred to in the literature. Compared with the removal of aboveground material, logging of both above- and belowground material is believed to cause relatively large effects due to mechanical disturbances. Hence stump removal studies have been omitted. Logging residues in thinnings are not discussed since the effects are less pronounced compared with final felling (e.g. Egnell & Leijon, 1997; Olsson, 1999; Rosenberg & Jacobson, 2004). Statistically significant differences are not always reported, but yet these studies are included.

Depending on country, tradition and time, a range of different forest management practices have been used including various methods for harvest and for regeneration. For example, in a final felling study by Simard et al. (2001) cutting with protection of regeneration and soils (CPRS) was used. CPRS resembles clear-cutting but great care is taken to ensure that the natural regeneration of the new stand is successful without the need to replant. Sometimes the entire catchment has been harvested (e.g. Briggs et al., 2000) or only parts of it, referred to as partial cutting by Feller (2000) or patch cut by Tiedemann et al. (1988). In other experiments measures have been taken to limit erosion, nutrient leaching and a rise in groundwater table including progressive strip-cutting (alternating uncut and cut strips, e.g. Martin et al., 2000) or shelterwood-cutting (where trees are left, e.g. Lundin, 1999; Wang et al., 2006). In some studies, diameter-limit cutting (or single tree selection) is practiced where only trees above a certain diameter are logged (e.g. Patric & Aubertin, 1977: Rothe & Mellert, 2004). Finally, in some cases the effect of progressive harvesting on stream water response has been studied (Rusanen et al., 2004; Neal et al., 2004).

In some studies buffer zones varying between 10-30 meters (e.g. Hornbeck & Kropelin, 1982) up to 50-75 meter (Tiedemann et al., 1988) on each side of the stream have been left uncut. In other studies herbicides and pesticides have been used to control competing vegetation (e.g. Bormann et al., 1968; McLaughlin & Phillips, 2006; Smethurst & Nambiar, 1990). In conclusion, the difference between the techniques lies in the areal extent of the catchment being harvested (0-100%) and if trees are left in the catchment to reduce nutrient leaching and a rise in groundwater table. Following harvest, the logging residues are either left on ground or removed. When the logging debris is left on ground to decompose it may be evenly spread on the ground (e.g. Olsson et al., 1996a), concentrated in piles (e.g. Rosén & Lundmark-Thelin, 1987) or placed in rows (Brais et al., 1995). In a few instances the logging debris has been burned (e.g. Harr & Fredriksen, 1988; Belleau et al., 2006)

In some soil and soil water studies it has not always been easy to determine logging techniques. Therefore, measurements carried out between piles have been termed WTH and measurements below piles CH (e.g. Wall, 2008; Rosén & Lundmark-Thelin, 1987; Hornbeck & Kropelin, 1982). Sometimes the vocabulary for logging technique varies but in all cases it has been possible to determine whether or not logging residues have been left on ground or not. For instance Ranger et al. (2008) studied the effect of clear cutting on soil water chemistry, which could easily have been interpreted as stem-only harvesting. However, in the material and method section it was stated "*slashes were manually windrowed outside the measurement area"*. Hence, it is the effect of whole-tree harvesting that is described.

Silvicultural methods used for regeneration include soil scarification, fire, protective ditching or ditch maintenance, planting, application of pesticides or herbicides, pre-commercial thinning, thinning, fertilization, liming, ash recycling etc. These operations are not always specified in the literature making it difficult to separate the effect of felling from other measures. As a result, it is many times the combined effect of forest harvesting (with or without logging residues) and silvicultural measures that is described (e.g. Rosén, 1996). Sometimes the differences are too great to allow for comparisons to be fully made. For example, Ring et al. (2001) studied the effects of clear-felling of a previously fertilized forest (doses ranging between 0 and 2400 kg per hectare) with and without logging residues. In spite of the last dose being applied thirteen years before clear-felling, residual effects were still measurable in the soil. Hence, only data from the sub-plots where no fertilization were applied is included in this study. In another example, soil scarification including ploughing, hummocking and drainage was carried out in the Nurmes-project in Finland, three years after harvesting and drainage (Ahtiainen, 1992; Ahtiainen & Huttunen, 1999). In order to reduce the number of factors, data between the third and fifth year was not considered in this report.

Other treatments have also been omitted. For example, Smethurst and Nambiar (1990) studied the effects of slash and litter management using four different treatments; slash and litter retained (SL), only litter retained (L), litter ploughed (LP) and slash and litter removed (SLR). The LP and SLR treatments have been omitted and although the authors did not use the terminology usually associated with biofuel research, the SL and L treatments have been regarded as the equivalent to CH and WTH. Other treatments such as winter windrowing following whole-tree harvest (see Brais et al., 1995) have been also been omitted.

Besides the use of different forest management practices, other challenges were presented when compiling the data. For example, the studies cover a broad range of tree species, ages, soil types, soil horizons, site quality classes adding to the difficulty to evaluate treatment effects. Hence, some simplifications were necessary. Different tree species have been categorized into two groups: hardwood and softwood species. Sometimes mixed softwood/hardwood stands occur in the studies. Note that as a result of geographical segmentation and active forest management decisions, different tree types naturally dominate in different parts of the world. Example of common tree species in the US and Canada are Sugar maple, Yellow birch and American beech often found in the Northern hardwood forests. In Europe, Scots pine, Norway spruce and Sitka spruce are example of traditional softwood species used primarily in forest whereas Radiata pine is used in New Zeeland and Australia. Special attention is made to studies including alder species because of their N fixing ability, which fertilizes soils (e.g. Johnson et al., 1988; Robertson et al., 2000; Titus et al., 1997; Mann et al., 1988).

Most countries have some system for classifying soils, which means that different names are applied for soils with the same characteristics. In this review the World Reference Base (WRB) system is used and the corresponding name according to other classification system is given in Table 4. Note that suborders of the order Inceptisols can be either Cambisols or Gleysols, which is problematic when the soils are only defined as “Inceptisols” (e.g. Mann et al., 1998).

Table 4. The soil types included in this review according to WRB and the corresponding name at the order level (sub-order in parenthesis) in the USDA Soil Taxonomy system according to Chesworth (2008) and in the Référentiel pédologique according to Baize & Girard (2009).

World Reference Base for Soil Resources (WRB)	USDA Soil Taxonomy	Référentiel pédologique (RP)
Acrisols	Ultisols (all suborders)	-
Cambisols	Inceptisols (Ochrepts, Umbrepts)	Alocrisol
Podzols	Spodsols (Aquods, Orthods, Humods)	-
Leptosols	Lithosols and lithic subgroups	-
Andosols	Andisols	-
Gleysols	Entosols (Aquents) and Inceptisols (Aquepts, Aquolls)	-
Histosols	Histosols (all suborders)	-
Luvisols	Alfisols (Udalfs, Xeralfs, Ustalfs, Aqualfs, Boralfs)	-

Considering the different horizons from which soil and soil water data have been collected, it is necessary to find a way to rearrange the results into a few categories only. First a distinction is made between the forest floor (all varieties of L, F, H, forest floor, O, Oa, Oi, Oe, humus etc.) and the mineral soil (A, E, B and C-horizons). Secondly, the mineral soil is further divided into horizons were materials are leached (eluviated) or accumulated (illuviated). Obviously, this is much easier when the name of the horizon has been specified (e.g. A, E, B or C) compared with when the soil depth is defined in centimetres.

Podzols have been used for guidance when categorizing mineral soil data. The depth of the A-, E- and B- horizon in Swedish forest soils usually reaches 4-8 cm (Eriksson et al., 2005). In the new WRB classification systems, podzols are merely defined as soils with a spodic horizon within 200 cm of the soil surface. However, in previous systems, podzols were defined as having a spodic B-horizon *below* 12.5 cm depth from the soil surface or below an A- or E-horizon (being more than 2.5 cm thick) (FAO-UNESCO, 1997). Although not in use anymore, this definition still provides information on how to separate the eluviated horizon from the illuviated, using an arbitrary boundary of circa 10-15 cm. Hence, data from 0-5, 5-10, 0-10, 0-15 cm is grouped into soils ≤10 cm and soils from 10-15, 10-20, 10-30, 15-20, 15-30, 20-C, 30-45 and 45-60 cm are grouped into soils ≥10 cm. This boundary has been used regardless of soil type. Similarly, data from the A, Ap, E, E/B-horizon is compared with soil data from ≤10 cm and Bh, Bs, Bs1, Bs2, Bt, BC, and C with soils ≥10 cm. However, this approach made it difficult when mineral soil data is treated as a single unit (e.g. mineral, 0-100, 0-C or ∑ FH 0-20, total solumn etc.). For simplicity, data from the “mineral”, 0-100, 0-C and total solumn were grouped into soils ≥10 cm. Summarized data from the forest floor and mineral soil (e.g. ∑ FH 0-20 and forest floor+mineral soil) has not been included, but are referred to in the discussion when necessary.

In terms of soil water data, plant uptake of water and nutrients generally take place in the upper 50 cm of the soil (including the A, E- and B-horizons of the soil) why this has been considered an appropriate limit to separate upper from lower horizons. Changes in the soil percolates below 50 cm will eventually reach the groundwater and possibly stream water and is thus an indicator on whether or not to expect any changes further down the system.

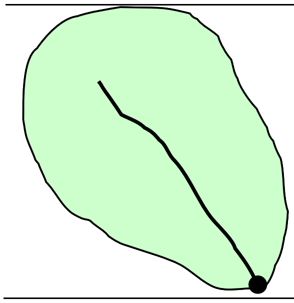
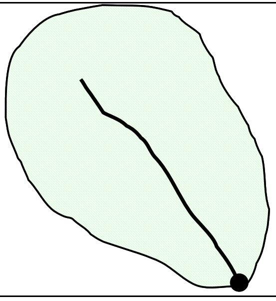
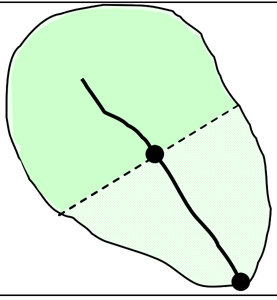
Effects on soil, soil water, groundwater and stream water are described separately with respect to concentrations ($cmol_c$ per kg, mg per g and µeq per l), pools ($kmol_c$ or kg per hectare and year) and fluxes ($kmol_c$ or kg per hectare and year). In some cases, data is only reported in the form of figures (e.g. Staaf & Olsson, 1994). When this is the case, data from the figures are used to express the treatment effect as higher/lower of increase/decrease. Sometimes only the sum of NO_3^- and NH_4^+ (inorganic N) is reported (Martin et al., 1984; Stevens et al., 1995). However, inorganic N is often dominated by NO_3^- why these results have been interpreted as NO_3^-.

Treatment effects on soil chemical properties have been measured at ambient soil pH (using unbuffered extraction solutions) or at pH=7 using a buffered extraction solution. A direct comparison between CEC and cation concentrations measured at different pH is not recommended since a buffered extraction solution will results in an overestimation of CEC due to an increase in pH-dependable charges (Borge, 1997; Skinner et al., 2001). In two Swedish studies (Nykvist, 1977b; Nykvist, 1990) ammonium lactate extract were used in determining K and P concentrations (according to Egner et al., 1960). It is unclear how this affects the results. pH is usually determined using different salt solutions (KCl, $BaCl_2$, $CaCl_2$) or deionized water (H_2O). When measuring pH in soils using these methods, the latter produces a higher pH-value compared with the salt method. Also, using monovalent salt solutions of the same molality results in higher pH-values compared with divalent solutions. Thus, $pH(H_2O) > pH(KCl) > pH(BaCl_2) = pH(CaCl_2)$.

Soil water has been collected using ceramic suction cups (Jewett et al., 1995) or zero-tension lysimeters (e.g. Ranger et al., 2007; Wall, 2008) while groundwater has been collected using piezometers (e.g. Parfitt et al., 2002), wells (Kubin, 1998) and boreholes (Neal et al., 2004), spring water (Tamm et al., 1974) and large aquifers (Rusanen et al., 2004). Stream water is collected from stream weirs allowing for water discharge measurements to be made (e.g. Rosén, 1984). Sometimes changes in discharge have been calculated using models such as BROOK (Hornbeck et al., 1990; Briggs et al., 2000) and PULSE (Brandt et al., 1988). Effects are usually measured in first order streams since studies describing downstream effects are scarce.

The studies also differ with respect to experimental design (plot, catchment and chronosequence). The most common experimental design in measuring treatment differences in surface waters is the paired catchment approach in which one of two adjacent catchments is treated and the other is untreated and thus used as a control (e.g. Bormann et al., 1968). Alternatively, the single catchment approach can be used in which the upper part of the area remains as a control and the lower part are used for various treatments (e.g. Löfgren et al., 2009; Neal et al., 1992). In the latter case, the same stream drains both the upper and the lower part of the catchment. The two types of experimental designs are illustrated in Table 5. The treatment can either be carried out in the entire catchment or only parts of it. In addition, chronosequences have been used (Katzensteiner, 2003; Snyder & Harter, 1984) as well as transect studies (Parfitt et al., 2002). In soil- and groundwater experiments, block experiments are traditionally used where some of the plots are treated and others left untreated (control) at each site. However, in some groundwater studies the wells are located below the clear-felled area (e.g. Mannerkoski et al., 2005).

Table 5. Examples of experimental designs for measuring treatment differences in surface water. Water is collected at the catchment outlet indicated by a black dot.

Pairwise catchment approach		**Single catchment approach**
Control	Treatment	Control + Treatment
		

Treatment effects are calculated by subtracting data from the treated plot/catchment with a control plot/catchment or by comparisons with statistical models between the treated and control site before and after treatment (e.g. Löfgren et al., 2009b). That is, differences *between* plots/catchments are described (Table 6, pairwise comparisons). However, on rare occasions, data for the control catchment is reported as a mean for the whole study period while the data for the treated catchment is divided into a pre- and post-treated period (e.g. Neal et al., 2004). Thus, the pre-treatment or post-treatment period for the treated catchment is compared with the data from the whole study period for the control catchment. Another way to calculate the treatment effect is to calculate the ratio between stream water chemistry from the control and treated catchment *before* harvest and used this ratio to multiply with data from the control catchment *after* the post-treatment period. This data are assumed to represent what the stream chemistry of the treated catchment would have been if it not had been harvested ("if uncut"). The difference between the actual data and this estimate is attributed to harvest effects (e. g. Hornbeck et al., 1990; Lundin, 1999). Comparisons can also be made *within* a treated catchment describing the combined effect of time and treatment (Table 6, time series). This can be made either for the pre- and post-treatment period or, if the pre-treatment period is lacking, between the post- and post treatment period. For the control catchment, only changes over time are described using this approach. Finally, comparison can be made in different stands of varying ages using a chronosequence approach (Table 6 chronosequence).

The results are thus presented as increases or decreases between the pre- and postharvest period or as differences (higher or lower) between treatments postharvest. In some experiment pre-harvest values are missing (e.g. Wiklander, 1983; Lundin, 1999; Stevens & Hornung, 1988; Westling et al., 2004; Örlander et al., 1997; Nykvist, 1977b; Smethurst & Nambiar, 1990; Ranger et al., 2008). This is usually the case when the studies begun in conjunction with harvesting, i.e. year 0 (e.g. Ranger et al., 2008). The different treatments include control stands (CTR), logged stands using conventional methods for final felling (CH) and various methods for logging above- and belowground residue fuel including conventional harvest plus branches (BSH) and whole-tree harvest where conventional harvest plus branches and needles was used (WTH) (Table 7). In terms of biomass removal; WTH > BSH > CH > CTR.

Table 6. Changes in soil, soil water, groundwater or stream water is usually described as differences over time or between treatments using the following combinations. t=0; time for treatment, -t and +t are time before and after treatment, respectively.

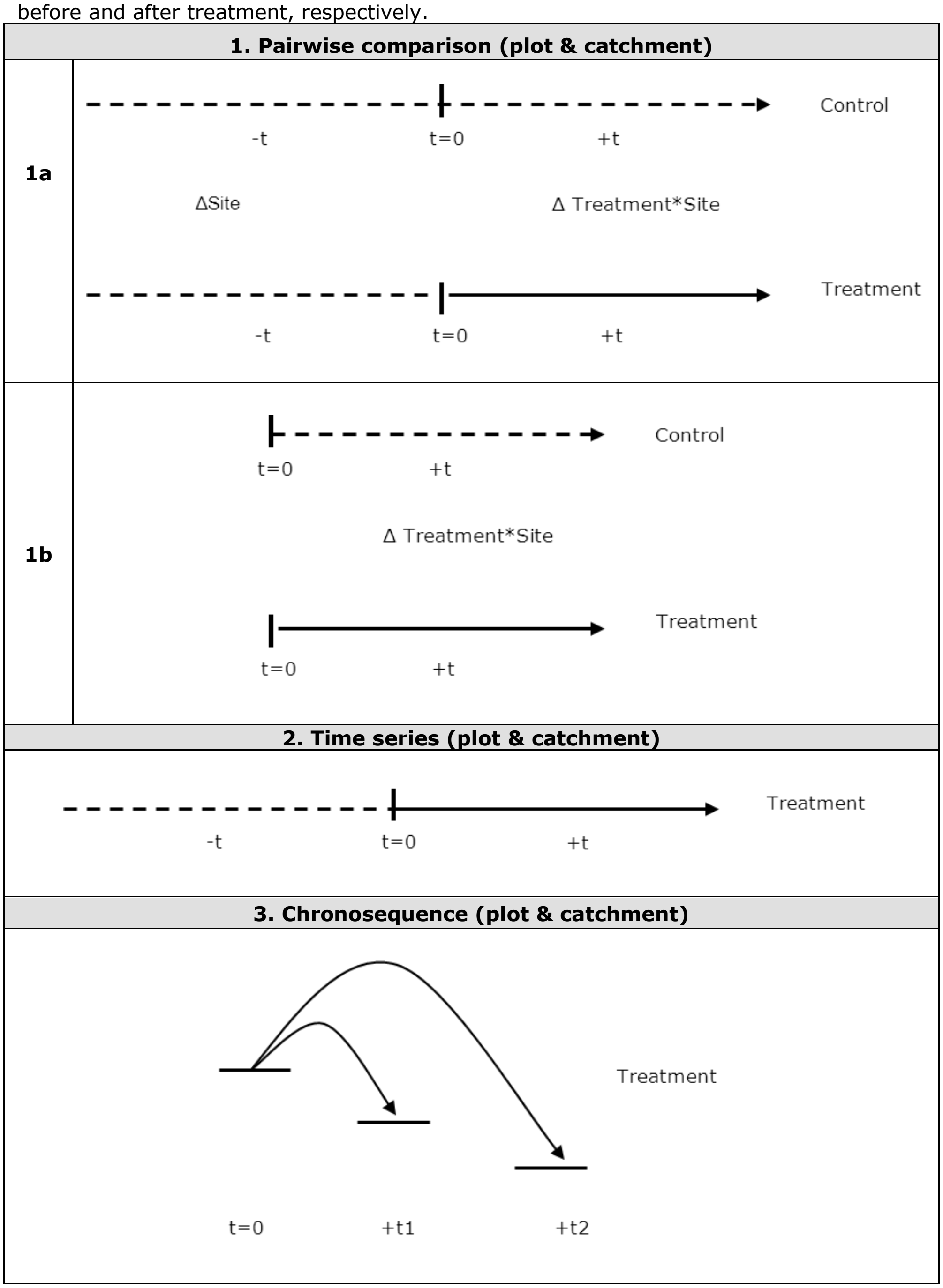


Table 7. Different logging techniques and their abbreviations used in this review.

Treatment	Abbreviation	Note
Control	CTR	Uncut often mature forest.
Conventional harvest	CH	Only stems are removed leaving the logging residues left on ground. Also known as stem-only harvest.
Conventional harvest + branches and needles	WTH	Removal of stems, branches and needles except for stumps. Also known as whole-tree harvest.

7 Results

7.1 Hydrology

In the short-term, stream runoff usually increases following final felling ranging between 4-450 mm per year as a result of decreased evapotranspiration, causing increased soil water content and groundwater tables (Table 8). In general, the greater the fraction of the catchment being cut, the larger the stream water response is. Most studies report short-term effects but changes up to 27 years have been measured (Table 8). During the first 7 years following stem-only cutting (CH) stream runoff increased between 46-117 mm per year but in the subsequent years (7-27 years) runoff decreased (Martin et al., 2000). In long-term (0-14 years) whole-tree harvesting (WTH) studies, stream runoff has been found to increase ranging between 51-152 mm.

Studies comparing differences between CH and WTH are largely lacking. However, according to Rosén (1984) no differences in stream runoff could be measured during the first two years after cutting (not shown in Table 8). Changes in soil water content and groundwater tables are only reported following CH in the range of -52 to +103 mm and -3 to +96 cm, respectively (Table 8).

Table 8. Hydrological changes following stem-only (CH) or whole-tree harvest (WTH). Pre-harvest regression analysis between a “cut” and uncut control area are often used to simulate post-harvest stream runoff if cutting had not occurred. Post-harvest difference between the “forested” and non-forested runoff is then calculated and reported as a mean cutting effect. This can be done for the whole study period or for individual years. Changes in soil water content and groundwater tables are reported as means. NS=non-significant changes.

Reference	Harvest intensity	Annual increases in runoff (Δ mm)	Soil water (Δ mm)	Groundwater (Δ cm)	Years after (yrs)
Hibbert (1967)	CH	+34-450			
Bosch & Hewlett (1982)	CH	+40 per 10% cut (conifer) +25 per 10% cut (hardwood) +10 per 10% cut (shrubs)			
Rosén (1984)	CH; WTH	+215 (CH)			0-2
		+371 (WTH)			
Rosén et al. (1996)	CH; WTH	+220 (CH)			0-8
		+274 (WTH)			
Grip (1982)	CH	+184			0-2
Lundin (1979)	CH		+5-46	+14-96	0-3
Brandt et al. (1988)	CH; WTH	+165 (CH)			0-3
		+220 (WTH)			
Haveraaen (1981)	CH	+200-250			0-2
Bormann et al. (1968)	CH	+40%			0-1
Likens et al. (1970)	CH	+270-330			0-2
Swank et al. (2001)	CH	+40-260 mm (4-28%)			0-5
		NS			0-14
		+6.6 (7%)			15
		NS			16
		-67 (-7.2%)			17
		NS			18-20
Martin et al. (2000)	CH; WTH	+46-114 (CH)			0-7
		-21 to -66 (CH)			8-27
		+51-152 (WTH)			1, 7, 8 & 13
Hornbeck et al. (1990)	WTH	+220-310			0-1
Mannerkoski et al. (2005)	CH			NS	0-5
Briggs et al. (2000)	WTH	+310			0-1
Patric & Aubertin (1977)	CH	+34-64			0-2

Table 8. Continued.

Reference	Harvest intensity	Annual increases in runoff (Δ mm)	Soil water (Δ mm)	Groundwater (Δ cm)	Years after (yrs)
Robinson & Dupeyrat (2005)	CH	+60-200			0-10
Sørensen et al. (2009)	CH	+253-254			0-2
Heikurainen & Päivänen	CH	+186		+5 -14	0-2
Jewett et al. (1995)	CH	+9.2%			0-12
Adamson et al. (1987)	CH	100%			0-2
Wang et al. (2006)	CH	NS			0-1
Kilhberg (1958)	CH	+85			0-5
Caissie et al. (2002)	CH	NS			0-2
Pyatt et al. (1985)	CH			+5, 20 and 35*	4
Lundin (1999)	CH	+4-125		+5-11	0-4
Rusanen et al. (2004)	CH			NS	0-22
Keppeler et al. (1994)	CH	increase			0-4
Adams et al. (1991)	CH	0-3 4-20	+10-103 -52 to -10		
Dubé et al. (1995)	CH CH			+7-52 -3	0-1 0-1
Min-max		**-66 to +450**	**-52 to +103**	**-3 to +96**	

*average increases during period 1-3 corresponding to week 1-12, 13-24 and 25-36 respectively.

7.2 Soil chemistry

In this chapter, short- and long-term treatment effects on soil chemistry after clear-cutting are compared with pre-harvest conditions (Δtime*treatment) at two harvest intensities. The results are presented in four different tables with a distinction made between exchangeable and total concentrations (Table 9 and Table 10) and exchangeable and total pools (Table 11 and Table 13) at three soil depths (forest floor, 0-10 cm and ≥10 cm mineral soil). Postharvest effects (Δtreatment) between different treatments are described in a similar way in Tab le 13 and Table 14 (exchangeable and total concentrations) and Table 15 and Table 16 (exchangeable and total pools). First, differences between clear-cut areas and growing forests (i.e. CH↔CTR and WTH↔CTR) are presented followed by a comparison between different harvest intensities (WTH↔CH).

7.2.1 pH and H^+

Most sites showed no short- or long-term changes in pH after cutting compared with pre-harvest conditions or non-harvested control forests, independent of harvest intensity (Table 9 and Table 13). However, short-term (≤5 years) increases in mineral soil pH have been observed following whole-tree harvest compared with pre-harvest values and controls (n=7, Table 9 and Table 13). When comparing stem-only harvested plots with control plots, higher pH was found in the 0-10 cm soil layer (n=2, Table 13). These increases are of short duration and have largely disappeared after six years. In comparison, whole-tree harvest has also been shown to cause decreased soil pH. For example, Johnson et al. (1991b) observed a short-term decrease in forest floor pH in the range of -0.2 to -0.3 pH-units after whole-tree harvest (Table 9). Similarly, Brais et al. (1995) observed lower pH values in the forest floor and mineral soil after cutting (WTH) compared with mature forest (Table 13).

Studies describing changes in exchangeable H^+ pools are only available after a whole-tree harvest study by Johnson et al. (1997) (Table 11). Following WTH, no short- or medium term changes could be found in the forest floor and the mineral soil (0-10 and ≥10 cm). The long-term effects are unknown. There are no studies describing differences in pools between clear-cut areas and control forests (Table 15).

The difference between harvest intensities (WTH↔CH) is usually non-significant except for the forest floor where lower pH-values (and higher exchangeable pools of H^+) have been reported after whole-tree compared with stem-only harvest (Table 13 and Table 15).

7.2.2 Carbon (C) and CEC

Harvest effects on total soil C concentrations and pools have been reviewed by several other authors such as Johnson & Curtis (2001), Sanchez et al. (2006) and Kreutzweiser et al. (2008) why emphasis in this report has been on cutting effects on base cations. This explains the low number of observations in Table 10, Table 12, Table 14 and Table 16). Still, some distinct trends can be seen regarding the pool changes. The total soil C concentration references (Table 10 and Table 14) are too few for making any statements.

In terms of total C pools, short- and long-term decreases in the forest floor and increases in the mineral soil have been documented after CH and WTH (Table 12). However, compared with uncut control plots, the long-term effects on soil C pools following CH and WTH are primarily non-significant (Table 16).

Generally, comparisons between different harvest intensities (WTH↔CH) show that there is no difference in total C pools (Table 16), with the exception of Johnson et al. (2002) who reported smaller forest floor C pools but higher C pools in the mineral soil following whole-tree harvest compared with stem-only (Table 16).

Short- and long-term treatment effects on CEC are generally non-significant when comparing clear-cut areas with pre-harvest conditions, regardless of whether or not logging residues have been collected (Table 9). Similar results were found when comparing data from whole-tree and stem-only harvested plots with uncut control plots (Table 13). However, individual observations of increases and decreases in CEC have been documented following cutting. For example, there is a short- and long-term tendency of decreased CEC-concentrations in the forest floor and increased CEC in the mineral soil, regardless of harvest method (Table 9). These results are similar to the findings for soil C and soil N. But, when comparing whole-tree harvested plots with control plots short- to medium-term decreases have also been observed in the mineral soil (Table 13). Treatment differences between CH and CTR are largely missing why no conclusions can be drawn whether or not this is also true between stem-only and unharvested control plots (Table 13).

When CEC is expressed as exchangeable mass per hectare, statistically significant effects are rare and have only been observed in one case as a short-term increase in the forest floor after whole-tree harvest compared with pre-harvest conditions (Table 11). Comparisons with uncut control forests are missing (Table 15).

Short- and long-term differences in CEC as a function of harvest intensity (WTH↔CH) are usually non-significant but there are some reports of lower concentrations and pools of CEC following whole-tree harvest compared with stem-only, both in the forest floor and in the mineral soil (Table 13 and Table 15). Data describing short- and medium-term changes in CEC pools are missing.

7.2.3 Nitrogen (N)

Changes in soil N pools following clear-cutting are similar to those in soil C (i.e. primarily a decrease in the forest floor and an increase in the mineral soil) but there are also differences where soil N has been shown to increase in the forest floor and decrease in the mineral soil. Thus, the effect of clear-cutting on soil N is highly variable and appears to be site-specific.

Following clear-cutting, there have been reports of short-term decreases in soil N concentrations in the forest floor after WTH (n=3, Table 10 and Table 14) and short- and long-term decreases in soil N pools (Table 12 and Table 16) following both CH (n=5) and WTH (n=9). In addition, long-term increases in mineral soil N concentrations and pools have frequently been observed after WTH and CH (Table 10, Table 12, Table 14 and Table 16).

In contrast to this pattern, increased forest floor concentrations and pools have been reported after CH and WTH during the first five years and for more than 15 years. Also, short-term decreases in mineral soil concentrations (0-10 cm, n=3 and ≥10 cm, n=5) and pools (≥10 cm, n=4) are often observed following WTH.

Short-and long-term treatment differences in total N concentrations between WTH and CH in the forest floor and mineral soil are usually non-significant (Table 14). Similarly, short- and long-term differences in mineral soil N pools are non-significant (Table 16). However, short-term decreases (n=6) in forest floor pools have been observed.

7.2.4 Base cations (Ca^{2+}, Mg^{2+} and K^+)

Short-term (≤5 years) treatment effects on base cation (Ca^{2+}, Mg^{2+} and K^+) concentrations and pools are well described in the literature, especially in the mineral soil, while medium- (6-14 years) to long-term (≥15 years) effects are less well known. To start with, short-term cutting effects on exchangeable base cation concentrations are described beginning with the forest floor before moving into the mineral soil. The results are compared with pre-harvest conditions (Table 9) and uncut controls (Table 13). Next, total concentrations are discussed but the results have not been included in Table 10 or Table 14 due to too few data. Changes in soil exchangeable pools are described in Table 11 and Table 15. Finally, treatment differences between whole-tree and stem-only harvest are presented in terms of concentrations and pools.

There is no clear pattern regarding the short-term treatment effects on exchangeable base cation concentrations in the forest floor after cutting (WTH and CH) compared with pre-harvest conditions (Table 9) or uncut control plots (CTR) (Table 13). There are records of increases, decreases or no treatment effects at all. In contrast, and based on few observations, short-term increases of the exchangeable BC concentrations have been observed in the mineral soil (0-10 and ≥10 cm) after stem-only harvest compared with pre-harvest values (Table 9) and uncut controls (Table 13). Treatment effects of whole-tree harvesting in the mineral soil (0-10 and ≥10 cm) are frequently shown as decreasing exchangeable concentrations of Mg^{2+} and K^+, while Ca^{2+} shows no change or increased exchangeable concentrations during the first five years.

A chronosequence study by Katzensteiner (2003) is the only investigation reporting treatment effects on the total BC concentrations compared with pre-harvest conditions. Higher total concentrations of Ca, Mg and K were found in the forest floor of a regenerating stand (10 years old) compared with a conventional clear-cut stand (0 year). The lowest total K concentration was found in the clear-cut, attributed to harvest losses. Belleau et al. (2006) did not find any differences between stem-only and control plots (CH and CTR) in the forest floor or mineral soil during the first two years following cutting, except for a small decrease in total K concentration (0.1 mg per gram) in the forest floor, an effect which was gone the following year. Simard et al. (2001) discovered higher total Ca (1-1.8 mg per gram) and Mg (0.12 mg per gram) concentrations in the forest floor more than 15 years after stem-only and whole-tree harvest compared with control plots. No effect was seen on the total K concentrations. Finally, Hendrickson et al. (1989) measured significantly higher concentrations of total Ca in the forest floor (only WTH) and upper mineral soil (0-5 cm, WTH and CH) compared with a control forest, three years after harvest. No effects were found further down in the mineral soil (down to 20 cm). Increases in total Mg were observed in the forest floor but not the mineral soil after stem-only cutting. Whole-tree harvest did not lead to any significant Mg effects. Total K showed a decrease in the forest floor after whole-tree harvest and an increase in the mineral soil (10-20 cm) after stem-only harvest. No other soil layers showed any treatment effects.

In general, the short-term cutting effects on the exchangeable base cation pools in the forest floor and mineral soil are usually non-significant compared with pre-harvest conditions (Table 11). However, short-term significant decreases in the forest floor BC pools have been observed, especially after whole-tree harvest and it appears that the pools of exchangeable K^+ is most affected. Lower pools of exchangeable Ca^{2+}, Mg^{2+} and K^+ in the forest floor were also found up to 14 years after treatment, when comparing whole-tree harvested plots (but not stem-only harvested plots) with control plots (Table 15).

Treatment effects in the upper mineral soil (0-10 cm) are not well described in the literature (Table 11 and Table 15). Based on few observations, no short- or medium-term changes in exchangeable pools of Ca^{2+}, Mg^{2+} or K^+ have been found after WTH compared with pre-cutting data (Table 11). Long-term data and comparisons between WTH and control plots are missing (Table 15). Similar results were found following CH with the exception of a medium- to long-term increase in exchangeable pools of Mg^{2+} compared with pre-harvest conditions (Table 11).

Short- and long-term treatment effects on exchangeable BC pools deeper down in the mineral soil (≥10 cm) are usually non-significant compared with pre-harvest conditions (Table 11) and control plots (Table 15) regardless of harvest method. However, short- and medium-term increases in exchangeable K^+ pools have been registered following stem-only compared with pre-harvest conditions (Table 11). The effect of whole-tree harvest is not as clear. Similar to stem-only harvest, there have been records of increased or higher pools of exchangeable K^+ compared with pre-harvest conditions (Table 11) and control forests (Table 15). In contrast, lower BC pools have also been registered following whole-tree harvest during the first 15 years. The effect is most notably for the divalent ions (Ca^{2+} and Mg^{2+}) but can no longer be detectable after 15 years (Table 11 and Table 15).

Significant short-term (≤5 years) decreases in total pools of base cations have been observed in the forest floor and mineral soil (≤ 60 cm) after whole-tree harvest compared with pre-cutting data (Ranger et al., 2008). In contrast, Nykvist (1977a) found that the total Ca-pools in the forest floor increased regardless of harvest method, while total K-pools generally decreased. The effects on total pools of Mg were not conclusive. The Mg pools decreased at one site and increased at another one. Medium- to long-term effects were not described. Simard et al. (2001) reported post treatment differences with higher total pools of Ca (+270 kg per hectare) and Mg (+35 kg per hectare) (but not K) in the forest floor, five years after stem-only harvest compared with a control forest in St. Pére, Canada. At two other sites (Dieppe and Mathieu), higher total pools of Ca (but not Mg and K), varying between 280-500 kg per hectare, were found in whole-tree harvested plots compared with control plots up to 22 years after cutting.

There are few significant short- or long-term differences in exchangeable base cation concentrations between different harvest intensities (WTH and CH) in the forest floor or in the mineral soil (0-10 and ≥10 cm). Lower exchangeable Ca^{2+}, Mg^{2+} and K^+ concentrations have been observed both in the forest floor and mineral soil lasting more than fifteen years after WTH compared with CH (Table 13). Lower soil exchangeable pools of Ca^{2+}, Mg^{2+} and K^+ have been found as well (Table 15). For example, lower pools of K^+ have frequently been observed in the forest floor (n=7) and mineral soil (0-10 cm, n=2 and ≥10 cm, n=3) during the first five years after harvest. Sometimes, this difference can still be observed after 15 years. However, at most sites the differences are non-significant.

Treatment differences in total soil BC pools have been observed in a few studies. Wall (2008) measured lower pools of Ca (-387 to -64 kg per hectare), Mg (-35 to -6 kg per hectare) and K (-34 to -6 kg per hectare) in the forest floor (litter layer) after WTH compared with CH, four years after harvest. No differences were found in the humus layer (F+H) or mineral soil (0-10 cm). Nykvist (1977a) also reported lower total BC pools in the forest floor. During the first year after harvest the pools of Ca, Mg and K were lower following WTH compared with CH ranging between -213 to -207 (Ca), -25 to -16 (Mg) and -52 to -32 (K) kg per hectare. During the fourth year the differences were less; -150 to -37 (Ca), -7 to -1 (Mg) and -13 to 0 (K) kg per hectare. In general, the largest losses seem to occur for Ca.

The number of observations describing medium- to long-term treatment effects is limited. Single studies indicate either positive or negative trends and no clear patterns emerge.

Table 9. The number of observations* (n=tot, nd=no data) showing increasing (n=pos), decreasing (n=neg) or no (n=0) trend in **soil exchangeable concentrations** (cmol$_c$ per kg = meq 100 per gram) following clear-cutting. CH = conventional harvest, WTH = whole tree harvest. **Pre- and post-treatment differences**.

Comparison	**Variable**	**Soil depth (cm)**	**Short-term (≤5)**				**Medium-term (6-14)**				**Long-term (≥15)**			
			n=tot	**n=pos**	**n=neg**	**n=0**	**n=tot**	**n=pos**	**n=neg**	**n=0**	**n=tot**	**n=pos**	**n=neg**	**n=0**
CH	pH	forest floor	nd				2	1		1	2			2
WTH	pH	forest floor	1			1	1			1	1			1
CH	pH	0-10	1			1	1			1	1			1
WTH	pH	0-10	4	1	1	2	2			2	nd			
CH	pH	≥10	3			3	3			3	3			3
WTH	pH	≥10	8	4	1	3	4	3		1	1	1		
CH	CEC	forest floor	nd				1	1			1		1	
WTH	CEC	forest floor	1			1	1			1	1			1
CH	CEC	0-10	nd				nd				2	1		1
WTH	CEC	0-10	4		1	3	2			2	2	2		
CH	CEC	≥10	nd				nd				1			1
WTH	CEC	≥10	8	2	1	5	4	3		1	1	1		1
CH	Ca^{2+}	forest floor	nd				1	1			1			1
WTH	Ca^{2+}	forest floor	4	3	1		1			1	1			1
CH	Ca^{2+}	0-10	1	1			1			1	3	2		1
WTH	Ca^{2+}	0-10	4			4	1			1	2			2
CH	Ca^{2+}	≥10	1			1	1			1	2			2
WTH	Ca^{2+}	≥10	8	2	1	5	3	2		1	2	1		2
CH	Mg^{2+}	forest floor	nd				1		1		1		1	
WTH	Mg^{2+}	forest floor	4	3	1		1			1	1			1
CH	Mg^{2+}	0-10	1	1			1			1	3	2		1
WTH	Mg^{2+}	0-10	4	1	2	1	1			1	2			2
CH	Mg^{2+}	≥10	1	1			1			1	2	1		1
WTH	Mg^{2+}	≥10	8	1	4	3	3	2		1	2			2
CH	K^+	forest floor	nd				1		1		1			1
WTH	K^+	forest floor	4	3	1		1			1	1			1
CH	K^+	0-10	1	1			1			1	2	1		1
WTH	K^+	0-10	7	1	4	2	1			1	2			2
CH	K^+	≥10	1	1			1	1			2	1		1
WTH	K^+	≥10	17	1	13	3	3	2		1	2	1		1

*Katzensteiner (2003), McLaughlin & Phillips (2006), Mroz et al. (1985), Johnson et al. (1991b), Johnson et al. (1997), Johnson & Todd (1998), Knoepp & Swank (1997), Ranger et al. (2008) and Snyder & Harter (1984).

Table 10. The number of observations* (n=tot, nd=no data) showing increasing (n=pos), decreasing (n=neg) or no (n=0) trend in **soil total concentrations** (mg per gram) following clear-cutting. CH = conventional harvest, WTH = whole tree harvest. **Pre- and post treatment differences**.

Comparison	**Variable**	**Soil depth (cm)**	**Short-term (≤5)**				**Medium-term (6-14)**				**Long-term (≥15)**			
			n=tot	**n=pos**	**n=neg**	**n=0**	**n=tot**	**n=pos**	**n=neg**	**n=0**	**n=tot**	**n=pos**	**n=neg**	**n=0**
CH	C	forest floor	nd				2			2	2			2
WTH	C	forest floor	nd				nd				1			1
CH	C	0-10	nd				nd				2	1		1
WTH	C	0-10	nd				nd				2	1		1
CH	C	≥10	nd				nd				1			1
WTH	C	≥10	nd				nd				2			2
CH	N	forest floor	nd				2			2	2			2
WTH	N	forest floor	3	1	2		nd				1			1
CH	N	0-10	nd				nd				2	2		
WTH	N	0-10	3		3		nd				2	1		1
CH	N	≥10	nd				nd				1	1		
WTH	N	≥10	9	3	4	1	nd				2	1		1

*Johnson & Todd (1998), Mroz et al. (1985), McLaughlin & Phillips (2006) and Katzensteiner (2003).

Table 11. The number of observations* (n=tot, nd=no data) showing increasing (n=pos), decreasing (n=neg) or no (n=0) trend in **soil exchangeable pools** (kg per hectare) following clear-cutting. CH = conventional harvest, WTH = whole tree harvest. **Pre- and post-treatment differences**. In some cases flux-data is missing entirely. Consequently, these rows have been excluded from the table.

Comparison	**Variable**	**Soil depth (cm)**	**Short-term (≤5)**				**Medium-term (6-14)**				**Long-term (≥15)**			
			n=tot	**n=pos**	**n=neg**	**n=0**	**n=tot**	**n=pos**	**n=neg**	**n=0**	**n=tot**	**n=pos**	**n=neg**	**n=0**
WTH	H^+	forest floor	2			2	2			2	nd			
WTH	H^+	0-10	1			1	1			1	nd			
WTH	H^+	≥10	4			4	4			4	nd			
WTH	CEC	forest floor	2	1		1	2			2	nd			
WTH	CEC	0-10	1			1	1			1	nd			
WTH	CEC	≥10	4			4	4			4	nd			
WTH	Ca^{2+}	forest floor	5	2	1	2	2			2	1			1
CH	Ca^{2+}	0-10	1			1	1			1	1			1
WTH	Ca^{2+}	0-10	1			1	1			1	nd			
CH	Ca^{2+}	≥10	3			3	3			3	3			3
WTH	Ca^{2+}	≥10	7		3	4	4			4	1	1		
WTH	Mg^{2+}	forest floor	5		1	4	2			2	1		1	
CH	Mg^{2+}	0-10	1			1	1	1			1	1		
WTH	Mg^{2+}	0-10	1			1	1			1	nd			
CH	Mg^{2+}	≥10	3			3	3			3	3			3
WTH	Mg^{2+}	≥10	4		2	5	4			4	1			1
CH	K^+	forest floor	2		2		nd				nd			
WTH	K^+	forest floor	10	1	3	6	5		1	4	nd			
CH	K^+	0-10	1			1	1			1	1			1
WTH	K^+	0-10	1			1	1			1	nd			
CH	K^+	≥10	5	1	1	3	3	3			3			3
WTH	K^+	≥10	12	4	5	3	7	1		6	nd			

* Mroz et al. (1985), Johnson et al. (1997), Romanowicz et al. (1996), McLaughlin & Phillips (2006), Nykvist (1977b) and Snyder & Harter (1984).

Table 12. Number of observations* (n=tot, nd=no data) showing increasing (n=pos), decreasing (n=neg) or no (n=0) trend **in soil total pools** (kg per hectare) following clear-cutting. CH = conventional harvest, WTH = whole tree harvest. **Pre- and post-treatment differences**.

Comparison	**Variable**	**Soil depth (cm)**	**Short-term (≤5)**				**Medium-term (6-14)**				**Long-term (≥15)**			
			n=tot	**n=pos**	**n=neg**	**n=0**	**n=tot**	**n=pos**	**n=neg**	**n=0**	**n=tot**	**n=pos**	**n=neg**	**n=0**
CH	C	forest floor	nd				nd				4		4	
WTH	C	forest floor	1		1		nd				5		4	1
CH	C	≥10	nd				nd				4	3		1
WTH	C	≥10	1	1			nd				5	3		2
CH	N	forest floor	2	1	1		nd				4		4	
WTH	N	forest floor	6	1	5		nd				5	1	4	
CH	N	0-10	1			1	nd				nd			
WTH	N	0-10	1			1	nd				nd			
CH	N	≥10	nd				nd				4	2		2
WTH	N	≥10	4		4		nd				4	2		2

*Mroz et al. (1985), McLaughlin & Phillips (2006), Ranger et al. (2008), Smethurst & Nambiar (1990), Olsson et al. (1996b) and Nykvist (1977a).

Table 13. Number of observations* (n=tot, nd=no data) showing higher (n=pos), lower (n=neg) or no (n=0) trend in **soil exchangeable concentrations** following clear-cutting. CH=conventional harvest, WTH=whole tree harvest, CTR=control. **Post-treatment differences**. In some cases concentration data is missing entirely. Consequently, these rows have been excluded from the table.

Comparison	Variable	Soil depth (cm)	Short-term (≤5)				Medium-term (6-14)				Long-term (≥15)			
			n=tot	n=pos	n=neg	n=0	n=tot	n=pos	n=neg	n=0	n=tot	n=pos	n=neg	n=0
CH↔CTR	pH	forest floor	3	2		1	nd				nd			
WTH↔CTR	pH	forest floor	3	1	1	1	3	1	1	1	3	1		2
WTH↔CH	pH	forest floor	7			7	12		8	4	21		4	16
CH↔CTR	pH	0-10	4	2		2	nd				nd			
WTH↔CTR	pH	0-10	4	2	1	1	3		1	2	2			2
WTH↔CH	pH	0-10	5			5	7			7	18			18
CH↔CTR	pH	≥10	1			1	nd				nd			
WTH↔CTR	pH	≥10	3		2	1	2		2		1	1		
WTH↔CH	pH	≥10	2			2	nd				17			17
CH↔CTR	CEC	forest floor	1	1			nd				nd			
WTH↔CTR	CEC	forest floor	2		1	1	2		1	1	1			1
WTH↔CH	CEC	forest floor	2			2	1			1	13		2	11
CH↔CTR	CEC	0-10	1			1	nd				nd			
WTH↔CTR	CEC	0-10	2		1	1	2		1	1	nd			
WTH↔CH	CEC	0-10	1			1	nd				10			10
WTH↔CTR	CEC	≥10	2		1	1	2		1	1	1	1		
WTH↔CH	CEC	≥10	1			1	nd				8		3	5
CH↔CTR	Ca^{2+}	forest floor	2	2			nd				nd			
WTH↔CTR	Ca^{2+}	forest floor	2		1	1	3	1	1	1	3	2		1
WTH↔CH	Ca^{2+}	forest floor	2			2	1		1		4		4	
CH↔CTR	Ca^{2+}	0-10	2	1		1	nd				nd			
WTH↔CTR	Ca^{2+}	0-10	2		1	1	3		1	2	2			2
WTH↔CH	Ca^{2+}	0-10	1			1	nd				2		2	
WTH↔CTR	Ca^{2+}	≥10	2		1	1	2		1	1	1	1		
WTH↔CH	Ca^{2+}	≥10	1			1	nd				4			4
CH↔CTR	Mg^{2+}	forest floor	2	1		1	nd				nd			
WTH↔CTR	Mg^{2+}	forest floor	2		1	1	3	1	1	1	3	1	1	1
WTH↔CH	Mg^{2+}	forest floor	2		2		1		1		4		1	3
CH↔CTR	Mg^{2+}	0-10	2	1		1	nd				nd			
WTH↔CTR	Mg^{2+}	0-10	2		1	1	3	1	1	1	2	1		1
WTH↔CH	Mg^{2+}	0-10	1			1	nd				2		1	1
WTH↔CTR	Mg^{2+}	≥10	2		1	1	2		1	1	1	1		
WTH↔CH	Mg^{2+}	≥10	1			1	nd				4			4
CH↔CTR	K^{+}	forest floor	2			2	nd				nd			
WTH↔CTR	K^{+}	forest floor	2			2	3			3	3		2	1
WTH↔CH	K^{+}	forest floor	2			2	1			1	4			4
CH↔CTR	K^{+}	0-10	2	1		1	nd				nd			
WTH↔CTR	K^{+}	0-10	2			2	3			3	2			2
WTH↔CH	K^{+}	0-10	1			1	nd				2		1	1
WTH↔CTR	K^{+}	≥10	2			2	2			2	1			1
WTH↔CH	K^{+}	≥10	1		1		nd				4		2	2

*Wall (2008), Hendrickson et al. (1989), Johnson & Todd (1998), McLaughlin & Phillips (2006), Simard et al. (2001), Bélanger et al. (2003), Thiffault et al. (2006), Olsson et al. (1996b), Olsson et al. (2004), Staaf & Olsson (1991), Nykvist & Rosén (1985), Brais et al. (1995) and Belleau et al. (2006).

Table 14. Number of observations* (n=tot, nd=no data) showing higher (n=pos), lower (n=neg) or no (n=0) trend in **soil total concentrations** following clear-cutting. CH = conventional harvest, WTH = whole tree harvest, CTR = control. **Post-treatment differences**. In some cases concentration data is missing entirely. Consequently, these rows have been excluded from the table.

Comparison	**Variable**	**Soil depth (cm)**	**Short-term (≤5)**				**Medium-term (6-14)**				**Long-term (≥15)**			
			n=tot	**n=pos**	**n=neg**	**n=0**	**n=tot**	**n=pos**	**n=neg**	**n=0**	**n=tot**	**n=pos**	**n=neg**	**n=0**
WTH↔CTR	C	forest floor	nd				nd				1		1	
WTH↔CH	C	forest floor	nd				nd				2			2
WTH↔CTR	C	0-10	nd				nd				1			1
WTH↔CH	C	0-10	nd				nd				1			1
CH↔CTR	N	forest floor	3	1		2	nd				nd			
WTH↔CTR	N	forest floor	1		1		1			1	3	1		2
WTH↔CH	N	forest floor	2	1		1	nd				nd			
CH↔CTR	N	0-10	3			3	nd				nd			
WTH↔CTR	N	0-10	2			2	nd				nd			
WTH↔CH	N	0-10	3			3	nd				2			2
CH↔CTR	N	≥10	1			1	nd				nd			
WTH↔CTR	N	≥10	1			1	nd				1			1
WTH↔CH	N	≥10	1			1	nd				1			1

*Johnson & Todd (1998), McLaughlin & Phillips (2006), Simard et al. (2001), Hendrickson et al. (1989) and Belleau et al. (2006).

Table 15. Number of observations* (n=tot, nd=no data) showing higher (n=pos), lower (n=neg) or no (n=0) trend in **soil exchangeable pools** (kmol$_c$ per hectare) following clear-cutting. CH = conventional harvest, WTH = whole tree harvest, CTR = control. **Post-treatment differences**. In some cases flux-data is missing entirely. Consequently, these rows have been excluded from the table.

Comparison	**Variable**	**Soil depth (cm)**	**Short-term (≤5)**				**Medium-term (6-14)**				**Long-term (≥15)**			
			n=tot	**n=pos**	**n=neg**	**n=0**	**n=tot**	**n=pos**	**n=neg**	**n=0**	**n=tot**	**n=pos**	**n=neg**	**n=0**
WTH↔CH	H^+	forest floor	nd				nd				8	3		5
WTH↔CH	H^+	0-10	nd				nd				8			8
WTH↔CH	H^+	≥10	nd				nd				4			4
WTH↔CH	CEC	forest floor	nd				nd				8		2	6
WTH↔CH	CEC	0-10	nd				nd				8			8
WTH↔CH	CEC	≥10	nd				nd				4			4
CH↔CTR	Ca^{2+}	forest floor	1			1	nd				nd			
WTH↔CTR	Ca^{2+}	forest floor	2		2		2		2		1			1
WTH↔CH	Ca^{2+}	forest floor	5		2	3	nd				8		3	5
CH↔CTR	Ca^{2+}	0-10	1			1	nd				nd			
WTH↔CH	Ca^{2+}	0-10	3			3	nd				8		3	5
WTH↔CTR	Ca^{2+}	≥10	4		2	2	4		2	2	1	1		
WTH↔CH	Ca^{2+}	≥10	nd				nd				4		1	3
CH↔CTR	Mg^{2+}	forest floor	1			1	nd				nd			
WTH↔CTR	Mg^{2+}	forest floor	2		1	1	2		1	1	1			1
WTH↔CH	Mg^{2+}	forest floor	5		2	3	nd				8	1	1	6
CH↔CTR	Mg^{2+}	0-10	1			1	nd				nd			
WTH↔CH	Mg^{2+}	0-10	3			3	nd				8		2	6
CH↔CTR	Mg^{2+}	≥10	4		2	2	4		2	2	nd			
WTH↔CTR	Mg^{2+}	≥10	nd				nd				1			1
WTH↔CH	Mg^{2+}	≥10	nd				nd				4		1	3
CH↔CTR	K^+	forest floor	1			1	nd				nd			
WTH↔CTR	K^+	forest floor	2		1	1	2		1	1	nd			
WTH↔CH	K^+	forest floor	9	1	7	1	2		2		10	1	3	6
CH↔CTR	K^+	0-10	1			1	nd				nd			
WTH↔CH	K^+	0-10	3		2	1	nd				8			8
WTH↔CTR	K^+	≥10	4	1		3	4	1		3	nd			
WTH↔CH	K^+	≥10	4	1	3		2	1	1		6	1	1	6

***** McLaughlin & Phillips (2006), Wall (2008), Olsson et al. (1996b), Nykvist (1977b), Nykvist (1990), Brais et al. (1995), and Belleau et al. (2006).

Table 16. Number of observations* (n=tot, nd=no data) showing higher (n=pos), lower (n=neg) or no (n=0) trend in **soil total pools** following clear-cutting. CH = conventional harvest, WTH = whole tree harvest, CTR = control. **Post-treatment differences**. In some cases flux-data is missing entirely. Consequently, these rows have been excluded from the table.

Comparison	**Variable**	**Soil depth (cm)**	**Short-term (≤5)**				**Medium-term (6-14)**				**Long-term (≥15)**			
			n=tot	**n=pos**	**n=neg**	**n=0**	**n=tot**	**n=pos**	**n=neg**	**n=0**	**n=tot**	**n=pos**	**n=neg**	**n=0**
WTH↔CTR	C	forest floor	nd				nd				1			1
WTH↔CH	C	forest floor	nd				nd				10	1		9
WTH↔CH	C	0-10	nd				nd				8			8
WTH↔CTR	C	≥10	nd				nd				1			1
WTH↔CH	C	≥10	nd				nd				15	1		14
CH↔CTR	N	forest floor	1			1	nd				nd			
WTH↔CTR	N	forest floor	nd				1			1	3	1		2
WTH↔CH	N	forest floor	8		6	2	nd				8	1	1	6
WTH↔CH	N	0-10	2			2	nd				8			8
WTH↔CTR	N	≥10	nd				nd				1			1
WTH↔CH	N	≥10	nd				nd				12			12

*Wall (2008), Simard et al. (2001), Olsson et al. (1996a), Olsson et al. (2004), Nykvist (1977a), Johnson et al. (2002) and McLaughlin & Phillips (2006).

7.3 Soil water, groundwater and stream water

In this chapter, short- and long-term clear-cutting effects on soil water, groundwater and stream water are compared with pre-harvest conditions and uncut control forests after two different harvest intensities (CH and WTH). The results are presented in four different tables with a distinction made between *concentrations* (Table 17 and Table 19) and *fluxes* (Table 18 and Table 20).

7.3.1 pH

There are few studies describing changes in pH after harvest compared with pre-harvest conditions. Regarding groundwater and stream water, observations after whole-tree harvest is missing entirely (Table 17). In addition, the medium- to long-term effects are largely unknown while the short-term (≤5 years) effects are better described.

In general, the short-term (≤5 years) cutting effects of stem-only harvest on forest floor soil water pH is non-significant in contrast to the mineral soil where decreased pH frequently has been reported in soil water (0-50 cm and ≥50 cm), groundwater and stream water after stem-only harvest compared with pre-harvest conditions (Table 17). In contrast, following whole-tree harvest, soil water pH has been shown to increase during the first five years (Table 17). Ground- and stream water data describing treatment effects of whole-tree harvest is missing (Table 17).

Short- and medium-term decreases in soil water, groundwater and stream water pH are also observed when clear-cut areas (CH and WTH) are compared with uncut control plots (Table 19). Occasionally, statistically higher pH-values have been registered in soil water and stream water in harvested plots (CH and WTH) during the first five years (Table 19). Based on a few observations, the increase in pH may persist for more than 15 years following whole-tree harvest.

Short-term differences in harvest intensity (WTH and CH) are only available for soil water data (Table 19) Long-term data is missing entirely. Based on the data, no clear pattern is observed.

Studies comparing hydrogen ion fluxes with pre-harvest conditions are only available for stream water and based on a single Swedish study by Rosén et al. (1996) (Table 18). Following stem-only harvest, the H^+ flux did not change but decreased after whole-tree harvest. Compared with uncut controls, short-term increases in stream water hydrogen fluxes have been found following stem-only harvest (Table 20). Compared with stem-only harvest, the flux of H+ seems to be lower after whole-tree harvest (Table 20).

7.3.2 Base cations (Ca^{2+}, Mg^{2+} and K^+)

Short-term treatment effects on base cation concentrations and fluxes in soil water, and stream water are well described in the literature while groundwater data is largely lacking. In general, the concentration of base cations (Ca^{2+}, Mg^{2+} and K^+) increases following clear-cutting (whole-tree or stem-only) compared with pre-harvest conditions and non-harvested controls (Table 17 and Table 19). These increases are usually observed both in the soil water (throughout the soil profile), groundwater and stream water linked to mineralization and increased anion mobility, mostly NO_3^-. However, decreasing concentrations are frequently observed in soil water and groundwater indicating retention of these ions. In contrast, decreasing concentrations are seldom observed in stream water.

Data describing post-treatment concentration differences between harvest intensities (WTH and CH) are only available for soil water data. Lower concentrations of base cations, especially K^+, following WTH compared with CH have been registered for more than 15 years (Table 19). This often coincides with lower concentrations of NO_3^- (chapter 7.3.3). However, there are also reports of short-term higher base cation leaching in soil water following whole-tree harvest compared with stem-only, both in the forest floor and mineral soil (≥50 cm, Table 19).

Soil water and stream water fluxes of base cations usually increase after cutting due to increased runoff (Table 18 and Table 20). These increases occur regardless of harvest intensity and may last more than 15 years according to stream water data (Table 20).

7.3.3 Anions (NO_3^-, SO_4^{2-}, Cl^- and DOC)

There is a large number of observations in soil water, groundwater and stream water of cutting effects on NO_3^-- concentrations and fluxes following clear- felling. In general, the NO_3^- concentrations increase compared with pre-harvest conditions and uncut control forests (Table 17 and Table 19). These increases are of short duration and have largely disappeared within five years although there are a few reports of medium- to long-term leaching to groundwater and stream water. In contrast to these results, clear-cutting has also been shown to cause decreased NO_3^- concentrations during the first five years, regardless of harvest methods (Table 17 and Table 19). Generally, these decreases take place in soil water, both in the forest floor and the mineral soil (≤50 cm and ≥50 cm). Medium-term decreases are rare and there are no reports of long-term decreases (Table 17 and Tab le 19). In accordance to the observed NO_3^-concentration trends, short-term increases and decreases in NO_3^-fluxes have been observed in soil water and stream water following harvest (Table 18 and Table 20).

The data also show that the effect of cutting is more important for NO_3^--concentrations than differences in harvest intensity (WTH and CH). However, whole-tree harvesting does not always show the same magnitude of increase as stem-only harvest. Compared with stem-only, whole-tree harvest exhibited lower NO_3^- concentrations in the forest floor (n=3) and mineral soil (n=6) (Table 19). No treatment differences have been observed in groundwater while stream water studies are lacking. There are reports of lower fluxes from the forest floor soil water (n=2) within 5 years after whole-tree harvest compared with stem-only, but in general there are no statistical significant differences (Figure 20).

Following clear-cutting, there is generally a short-term decrease in SO_4^{2-}-concentration in soil water, groundwater and stream water, regardless of harvest method (Table 17 and Table 19). Short-term increases in SO_4^{2-}-concentration have occasionally been found in soil water and stream water after whole-tree harvest compared with pre-harvest conditions and control forest. Based on a few observations, the medium- to long-term effects appear to be similar to the short-term (i.e. decreasing or lower SO_4^{2-}-concentrations). Treatment differences between WTH and CH indicate lower SO_4^{2-}-concentration following whole-tree harvest compared with stem-only (Table 19). In terms of SO_4^{2-} fluxes, they generally increase after cutting (Table 18 and Table 20).

Based on a few observations, the concentrations of Cl^- seem to decrease in soil water following stem-only and whole-tree harvest, while the trends in groundwater and stream water are more variable showing either up- or downward trends (Table 17 and Table 19). The long-term effects are poorly studied, although there are some reports of long-term Cl^- concentration decreases (Table 19). Generally, the soil- and stream water Cl^--fluxes increase following harvest (Table 18 and Table 20). The treatment effects between WTH and CH on the Cl^- concentrations and fluxes are not well described in the literature. However, based on a few observations it appears that the differences are small and insignificant (Table 19 and Table 20).

Following stem-only harvest, the concentrations of DOC increase in soil water, groundwater and stream water during the first five years compared with pre-harvest data and uncut controls (Table 17 and Table 19). Similar results were found when comparing whole-tree harvested areas with uncut mature forests but not with the pre-harvest conditions (Table 17 and Table 19). Instead, there are a few reports of short-and long-term decreases in DOC-concentrations after WTH compared with the pre-cutting data (Table 17 and Tab le 19). Medium- to long-term harvest effects are largely unknown, but there are a few observations indicating increased concentrations, which may last up to fourteen years (Table 17 and Table 19). Based on few observations, the DOC fluxes in soil water (forest floor, 0-50 and ≥50 cm) and stream water appear to increase after stem-only harvest compared with uncut controls (Table 20). Too few data are available for making meaningful comparisons with pre-harvest conditions (Table 18).

Comparisons between WTH and CH are seldom reported but according to a soil water (≥50 cm) study by Zetterberg et al. (2008), lower concentrations of DOC were found in the WTH plots after more than 15 years. However, the differences are generally non-significant (Table 19). There are too few observations to draw any conclusions on DOC flux differences between WTH and CH (Table 20).

Table 17. Number of observations* (n=tot, nd=no data) showing increasing (n=pos), decreasing (n=neg) or no (n=0) trend in soil water, groundwater and stream water **concentration** following clear-cutting. CH = conventional harvest, WTH = whole tree harvest. **Pre- and post-treatment differences**. In some cases concentration data is missing entirely. Consequently, these rows have been excluded from the table.

	Comparison	**Variable**	**Soil depth (cm)**	**Short-term (≤5)**				**Medium-term (6-14)**				**Long-term (≥15)**			
				n=tot	**n=pos**	**n=neg**	**n=0**	**n=tot**	**n=pos**	**n=neg**	**n=0**	**n=tot**	**n=pos**	**n=neg**	**n=0**
Soil water	CH	pH	forest floor	1			1	1	1			nd			
Soil water	CH	pH	0-50	2		2		nd				nd			
Soil water	WTH	pH	0-50	4	1		3	nd				nd			
Soil water	CH	pH	≥50	2		2		nd				nd			
Soil water	WTH	pH	≥50	4	2		2	nd				nd			
Groundwater	CH	pH	-	3		2	1	2		2		nd			
Stream water	CH	pH	-	4	1	2	1	1		1		nd			
Soil water	CH	DOC	forest floor	1	1			nd				nd			
Soil water	WTH	DOC	forest floor	1		1		nd				nd			
Soil water	CH	DOC	0-50	1	1			nd				nd			
Soil water	WTH	DOC	0-50	2		1	1	nd				nd			
Soil water	CH	DOC	≥50	1	1			nd				nd			
Soil water	WTH	DOC	≥50	2		2		nd				nd			
Groundwater	CH	DOC	-	3	2		1	2	2			nd			
Stream water	CH	DOC	-	11	11			1	1			nd			
Soil water	CH	Ca^{2+}	forest floor	1		1		nd				nd			
Soil water	WTH	Ca^{2+}	forest floor	1		1		nd				nd			
Soil water	CH	Ca^{2+}	0-50	3	2	1		nd				nd			
Soil water	WTH	Ca^{2+}	0-50	5	1	4		nd				nd			
Soil water	CH	Ca^{2+}	≥50	4	3	1		nd				nd			
Soil water	WTH	Ca^{2+}	≥50	11	5	4	2	nd				nd			
Groundwater	CH	Ca^{2+}	-	4	1	2	1	2		2		nd			
Stream water	CH	Ca^{2+}	-	9	5		4	1		1		nd			
Stream water	WTH	Ca^{2+}	-	5	4		1	nd				nd			
Soil water	CH	Mg^{2+}	forest floor	1		1		nd				nd			
Soil water	WTH	Mg^{2+}	forest floor	1		1		nd				nd			
Soil water	CH	Mg^{2+}	0-50	3		3		nd				nd			
Soil water	WTH	Mg^{2+}	0-50	4		3	1	nd				nd			
Soil water	CH	Mg^{2+}	≥50	3		3		nd				nd			
Soil water	WTH	Mg^{2+}	≥50	4		3	1	nd				nd			
Groundwater	CH	Mg^{2+}	-	4	2	1	1	2	1	1		nd			
Stream water	CH	Mg^{2+}	-	8	4		4	1		1		nd			
Stream water	WTH	Mg^{2+}	-	1			1	nd				nd			
Soil water	CH	K^{+}	forest floor	1	1			nd				nd			
Soil water	WTH	K^{+}	forest floor	1		1		nd				nd			
Soil water	CH	K^{+}	0-50	3	1	2		nd				nd			
Soil water	WTH	K^{+}	0-50	4		2	2	nd				nd			
Soil water	CH	K^{+}	≥50	4	2	2		nd				nd			
Soil water	WTH	K^{+}	≥50	9	2	5	2	nd				nd			
Groundwater	CH	K^{+}	-	4	2		2	2	2			nd			
Stream water	CH	K^{+}	-	8	4		4	1	1			nd			
Stream water	WTH	K^{+}	-	5	5			nd				nd			

Table 17. Continued.

	Comparison	Variable	Soil depth (cm)	Short-term (≤5)				Medium-term (6-14)				Long-term (≥15)			
				n=tot	n=pos	n=neg	n=0	n=tot	n=pos	n=neg	n=0	n=tot	n=pos	n=neg	n=0
Soil water	CH	SO_4^{2-}	forest floor	1		1		nd				nd			
Soil water	CH	SO_4^{2-}	0-50	1		1		nd				nd			
Soil water	WTH	SO_4^{2-}	0-50	4		2	2	nd				nd			
Soil water	CH	SO_4^{2-}	≥50	1		1		nd				nd			
Soil water	WTH	SO_4^{2-}	≥50	4	2	1	1	nd				nd			
Groundwater	CH	SO_4^{2-}	-	4		3	1	2		2		nd			
Stream water	CH	SO_4^{2-}	-	4		2	2	1		1		nd			
Stream water	WTH	SO_4^{2-}	-	1		1		nd				nd			
Soil water	CH	Cl^-	forest floor	1		1		nd				nd			
Soil water	WTH	Cl^-	forest floor	1			1	nd				nd			
Soil water	CH	Cl^-	0-50	1		1		nd				nd			
Soil water	CH	Cl^-	≥50	1		1		nd				nd			
Groundwater	CH	Cl^-	-	7	4	2	1	2		2		nd			
Stream water	CH	Cl^-	-	4	1	2	1	1		1		nd			
Soil water	CH	NO_3^-	forest floor	2	2			1		1		nd			
Soil water	WTH	NO_3^-	forest floor	1		1		nd				nd			
Soil water	CH	NO_3^-	0-50	3	3			nd				nd			
Soil water	WTH	NO_3^-	0-50	5	1	2	2	nd				nd			
Soil water	CH	NO_3^-	≥50	5	5			nd				nd			
Soil water	WTH	NO_3^-	≥50	13	7	4	2	nd				nd			
Groundwater	CH	NO_3^-	-	13	10	2	1	4	4			nd			
Stream water	CH	NO_3^-	-	11	9	1	1	1	1			nd			
Stream water	WTH	NO_3^-	-	5	4		1	nd				nd			

*__Soil water__: Örlander et al. (1997), Westling et al. (2004), Stevens & Hornung (1988), Briggs et al. (2000), Katzensteiner (2003), Neal et al. (1992), Ranger et al. (2007), Parfitt et al. (2002), Mann et al. (1988), Jewett et al. (1995), Robertson et al. (2000); Welsch et al. (2004) and Rothe & Mellert (2004). **Groundwater**: Neal et al. (2004), Mannerkoski et al. (2005), Parfitt et al. (2002), Tamm et al. (1974), Wiklander (1974), Wiklander (1983), Tamm & Wiklander (1972), Welsch et al. (2004) and Rusanen et al. (2004). **Stream water**: Neal et al. (2004), Ahtiainen & Huttunen (1999), Parfitt et al. (2002), Wiklander (1974), Likens et al. (1970), Mann et al. (1988), Tiedemann et al. (1988), Welsch et al. (2004), Wang et al. (2006) and Jewett et al. (1995).

Table 18. Number of observations* (n=tot, nd=no data) showing increasing (n=pos), decreasing (n=neg) or no (n=0) trend in soil water, groundwater and stream water **fluxes** following clear-cutting. CH = conventional harvest, WTH = whole tree harvest. **Pre- and post-treatment differences**. Groundwater fluxes are missing entirely compared with soil water and stream water fluxes which occasionally are missing for some variables and treatment. Consequently, these rows have been excluded from the table.

	Comparison	**Variable**	**Soil depth (cm)**	**Short-term (≤5)**				**Medium-term (6-14)**				**Long-term (≥15)**			
				n=tot	**n=pos**	**n=neg**	**n=0**	**n=tot**	**n=pos**	**n=neg**	**n=0**	**n=tot**	**n=pos**	**n=neg**	**n=0**
Stream water	CH	H^+	-	1			1	1			1	nd			
Stream water	WTH	H^+	-	1		1		1		1		nd			
Soil water	CH	DOC	forest floor	1		1		nd				nd			
Soil water	WTH	DOC	≥50	2	1	1		nd				nd			
Soil water	CH	Ca^{2+}	forest floor	1		1		nd				nd			
Soil water	WTH	Ca^{2+}	≥50	2	1	1		nd				nd			
Stream water	CH	Ca^{2+}	-	9	9			2	2			nd			
Stream water	WTH	Ca^{2+}	-	1	1			1	1			nd			
Soil water	CH	Mg^{2+}	forest floor	1		1		nd				nd			
Soil water	WTH	Mg^{2+}	≥50	2	1	1		nd				nd			
Stream water	CH	Mg^{2+}	-	9	9			2	2			nd			
Stream water	WTH	Mg^{2+}	-	1	1			1	1			nd			
Soil water	CH	K^+	forest floor	1		1		nd				nd			
Soil water	WTH	K^+	≥50	2	2			nd				nd			
Stream water	CH	K^+	-	9	9			2	2			nd			
Stream water	WTH	K^+	-	1	1			1	1			nd			
Soil water	CH	SO_4^{2-}	forest floor	1		1		nd				nd			
Soil water	WTH	SO_4^{2-}	≥50	2	2			nd				nd			
Stream water	CH	SO_4^{2-}	-	9	9			2	2			nd			
Stream water	WTH	SO_4^{2-}	-	1	1			1	1			nd			
Soil water	WTH	Cl^-	≥50	2	2			nd				nd			
Stream water	CH	Cl^-	-	9	9			2	2			nd			
Stream water	WTH	Cl^-	-	1	1			1	1			nd			
Soil water	CH	NO_3^-	0-50	4	4			nd				nd			
Soil water	CH	NO_3^-	≥50	2	2			nd				nd			
Soil water	WTH	NO_3^-	≥50	2	1	1		nd				nd			
Stream water	CH	NO_3^-	-	11	11			2	2			nd			
Stream water	WTH	NO_3^-	-	1	1			1	1			nd			

***Soil water:** Stevens & Hornung (1988), Katzensteiner (2003) and Ranger et al. (2007). **Stream water:** Rosén et al. (1996). Ahtiainen (1992), Ahtiainen & Huttunen (1999), Likens et al. (1970), Swank et al. (2001) and Welsch et al. (2004).

Table 19. Number of observations* (n=tot, nd=no data) showing higher (n=pos), lower (n=neg) or no (n=0) trend in soil water, groundwater and stream water **concentrations** following clear-cutting. CH=conventional harvest, WTH=whole tree harvest, CTR=control. **Post-treatment differences**. In some cases concentration data is missing entirely. Consequently, these rows have been excluded from the table.

	Comparison	**Variable**	**Soil depth (cm)**	**Short-term (≤5)**				**Medium-term (6-14)**				**Long-term (≥15)**			
				n=tot	**n=pos**	**n=neg**	**n=0**	**n=tot**	**n=pos**	**n=neg**	**n=0**	**n=tot**	**n=pos**	**n=neg**	**n=0**
Soil water	CH↔CTR	pH	forest floor	3		1	2	nd				nd			
Soil water	WTH↔CTR	pH	forest floor	4		3	1	2		1	1	nd			
Soil water	WTH↔CH	pH	forest floor	2		1	1	nd				nd			
Soil water	CH↔CTR	pH	0-50	4	2		2	nd				nd			
Soil water	WTH↔CTR	pH	0-50	6	2	4		1		1		1	1		
Soil water	WTH↔CH	pH	0-50	1	1			nd				nd			
Soil water	CH↔CTR	pH	≥50	1		1		nd				nd			
Soil water	WTH↔CTR	pH	≥50	1	1			nd				nd			
Soil water	WTH↔CH	pH	≥50	2	1	1		nd				5		1	4
Groundwater	CH↔CTR	pH	-	2		1	1	1		1		nd			
Stream water	CH↔CTR	pH	-	20	3	9	8	1		1		nd			
Stream water	WTH↔CTR	pH	-	8		8		5	2	3		1	1		
Soil water	CH↔CTR	DOC	forest floor	3	3			nd				nd			
Soil water	WTH↔CTR	DOC	forest floor	3	2		1	1			1	nd			
Soil water	CH↔CTR	DOC	0-50	7	5		2	nd				nd			
Soil water	WTH↔CTR	DOC	0-50	6	4	1	1	2	1		1	1		1	
Soil water	CH↔CTR	DOC	≥50	1			1	nd				nd			
Soil water	WTH↔CH	DOC	≥50	nd				2		1	1	5		1	4
Groundwater	CH↔CTR	DOC	-	2	1		1	1	1			nd			
Stream water	CH↔CTR	DOC	-	6	3		3	1	1			nd			
Stream water	WTH↔CTR	DOC	-	3	3			3	3			nd			
Soil water	CH↔CTR	Ca^{2+}	forest floor	3	1		2	nd				nd			
Soil water	WTH↔CTR	Ca^{2+}	forest floor	4	3		1	2	1		1	nd			
Soil water	WTH↔CH	Ca^{2+}	forest floor	3	1	1	1	nd				nd			
Soil water	CH↔CTR	Ca^{2+}	0-50	4			4	nd				nd			
Soil water	WTH↔CTR	Ca^{2+}	0-50	7	7			2	2			1		1	
Soil water	CH↔CTR	Ca^{2+}	≥50	4	3		1	nd				nd			
Soil water	WTH↔CTR	Ca^{2+}	≥50	4	3	1		nd				nd			
Soil water	WTH↔CH	Ca^{2+}	≥50	5	1	3	1	2		2		5		3	2
Groundwater	CH↔CTR	Ca^{2+}	-	2		1	1	1		1		nd			
Stream water	CH↔CTR	Ca^{2+}	-	21	11	2	8	3	2	1		2	2		
Stream water	WTH↔CTR	Ca^{2+}	-	4	4			3	2		1	1	1		
Soil water	CH↔CTR	Mg^{2+}	forest floor	3	2		1	nd				nd			
Soil water	WTH↔CTR	Mg^{2+}	forest floor	4	3		1	2	1		1	nd			
Soil water	WTH↔CH	Mg^{2+}	forest floor	2			2	nd				nd			
Soil water	CH↔CTR	Mg^{2+}	0-50	4	3		1	nd				nd			
Soil water	WTH↔CTR	Mg^{2+}	0-50	7	6		1	2	2			1		1	
Soil water	CH↔CTR	Mg^{2+}	≥50	4	3		1	nd				nd			
Soil water	WTH↔CTR	Mg^{2+}	≥50	4	4			nd				nd			
Soil water	WTH↔CH	Mg^{2+}	≥50	4	2	1	1	2		1	1	5		2	3
Groundwater	CH↔CTR	Mg^{2+}	-	2		1	1	1		1		nd			
Stream water	CH↔CTR	Mg^{2+}	-	16	7	2	7	1		1		nd			
Stream water	WTH↔CTR	Mg^{2+}	-	3	2		1	1	1			1	1		

Table 19. Continued.

	Comparison	Variable	Soil depth (cm)	Short-term (≤5)				Medium-term (6-14)				Long-term (≥15)			
				n=tot	n=pos	n=neg	n=0	n=tot	n=pos	n=neg	n=0	n=tot	n=pos	n=neg	n=0
Soil water	CH↔CTR	K^+	forest floor	3	2		1	nd				nd			
Soil water	WTH↔CTR	K^+	forest floor	7	6		1	2	1		1	nd			
Soil water	WTH↔CH	K^+	forest floor	3	1	1	1	nd				nd			
Soil water	CH↔CTR	K^+	0-50	4	4			nd				nd			
Soil water	WTH↔CTR	K^+	0-50	14	12		2	10	9		1	1		1	
Soil water	WTH↔CH	K^+	0-50	1		1		nd				nd			
Soil water	CH↔CTR	K^+	≥50	4	4			nd				nd			
Soil water	WTH↔CTR	K^+	≥50	4	3	1		nd				nd			
Soil water	WTH↔CH	K^+	≥50	5	2	3		2		2		5		1	4
Groundwater	CH↔CTR	K^+	-	2	1		1	1	1			nd			
Stream water	CH↔CTR	K^+	-	22	21		1	2	2			1	1		
Stream water	WTH↔CTR	K^+	-	7	7			5	5			1			1
Soil water	CH↔CTR	SO_4^{2-}	forest floor	2		2		nd				nd			
Soil water	WTH↔CTR	SO_4^{2-}	forest floor	4	3		1	nd				nd			
Soil water	CH↔CTR	SO_4^{2-}	0-50	4		2	2	nd				nd			
Soil water	WTH↔CTR	SO_4^{2-}	0-50	8	2	3	3	4	1		3	1		1	
Soil water	WTH↔CH	SO_4^{2-}	≥50	nd				nd		1	1	5		3	2
Groundwater	CH↔CTR	SO_4^{2-}	-	2		1	1	1		1		nd			
Stream water	CH↔CTR	SO_4^{2-}	-	11		8	3	1		1		nd			
Stream water	WTH↔CTR	SO_4^{2-}	-	8	3	5		3	3			1		1	
Soil water	WTH↔CTR	Cl^-	forest floor	1		1		nd				nd			
Soil water	WTH↔CTR	Cl^-	0-50	1			1	nd				1		1	
Soil water	WTH↔CTR	Cl^-	≥50	nd				nd		1	1	nd			
Groundwater	CH↔CTR	Cl^-	-	3	2	1		1		1		nd			
Stream water	CH↔CTR	Cl^-	-	8	4	2	2	1		1		nd			
Stream water	WTH↔CTR	Cl^-	-	2	1		1	nd				1			1
Soil water	CH↔CTR	NO_3^-	forest floor	3	3			nd				nd			
Soil water	WTH↔CTR	NO_3^-	forest floor	5	4		1	3	3			nd			
Soil water	WTH↔CH	NO_3^-	forest floor	5		3	2	nd				nd			
Soil water	CH↔CTR	NO_3^-	0-50	4	4			nd				nd			
Soil water	WTH↔CTR	NO_3^-	0-50	11	11			1	1			1			1
Soil water	WTH↔CH	NO_3^-	0-50	5		2	3	nd				nd			
Soil water	CH↔CTR	NO_3^-	≥50	4	3	1		nd				nd			
Soil water	WTH↔CTR	NO_3^-	≥50	4	2	1	1	nd				nd			
Soil water	WTH↔CH	NO_3^-	≥50	6	1	4	1	2				nd			
Groundwater	CH↔CTR	NO_3^-	-	6	5		1	4	4			nd			
Groundwater	WTH↔CTR	NO_3^-	-	2	2			2	2			nd			
Groundwater	WTH↔CH	NO_3^-	-	2			2	2			2	nd			
Stream water	CH↔CTR	NO_3^-	-	29	26		3	4	4			2	2		
Stream water	WTH↔CTR	NO_3^-	-	8	8			2		1	1	1			1

***Soil water**: Staaf & Olsson (1994), Westling et al. (2004), Ring et al. (2001), Dahlgren & Driscoll (1994), Romanowicz et al. (1996), Hendrickson et al. (1989), Stevens et al. (1995), Tituts et al. (1997), Piirainen et al. (2002), Piirainen et al. (2004), McLaughlin & Phillips (2006), Wall (2008), Rosén & Lundmark-Thelin (1987), Hornbeck & Kropelin (1982), Högbom et al. (manuscript), Mitchell et al. (1989), McClurkin et al. (1987), Zetterberg et al., (2008), Katzensteiner (2003) and Qualls et al. (2000). **Groundwater:** Neal et al. (2004), Mannerkoski et al. (2005), Kubin (1988) and Wiklander (1974). **Stream water:** Bormann et al. (1968), Dahlgren & Driscoll (1994), Romanowicz et al. (1996), Neal et al. (1992), Neal et al. 2004, Stevens et al. (1995), Tremblay et al. (2009), Briggs et al. (2000), McLaughlin & Phillips (2006), Harr & Fredriksen (1988), Adamson et al (1987), Adamson & Hornung (1990), Martin et al. (2000), Hornbeck & Kropelin (1982), Mitchell et al. (1989), Lawrence et al. (1987), Martin & Pierce (1980), Patric & Aubertin (1977), Swank et al. (2001), Löfgren et al (2009b), Grip (1982) and Kubin (1995).

Table 20. Number of observations* (n=tot, nd=no data) showing higher (n=pos), lower (n=neg) or no (n=0) trend in soil water, groundwater and stream water **fluxes** following clear-cutting. CH=conventional harvest, WTH=whole tree harvest, CTR=control. **Post- treatment differences**. Groundwater fluxes are missing entirely compared with soil water and stream water fluxes which occasionally are missing for some variables and treatment. Consequently, these rows have been excluded from the table.

	Comparison	**Variable**	**Soil depth (cm)**	**Short-term (≤5)**				**Medium-term (6-14)**				**Long-term (≥15)**			
				n=tot	**n=pos**	**n=neg**	**n=0**	**n=tot**	**n=pos**	**n=neg**	**n=0**	**n=tot**	**n=pos**	**n=neg**	**n=0**
Soil water	WTH↔CH	H^+	forest floor	1		1		nd				nd			
Soil water	WTH↔CH	H^+	≥50	2		1	1	2		1	1	nd			
Stream water	CH↔CTR	H^+	-	1	1			nd				nd			
Soil water	CH↔CTR	DOC	forest floor	3	3			nd				nd			
Soil water	CH↔CTR	DOC	0-50	7	6		1	nd				nd			
Soil water	WTH↔CH	DOC	≥50	2		1	1	2		1	1	nd			
Soil water	CH↔CTR	Ca^{2+}	forest floor	5	2		3	nd				nd			
Soil water	WTH↔CH	Ca^{2+}	forest floor	3		2	1	nd				nd			
Soil water	CH↔CTR	Ca^{2+}	0-50	7	1	3	3	nd				nd			
Soil water	WTH↔CH	Ca^{2+}	≥50	2			2	2			2	nd			
Stream water	CH↔CTR	Ca^{2+}	-	5	3		2	1	1			nd			
Stream water	WTH↔CTR	Ca^{2+}	-	9	9			1	1			1	1		
Soil water	CH↔CTR	Mg^{2+}	forest floor	5	2		3	nd				nd			
Soil water	WTH↔CH	Mg^{2+}	forest floor	1			1	nd				nd			
Soil water	CH↔CTR	Mg^{2+}	0-50	7	1	3	3	nd				nd			
Soil water	WTH↔CH	Mg^{2+}	≥50	2			2	2			2	nd			
Stream water	CH↔CTR	Mg^{2+}	-	5	3		2	1	1			nd			
Stream water	WTH↔CTR	Mg^{2+}	-	9	9			1	1			1	1		
Soil water	CH↔CTR	K^+	forest floor	5	3		2	nd				nd			
Soil water	WTH↔CH	K^+	forest floor	3		2	1	nd				nd			
Soil water	CH↔CTR	K^+	0-50	7	2	3	2	nd				nd			
Soil water	WTH↔CH	K^+	≥50	2		1	1	2		1	1	nd			
Stream water	CH↔CTR	K^+	-	7	7			1	1			nd			
Stream water	WTH↔CTR	K^+	-	9	9			1	1			1			1
Soil water	CH↔CTR	SO_4^{2-}	forest floor	5	3		2	nd				nd			
Soil water	CH↔CTR	SO_4^{2-}	0-50	4	4			nd				nd			
Soil water	WTH↔CH	SO_4^{2-}	≥50	2			2	2			2	nd			
Stream water	CH↔CTR	SO_4^{2-}	-	5	4		1	1	1			nd			
Stream water	WTH↔CTR	SO_4^{2-}	-	9	8	1		1	1			1		1	
Soil water	WTH↔CH	Cl^-	≥50	2			2	2			2	nd			
Stream water	CH↔CTR	Cl^-	-	5	5			1	1			nd			
Stream water	WTH↔CTR	Cl^-	-	8	5	3		1	1			1			1
Soil water	CH↔CTR	NO_3^-	forest floor	5		2	3	nd				nd			
Soil water	WTH↔CH	NO_3^-	forest floor	4		2	2	nd				nd			
Soil water	CH↔CTR	NO_3^-	0-50	7	1		6	nd				nd			
Soil water	WTH↔CH	NO_3^-	≥50	2			2	2			2	nd			
Stream water	CH↔CTR	NO_3^-	-	7	7			1	1			nd			
Stream water	WTH↔CTR	NO_3^-	-	9	8	1		1	1			1			1

***Soil water:** Örlander et al. (1997), **Stevens** et al. (1995), Emmett et al. (1991), Piirainen et al. (2002), Piirainen et al. (2004), Wall (2008), Feller (2000) and Qualls et al. (2000). **Stream water:** Hornbeck et al. (1990), Stevens et al. (1995), McLaughlin & Phillips (2006), Adamson et al (1987), Adamson & Hornung (1990), Haveraaen (1981), Lawrence et al. (1987), Rosén et al. (1996) and Löfgren et al (2009b).

8 Discussion

8.1 Hydrology

Following harvest in final felling the total leaf area of the catchment is drastically reduced affecting abiotic factors such as light, temperature, moisture and wind. In addition, the dry deposition of nutrient and pollutants decreases. A decrease in evapotranspiration and/or aerodynamic resistance (the movement of evaporating water from the forest floor to the air) will lead to an increase in runoff (Grip, 1982) in accordance with the water balance equation (Equation 11).

In Swedish forest till soils, an increase in soil- and groundwater tables will lead to an increase in runoff as the soil water reaches horizons with higher hydraulic conductivity (Figure 22). This will in turn change the chemical composition of stream water leading to an increase of TOC and a decrease in Ca^{2+} (Bishop et al., 2004). In addition, an increase in soil water table may lead to reduced conditions affecting the redox processes and consequently, ion concentrations. For example, increasing the anaerobic zone may lead to denitrification and prevent a build-up of NO_3^- (e.g. Vitousek et al., 1979).

8.1.1 Soil water and groundwater tables

There are few documented effects on soil water and groundwater levels following harvest. Most of them come from the Nordic countries. For example, Lundin (1979) studied short-term changes (0-3 years) in soil water and groundwater following stem-only harvest in Sweden. During this period soil water levels increased between 5-46 mm and ground water between 14-96 cm depending on site and terrain. In general, the largest increase occurred immediately following clear-cut and in sites located at the bottom or middle part of the terrain. The study was carried out during the growing season (May to September). Heikurainen & Päivänen (1970) observed increased groundwater tables in mires immediately following clear-cutting during a two-year study period in Finland. Based on a large number of measurements, a relationship between runoff and groundwater tables could be established (Figure 23). According to the empirical results, the groundwater table rise ranged between 5.3 cm and 13.7 cm at a depth of 25 and 55 cm. In a drainage experiment in UK, Pyatt et al (1985) observed a small increase in groundwater level four years after harvesting (growing season) ranging between 4.7 cm to 34.7 cm.

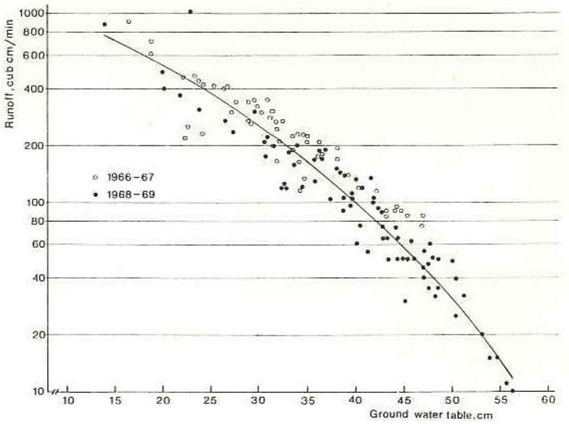


Figure 23. The relationship between runoff (cm³ per min) and groundwater table (cm). From Heikurainen & Päivänen, 1970.

In contrast to these results, Mannerkoski et al. (2005) could not detect any significant short-term (0-5 years) changes in groundwater level following stem-only harvest at five research sites in Finland, regardless of the location of the wells (mineral soil, peat land and till under peat). Individual changes were only noted for one well where the groundwater level decreased with 5 cm. The absence of groundwater response could be explained by firstly; the wells were located below the harvested area and secondly; only 10-30% of the catchments were harvested. Rusanen et al. (2004) investigated long-term (15-22 years) effects on groundwater tables and did not find any changes in groundwater tables of large aquifers after progressive felling covering 35-66 % of the recharge area. The non-saturated volumes of these aquifers were probably too large to allow for detection of small changes in groundwater recharge.

Up to this point, changes in soil water and groundwater following harvest in upland mineral soils have been discussed. However, harvest may also take place on forested wetland areas. Lundin (1999) described the hydrological changes in discharge areas (wet mineral soils or peat lands). Following harvest of five catchments, the groundwater levels rose by 0-17 cm during the first four years at one research site and by 0-13 cm during the first two years at the other site. As a result of increased groundwater tables, the annual stream runoff from the catchments increased by 4 to 125 mm or by 16-36%. No differences could be detected between harvest intensity (traditional clear-cutting or with saved shelter wood).

8.1.2 Stream runoff

Short-term stream runoff changes following clear-cutting have frequently been reported. In the Nordic countries, Kilhberg (1958) reported an increase in runoff of 90 mm per year but the validity of these results have been questioned by Grip (1987) since leakage took place at the weir. However, this would lead to an underestimation of the harvest effects. In comparison, Rosén (1984) reported increases in runoff during the first two years varying between 215-371 mm per year following clear-cutting. No difference could be seen with respect to harvest intensity (WTH↔CH) even though it was speculated that interception would be higher on sites with slash resulting in increased runoff. One reason for this could be due to differences in areal extent of the cutting. In the stem-only harvested area only 70% were logged compared with the whole-tree harvested catchment where all the trees were taken down (100%). Eight years later, increases were still noticed in the same areas varying between 220-274 mm per year as a mean over the eight year post-harvesting period (Rosén et al., 1996). Grip (1982) measured an increase in annual stream flow of 184 mm per year two years after stem-only harvest while Brandt et al. (1988) modelled increases varying between 165-220 mm per year using the PULSE-model. Sørensen et al. (2009) observed an initial increase in runoff following stem-only harvesting. Compared with an uncut control area, the runoff increased by 253 and 254 mm during the first and second year after harvest. This was caused in part by greater snow accumulation on the clear-cut sites. These Swedish results are in line with a Norwegian study by Haveraaen (1981) who documented increases between 200-250 mm per year two years after stem-only harvesting.

There are numerous studies describing short-term increases in runoff after cutting in Europe (e.g. Robinson & Dupeyrat, 2005; Adamson et al., 1987) and USA (e.g. Patric & Aubertin, 1977; Hornbeck et al., 1990; Briggs et al., 2000). For example, in the USA, Bormann et al. (1968) observed an increase in runoff by 40% one year after whole-tree harvest and herbicide treatment in the Hubbard Brook Experimental Area. Two years later the increase in runoff had decreased from 330 to 270 mm per year but was still elevated (Likens et al., 1970). In the same area Martin et al. (2000) found evidence of increased runoff in two harvested areas (strip cut and whole-tree harvesting). In the strip-cut catchment, the annual runoff increased significantly between 46-114 mm during the first seven years, followed by a decrease the next 8-27 years. The effects of whole-tree harvesting were not as clear. Significant increases of 51-152 mm were observed in the first, seventh, eight and thirteenth year after harvest. In the fourteenth year, no differences could be found.

Swank et al. (2001) reported long-term changes in annual stream runoff (31 years). Immediately following harvesting, the annual stream flow increased as a result of reduced evapotranspiration and thereafter decreased exponentially with time due to revegetation. During year 12-14 a number of trees died resulting in a temporary increase in runoff. In general, monthly increases occurred throughout the year but were not as high during spring flood (when soil water storage is high) compared with summer months where the largest increases were observed. Based on 250 catchment studies worldwide, Zhang et al. (2001) used a hydrological model to assess the long-term effect on water yield both in forests and grasslands.

Increases in runoff following cutting are not always observed. For example, Wang et al. (2006) could not find any significant increases in stream runoff (although there were strong indications) in spite of the fact that the lower two-thirds of the catchment were harvested and no efforts were made to keep a buffer zone. However, harvesting was carried out using a shelter wood technique, keeping up the transpiration rates. In Canada, Jewett et al. (1995) only saw minor changes (<10% for the first 12 years) following stem-only harvesting. The lack of response was attributed to a reduction in snow catch efficiency, reduced vegetation efficiency of fog and cloud water, decreased albedo affecting evaporation and soil disturbances (e.g. road tracks) which could have diverted water from the weir. In addition, Caissie et al. (2002) could not detect any changes in annual runoff two years after 2.3% of the catchment was harvested, but found increases in summer peak flows when 23.4% of an adjacent catchment was clear-cut.

Harvesting effects on annual runoff have been shown to vary according to season. For example, Rosén (1984) found that the largest increase took place during the snowmelt and snow-free period (i.e. spring and summer). Larger snow accumulation during winter and reduced transpiration and aerodynamic resistance during summer are suggested mechanisms. During winter, no differences in runoff were noticed. These results are partly in agreement with those of Grip (1982) who observed the largest increases during summer (74 mm) and snowmelt (44 mm) but also a small increase during winter (19 mm). In addition, Brandt et al. (1988) modelled that the largest increase would occur during spring flood (50-120 mm) compared with the summer period (75-95 mm). Sørensen et al. (2009) did not find any statistical differences during peak flow (except for spring-flood), which means that the increase in runoff took place during base flow (39 and 51 mm during year 1 and 2, respectively) and intermediate flow (58 and 51 mm during year 1 and 2, respectively). As a consequence, the largest increase was observed during the growing season with a successive decrease as summer turned into autumn. During the winter, small increases of 12 mm and 26 mm were recorded. Likens et al., (1970) also observed increases in base flow during summer and earlier snowmelt.

Harvesting effects may also change the peak (storm) flow magnitude and duration. For example, Guillemette et al. (2005) compared the results from a clear-felling experiment in Canada on rainfall generated peak flows with the results from 50 experiments across the world. Harvesting covering over 85% of a conifer-dominating catchment resulted in an increase in max peak flow of 57% and a mean peak flow increase of 54% and 36% for the smaller storm flows during a five-year period. These increases were noted even before the felling was fully completed. Compared with the 50 worldwide experiments these increases were in the upper end (average 49%, 70-100% clear felled area) probably owing to a modification of flow paths via skid trails and ditches. Martin et al (2000) also found moderate peak flow increases, which had disappeared within 4-6 years as new vegetation had established. The increase in runoff occurred as an increase in base flow during the growing season. However, Robinson & Dupeyrat (2005) found a general increase in low flows (Q_{95}) between 5 to 15 mm and no change in peak flow. Also, in a study by Patric and Aubertin (1977), runoff changes occurred mainly during the growing season and no change in peak-flow was observed. Swank et al. (2001) found significant changes in the flow prior to a storm, quick flow volume, peak flow rate, storm duration and recession time were higher following harvesting. The only parameter unaffected was the peak flow timing.

To summarize, one of the most noticeable effects following harvest is an increase in soil- and groundwater tables leading to increased runoff. However, the top-most soil usually becomes drier as a result from increased soil temperatures (e.g. Yanai et al., 2003). In most studies, short-term (usually less than five years) increases in annual runoff were reported ranging between 10 and 450 mm (Table 8). In the Nordic countries increases between 90-371 mm (average 212 mm) have been observed. However, in some studies no harvesting effects were observed (e.g. Jewett et al., 1995; Wang et al., 2006; Caissie et al., 2005) and in one case (Martin et al., 2000), stream runoff decreased compared with pre-harvest conditions most likely due to a rapid-growing young forest stand. The runoff increases are usually attributed to larger snow packs during winter, decreased evapotranspiration during summer and decreased aerodynamic resistance. There are also seasonal variations and reports of changes in peak flow (e.g. Guillemette et al., 2005; Martin et al., 2000; Swank et al., 2001) while other report no change (e.g. Patric & Aubertin, 1977; Sørensen et al., 2009; Robinson & Dupeyrat, 2005).

The largest runoff increase seems to take place during base flow conditions (i.e. during the growing season) when the evapotranspiration in growing forests peak (Swank et al., 1988). The effect magnitude depends on a number of factors and the most important seems to be the proportion of the catchment being harvested (Brown et al., 2005). No differences could, however, be seen between stem-only and whole-tree harvesting (Rosén, 1984) although shelter wood harvesting seems to cause only minor effects (Caissie et al., 2002). With time, the runoff returns to pre-harvest conditions as new vegetation is established, although long-term effects (>20 years) have been observed (e.g. Hibbert, 1967).

8.2 pH and H^+

There are a number of natural soil reactions that can lower or increase the pH. For instant, plant respiration, dissociation of organic acids, vegetation uptake of cations, sulfide oxidation, and nitrification are examples of natural acidifying processes (Brady & Weil, 1996). In addition to these processes, anthropogenic acid deposition has caused soil and surface water acidification. Weathering of silicates, carbonates, Al- and Fe-hydroxides, cation exchange processes, SO_4^{2-} adsorption and vegetation uptake of anions are examples of buffering mechanisms acting to prevent the pH to decrease (Brady & Weil, 1996).

In a review by Binkley and Högberg (1997) it was shown that the pH of Swedish forest soils have decreased over a period of decades, often related to increasing stand age. The decrease corresponds to increased ionic strength of the soil solution (e.g. via deposition of sulfate), reduced base saturation (via plant uptake and leaching losses) and soil organic matter increases (acting as an acid, increases the acid strength and providing more exchange sites). Thus, a decrease in pH may result from a number of factors acting independently or together.

8.2.1 Soils

The data suggests that harvest effects on soil exchangeable pools (H^+) are generally small and mostly non-significant. However, the short-term effect on soil pH can be high. Statistically significant treatment differences between clear-cut and mature forests (pre- and postharvest or cut with uncut) are most frequently reported during the first five years when soil pH usually increases, both in the forest floor and in the mineral soil. For example, Hendrickson et al. (1989) measured significantly higher pH-values in the forest floor (+0.3 to +0.4 units) and mineral soil (+0.3 to +0.4 units) three years after harvesting (both stem-only and whole-tree) compared with an adjacent growing control forest. They discussed the possibility of increased concentrations of Ca^{2+}, which could have replaced H^+ on the soil exchangeable complex. Belleau et al. (2006) observed a temporary increase in forest floor pH of +0.9 units two years after stem-only harvest which coincided with higher base cation concentrations, base saturation and CEC_{eff}. However, by the third year statistical differences could no longer be found (which has been used in this review in order to avoid duplicates). According to the reviewed material, short-term increases in forest floor pH vary between +0.3 to +0.9 units (n=3, CH and WTH).

Medium- to long-term increases was reported by Simard et al. (2001) who measured an increase in forest floor pH of circa +0.2 units up to sixteen years after whole-tree harvest compared with an uncut mature forest as a result of increased concentrations of Ca^{2+} and lower nitrification rates. Katzensteiner (2003) observed a medium-term increase in the forest floor by more than one unit after stem-only harvest compared with preharvest conditions. This was attributed to a liming effect caused by the mixing of the forest floor with Ca-rich surface debris and increased bioturbation.

Ranger et al. (2008) and Johnson et al. (1991b) reported short-term pH increases in the mineral soil. Ranger et al. (2008) observed increases (down to 45 cm) varying between +0.1 and +0.2 pH-units (H_2O) three years after whole-tree harvesting of an acid forest soil. Significant short-term increases were also observed when measuring pH in a salt solution (KCl) but after three years no difference could be detected compared with pre-harvest conditions. No reasons were given for the observed increase. However, before cutting the nitrification rate was high and decreased significantly after cutting. Ceased nitrification could have caused the increased pH. In comparison, Johnson et al. (1991b) observed a small increase of +0.1 units in the Bs-horizon of a Podzol, three years after whole-tree harvest of a northern hardwood forest. They explained the results by a mixing of the forest floor with the mineral soil, which lead to an accelerated decomposition. In a later study by Johnson et al. (1997) in the same study area, the pH had increased further by +0.2 to +0.3 units compared with pre-harvest conditions. Long-term changes in the mineral soil is reported by McLaughlin & Phillips (2006) who observed an increase in pH of +0.6 units in the mineral soil, 16-17 years after WTH resulting from higher CEC_{eff} and higher cation adsorption. Based on the reviewed material the observed mineral soil short-term increases vary between +0.1 to +0.4 pH-units (n=9, CH and WTH).

Short- and long-term decreases in soil pH after harvest (both in the forest floor and in the mineral soil) have been reported in a small number of studies. With time, the pH starts to decrease in the forest floor as a result of natural biological acidification. For example, in a chronosequence studied by Katzensteiner (2003) the pH in the LF- and H-horizon of a 140-year old forest stand was -0.6 and -1.1 units lower, respectively, compared with a 10-year old stand which was cut using conventional harvest methods. Johnson et al. (1991b) measured short-term (<3 years) decreases in the mineral soil pH (E- and Bh-horizon) varying between -0.2 and -0.3 units. Decreases in mineral soil pH appeared to be linked to nitrification during which time H^+ is produced and partly consumed via aluminium buffering (see references in Johnson et al., 1991b to

Lawrence et al, 1987 and Fuller et al., 1987). Brais et al. (1995) reported short- and medium-term decreases in the forest floor and mineral soil (down to 20 cm) after whole-tree harvest of a mixed softwood/hardwood forest. The effects were most pronounced on moist soils where pH decreased between –0.6 and -0.8 units in all soil layers. On dry to fresh sites, pH decreased significantly in the 10-20 cm soil layer. An increased water table after clear-cutting on the moist sites could have slowed down mineralization and the release of base cations. The pH decrease on dry to fresh sites was more difficult to explain since no differences in nitrification rates could be found between treatments.

Generally, the differences between whole-tree and stem-only harvest (WTH↔CH) are insignificant both in the short- and the long-term (Hendrickson et al., 1989; Wall, 2008; Bélanger et al., 2003; Thiffault et al., 2006; Belleu et al., 2006 and Hendrickson et al., 1989). However, there have been some medium- to long-term observations of lower forest floor pH after WTH. For example, Staaf & Olsson (1991) found a decrease in pH varying between -0.2 to -0.4 units in the humus layer 7-9 years after whole-tree harvest. No treatment differences were found in the mineral soil (0-5 and 5-10 cm). Considering that the sites were already quite acidic (pH ca. 4.0 at three out of four sites) the small change in pH was likely to be of minor importance (Staaf & Olsson, 1991), however, corresponding to large amounts of H^+. Resampling of the same sites after 15-16 and 30 years (Olsson et al., 1996b; 2004) did not show any significant treatment differences. Thus, the initial decrease in pH was only temporary. Nykvist & Rosén (1985) got similar results. Following WTH, the pH in the humus layer was significantly lower (-0.1 to -0.3 units in the pH-interval 4.2 to 4.7) in 4 out of 9 sites 6-21 years after final cutting. The same trend was observed at two other research sites although not statistically significant (Nykvist & Rosén, 1985 and Nykvist, 1990). No effect could be found in the mineral soil. In terms of pools, Olsson et al. (1996b) found significantly higher accumulation of exchangeable H^+ in the forest floor (but not the mineral soil) 15-16 years after whole-tree compared with stem-only harvesting.

8.2.2 Soil water, groundwater and stream water

After cutting, the pH in soil water, groundwater and stream water often decreases as a result of increased nitrification during the clear-cut phase. For example, Jewett et al. (1995) measured a short-term (5 years) decrease in soil water pH at 15-90 cm depth after stem-only cutting compared with pre-harvest conditions. The drop in pH was consistent with an increase in NO_3^- concentrations, especially below hardwood and mixed forests. Hendrickson et al. (1989) reported a significant short-term (3 years) pH decrease after stem-only and whole-tree harvest in forest floor leachate and mineral soil water (100 cm) compared with a control forest. The pH drop coincided with increased NO_3^- concentrations. Dahlgren & Driscoll (1994) and Ring et al. (2001) observed similar results following whole-tree harvest.

In contrast to these results, Ranger et al. (2007) observed a short-term increase in soil water pH of +0.2 to +0.8 units following whole-tree harvest compared with pre-harvest conditions. The limited effects on other elements were unexpected and it was hypothesized that the presence of Douglas fir (*Pseudotsuga menziesii F.*) stimulated nitrifiers and by removing the trees their activity decreased. In Finland, Piirainen et al. (2004) measured a short-term increase in soil water pH in the E- and B-horizon following stem-only harvest compared with an uncut forest. In this area, nitrification is small after cutting (Piirainen et al., 2002) and probably not significantly affecting pH. Mineralization and decomposition releasing base cations could be the reason for the observed pH increase.

A long-term change in soil water pH is reported by McLaughlin & Phillips (2006). Following whole-tree harvest, the pH increased in a regenerating stand compared with a control forest, 16-18 years after harvest. After such a long time, nitrification was of a minor importance. Instead, increased weathering, changes in soil organic matter chemistry and a mixing of the organic horizon with the mineral soil acting to increase the soil exchangeable pool of Ca^{2+} was suggested to be responsible for the higher pH.

Changes in groundwater pH after cutting are not well described in the literature and only available for stem-only harvest. In Finland, Rusanen et al. (2004) did not find any changes in spring water from large aquifers after stem-only cutting compared with pre-harvest conditions. Neal et al. (2004) observed a decrease in pH in the groundwater six years after stem-only harvest compared with pre-harvest conditions and a control forest. This decrease was consistent with the results from stream water measurements in the same area where pH dropped by 0.3 and 0.4 units up to six years after harvest as a result of increased NO_3^- concentrations. However, compared with stream water, the pH in groundwater was lower due to higher levels of carbon dioxide. As the groundwater discharge into the streambed, carbon dioxide degasses and stream water pH increases. In contrast, Mannerkoski et al. (2006) did not find any significant pH differences after cutting compared with a control forest.

Short-term decreases in stream water pH have been reported in many parts of the world. For example, Likens et al. (1970) observed a decrease by 0.7 units two years after stem-only harvest compared with pre-harvest conditions. Neal et al. (1992) got similar results. During the first two years after stem-only cutting, stream pH dropped by 0.7 units before it returning to preharvest levels. The stream water pH decrease only occurred during base flow conditions when pH normally is at its highest. Tremblay et al. (2009) measured a decrease in pH varying between 0.1 and 0.4 units compared with a control stream. The small change in pH did not indicate any substantial nitrification although NO_3^- concentrations increased slightly (mean increase of 0.4 mg per litre) compared with pre-harvest values. Nitrification and/or a release of organic acids from logging debris were considered to have caused the drop in pH. Neal et al. (2004), Lawrence et al. (1987) and Romanowicz et al. (1996) have also reported decreased pH and increased NO_3^- concentrations in stream water after stem-only and whole-tree harvesting.

In contrast to these results, Parfitt et al. (2002) did not observe any change in soil water pH (at soil depths of 10 and 60 cm) over a four-year study period after stem-only cutting, while stream water pH in the same study area increased compared with pre-harvest conditions. This increase was consistent with an increase in bicarbonate concentration resulting from an invasion of weed and grass. Grip (1982) and Kubin (1998) also reported small short-term increases in stream water pH after stem-only cutting, while McLaughlin & Phillips (2006) have recorded a long-term (16-18 years) increase. Compared with a control stream, the stream water pH was circa one unit higher after whole-tree harvesting. The lower pH (6.1) of the reference stand (77-85 years old) was associated with lower base cations pools and water fluxes most likely the result of long-term DOC- and SO_4-leaching. Additionally, the CEC_{eff} increases of the regenerating stand offer more exchange sites to which base cations can adsorb (J. McLaughlin, written communication).

Changes in soil water, groundwater and stream water pH following differences in harvest intensities (WTH and CH) are not well described and do not include medium- to long-term effects. No clear pattern has been found. For example, Wall (2008) did not find any significant soil water pH differences in forest floor leachates between whole-tree and stem-only harvest plots, while Staaf & Olsson (1994) measured higher pH after whole-tree harvesting compared with stem-only during a five year post-cutting period. In contrast, Högbom et al. (manuscript) found that the pH in soil water at 50 cm depths was lower when comparing whole-tree harvest with stem-only. Hendrickson et al. (1989) reported both lower and higher pH values in the leachate from the forest floor and mineral soil, respectively, after whole-tree harvest compared with stem-only.

There are few studies describing changes in H^+ flux and they only report short- to medium-term changes in soil water and stream water. Lower hydrogen fluxes from soil water in the forest floor (Emmett et al., 1991) and in the mineral soil below 50 cm (Örlander et al., 1997) have been reported after whole-tree harvest compared with stem-only. Due to the short duration of these studies the long-term effect is unknown. By removing logging residues, the potential for nitrification decreases which lowers the production of H^+. At the same time, stream runoff generally increases. For example, Rosén et al. (1996) reported a runoff increase by 274 mm and decreased H^+ and NO_3^- concentrations. This caused a short- to medium-term decrease of 0.2 kg H^+ per hectare in stream water following whole-tree harvest, but not stem-only, compared with pre-harvest conditions.

There is no clear pattern regarding stand characteristics between sites responding with a change in soil pH after harvest. Increases, decreases or no change in pH have been observed both in hardwood (Johnson et al., 1991b; Belleau et al., 2006), softwood (Nykvist & Rosén, 1985; Staaf & Olsson, 1991; Simard et al., 2001; Katzensteiner, 2003; McLaughlin & Phillips, 2006; Ranger et al., 2008) and mixed hardwood/softwood (Hendrickson et al., 1989; Brais et al., 1995) forests as well as in various soil types such as Leptosols (Katzensteiner, 2003), Podzols (Nykvist & Rosén, 1985; Johnson et al., 1991b; Staaf & Olsson, 1991; McLaughlin & Phillips, 2006; Hendrickson et al., 1989; Simard et al., 2001), Luvisols (Belleau et al., 2006; Brais et al., 1995) and Cambisols (Ranger et al., 2008).

Instead, the likelihood of significant changes in soil pH taking place in acid forest soils depends to a large extent on whether or not nitrification occurs (releasing H^+) or if the release of base cations (primarily Ca^{2+}) via decomposition and weathering is large enough to consume the produced H^+. Decomposition and nutrients release from various logging debris after cutting is well described in the literature (e.g. Romell, 1957; Tamm, 1964; Kimmins 1977). However, higher mineral weathering after harvest is less well documented. McLaughlin & Phillips (2006) discussed the possibility of increased weathering rates of silicate minerals following whole-tree harvest due to increased carbon dioxide concentrations as an explanation for the near doubling in exchangeable Ca^{2+} concentrations. Thus, higher weathering rates after harvest could increase soil exchangeable pools leading to an increase in soil pH. Increases in water table may in turn affect decomposition rates by slowing down the process. In less acidic forest soils, such as the Northern Limestone Alps, mixing of Ca^{2+}-rich surface debris has been suggested as a pH increasing process, a situation not likely to occur for most forest soils in Sweden.

Similarly, the results from soil, soil water, groundwater and stream water studies suggest that short-term increases or decreases in soil water pH after cutting is closely related to changes in NO_3^- production. This is in agreement with the reviews of Feller (2005) and Martin et al. (2000). Other processes, shown to affect pH after harvest, include decomposition (releasing base cations), aluminium buffering, changes in the quality of soil organic matter (affecting CEC) and the release of organic acids from logging debris.

In summary, mechanisms that can help explain the observed pH changes include:

- Nitrification (release of H^+)
- DOC-leaching (release of H^+ and ANC)
- Changes in CEC (affects acidity and base saturation)
- Changes in base cation concentrations (affects base saturation and ANC)
- Carbon dioxide production (increased weathering)
- Weathering (consuming H^+ and releasing base cations)

Regarding pH effects in aquatic environments, the occurrence of nitrification seems to be the most significant factor. It also seems that the effects of cutting per se are more important than differences in harvest intensity in terms of nitrification activity. Therefore, it is probably more crucial to limit excess NO_3^- production in forest soils compared with reducing the harvest intensity (base cation removal), to limit short-term pH declines in the aquatic environment. Unfortunately, the long-term effects of harvest intensity are largely unknown, but available results indicate that the pH-effects either level out and return to preharvest conditions or are small.

8.3 Carbon (C) and CEC

Organic matter plays a major role in the global carbon cycle and is important for soil fertility (cation-exchange properties, structure, bulk density, N, P, S and water status) and forest productivity (e.g. Johnson & Curtis, 2001; Kreutzweiser et al., 2008; Yanai et al., 2003). Soil organic carbon generally makes up circa 50% of organic matter content measured by weight loss on ignition (e.g. Bélanger et al., 1993; McLaughlin & Phillips, 2006; Thiffault et al., 2006; Brais et al., 1995). The relationship between cation exchange capacity (CEC = $H^++Al^{n+}+Na^++K^++Mg^{2+}+Ca^{2+}$, measured in 1M NH_4OAc, pH=7.0) and soil organic matter content in B-horizon podzols in central Sweden is illustrated in Figure 24. According to the strong positive relation, a change in organic matter content will lead to a change in CEC. In clay-rich soils, the relationship between CEC and OM is not as linear due to the exchange sites available on clay surfaces. According to Helling et al. (1964), CEC of organic matter increases by 30 meq per kg per unit increase in pH, which is 6.7 times higher than for clay (4.4 meq per kg).

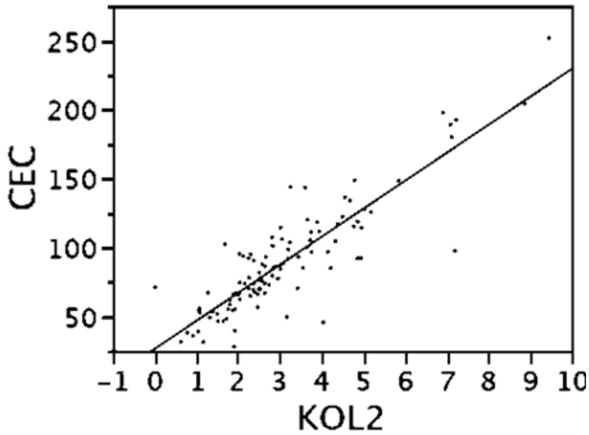


Figure 24. The relationship between cation exchange capacity (µeq per g = meq per kg) and soil organic matter content (KOL2 in %). Data: Swedish Soil Survey 1991-2003, n=108.

Forest harvesting could have a potential impact on the carbon dynamics in forest soils. Factors to consider are mechanical disturbances affecting the structure and function of the forest floor, biomass input if logging residues are left on the ground, biomass removal if logging residues are harvested, litter production in the regenerating stand and understory vegetation, abiotic changes in temperature and moisture, leaching of DOC, etc. A decline in organic matter content and

CEC may decrease the buffering capacity of the soil and subsequently increase the sensitivity to acid inputs.

Several authors such as Johnson & Curtis (2001), Sanchez et al. (2006) and Kreutzweiser et al. (2008) have recently reviewed harvest effects on the carbon dynamics. Therefore, only a few studies are included in this review describing cutting effects on soil carbon concentrations and pools over time.

In general, the results suggest that there is no short- or long-term cutting effect on total carbon concentrations compared with pre-harvest conditions and uncut forests regardless of harvest method (stem-only and whole-tree). For example, Katzensteiner (2003) used a chronosequence approach including a clear-cut stand (0 years), a regenerating forest (10 years) and an old mature forest (140 years old). No treatment effects could be found in the forest floor between the clear-cut and the regenerating stand or mature forest. However, when comparing the regenerating stand with the mature forest, a significant increase of 37.7 mg per gram (or 8%) in total C concentration was found in the humus layer after stem-only harvest. This increase is likely to be the result of litter input from dead leaves/needles, branches and roots with time as the forest grow old. In addition, McLaughlin & Phillips (2006) examined the long-term (16-18 years) effects of whole-tree harvest with pre-cutting values but could not find any significant treatment effects in the forest floor or mineral soil. However, when comparing the results with an uncut control forest a long-term decrease of -63 mg per gram was found in the forest floor but not in the mineral soil. Despite this change in total C concentration, no significant change was observed in soil carbon pools (see below) and it was concluded that whole-tree harvest does not lead to depletion in C. No reason was given for the decrease in total C concentration.

The few studies showing statistically significant harvest effects indicate that the total soil C concentrations decrease in the forest floor but increase in the mineral soil, which is in agreement with the observed soil C pool changes (see below). For example, pre- and postharvest differences following stem-only and whole-tree harvest, respectively, were studied by Johnson & Todd (1998) up to 15 years after harvest. Significant C concentration increases were found in the A-horizon (0-10 cm) but not deeper down in the soil, regardless of cutting method. Since logging residues were removed during whole-tree harvest, differences in litter could not explain the increase in total carbon concentration. Instead it is believed that an invasion of weeds and/or C losses during sample storage could be responsible for the increase.

Based on a low number of studies it appears as if the total C pools decrease in the forest floor and increase in the mineral soil. For example, Ranger et al. (2008) measured a short-term decrease in the forest floor C pools and an increase in the mineral soil (down to 60 cm) after whole-tree harvest compared with pre-cutting values. These results are in agreement with Olsson et al. (1996a) who measured long-term treatment effects on soil C pools at four sites in Sweden after stem-only and whole-tree harvest. In the humus layer, soil C decreased significantly at all sites regardless of harvest intensity, explained by accelerated decomposition rate (measured as a change in C/N over time). In contrast, C increased significantly in the mineral soil at three out of four sites explained by a downward transport of organic matter with low concentrations of N. The process was enhanced by the sandy texture facilitating transport and root decay in the mineral soil adding to the carbon pool. Mechanical mixing as an explanation was ruled out due to the gradual decrease. Regardless of harvest intensity, Olsson et al. (1996b) did not measure any change in soil C pools between the treatments (WTH↔CH). Slow decomposition of logging residues (still visible on the ground) and the fact that they were excluded in the analysis is one possible explanation. Resampling of the same sites, thirty years after final felling, demonstrated an increase in the soil C pools since the last sampling (1990/91) in the humus layer (Olsson et al., 2004). However, compared with pre-cutting values, the pools were still lower. In contrast to these results, McLaughlin & Phillips (2006) could not find any treatment effects in the forest floor or mineral soil following whole-tree harvest 16-18 years after harvest compared with pre-harvest conditions and an uncut forest.

Short- and long-term post treatment differences in total C concentrations and pools between whole-tree and stem-only harvest are summarized in Table 14 and Table 16, respectively. Following harvest of a northern hardwood forest no treatment difference could be found in mineral soil C concentrations (A-, E-, and Bt-horizon) fifteen years after harvest (Johnson & Todd, 1989). Similarly, no long-term treatment differences in soil C pools in the forest floor and mineral soil could be found in a Swedish study (Olsson et al., 1996a; Olsson et al., 2004). In contrast, Johnson et al. (2002) observed lower forest floor (litter + residues) in Oak Ridge, Tennessee and higher mineral soil C pools in Coweeta, North Carolina, after 15 and 17 years, respectively. The lower soil C pools in the forest floor at the WTH sites were explained by the removal of logging residues. No differences were found in the mineral soil in Oak Ridge. The C pool decrease in the mineral soil in Coweeta was not discussed. Despite individual differences between the sites (n=4), Johnson et al. (2002) concluded that *"in general, harvesting had no lasting effects on soil C"* despite initial increases compared with pre-harvest conditions. Also, "*the long-term effects of forest residues on ecosystem C pools, when they occur, will be manifested primarily as differences in biomass rather than soil or litter C".*

The result of this study describe a pattern of decreasing and increasing concentration and pools in the forest floor and mineral soil, respectively, compared with pre-harvest conditions or uncut control forests. However, this approach does not pick-up temporal changes that might occur between a young forest stand and a mature forest. For example, Olsson et al. (2004) described an overall trend with decreasing pools in the forest floor during the first 15 years followed by a gradual build-up during the next 15 years. However, when comparing data from the 30-year old stand with pre-harvest data, soil pools were still lower. Thus, the temporal trend is not accurately described.

The results by Olsson et al. (2004) are in agreement with the temporal pattern observed by Covington (1981) and Katzensteiner (2003). Using a chronosequence design, Covington (1981) measured a decrease in soil organic C pools by 50% (or 30.7 Mg per hectare) in the forest floor during the first fifteen years following stem-only harvest. This decrease was followed by a sharp increase as the stands grew older. Suggested mechanisms, which could explain these results, include accelerated decomposition rates (resulting from increased temperature and moisture content) and changes in litter input. The initial sharp decline in organic matter immediately following cutting was explained by biomass removal whereas the rapid increase 15-20 years later was attributed to a delayed decomposition of woody debris and litter from the new forest stand. However, decomposition studies by e.g. Prescott et al. (2000) are not in agreement with this explanation since increased temperatures led to a drying out of the top most layer of the forest floor leading to decreased and not increased decomposition rates. The accuracy of the "Covington curve" have thus been questioned (Yanai et al., 2003) and replication of the Covington study supported some but not all of the previous results (Federer, 1984). Alternative explanations for the Covington (1981) results have been mechanical disturbances leading to a mixing and movement of the forest floor and mineral soil and changes in logging techniques over time.

The short- and long-term effects of harvest on CEC are usually small and non-significant compared with controls and pre-harvest conditions, regardless of harvest method. These results are in agreement with those discussed for soil C. However, just like soil C, individual observations of increases and decreases in CEC have been documented.

For example, Johnson et al. (1991b) measured a short-term (<3 years) CEC *decrease* in the Oa- and the E-horizon and a CEC *increase* in the Bh- and Bs1-horizon following whole-tree harvest compared with pre-harvest conditions. Similar trends were observed for the base cation concentrations (Ca^{2+}, Mg^{2+} and K^{+}) and exchangeable acidity (H^{+} and Al^{n+}). Several explanations were given to the observed changes in CEC affecting either the quantity and/or quality of soil organic matter. The organic matter content e.g. decreased significantly by 37% in the E-horizon (and by 17% in the Oa-horizon). Decomposition and leaching to lower lying horizons could have explained the decrease in organic matter. The latter would have explained the increase in CEC in the B-horizon (Johnson et al., 1991a), but this horizon showed no change in organic matter content. Hence, increases in CEC further down in the mineral soil must have occurred due to in-situ processes affecting the organic matter quality. Increased negative charges on the soil organic matter due to increased humification is a plausible explanation. Measurements of the cation exchange capacity of the organic matter only (CEC_o) showed an increase in the Bh- and Bs1-horizon compared with pre-harvest conditions supporting this explanation. No such change occurred in the E-horizon.

In a follow-up study five years later (i.e. eight years after whole-tree harvest) the CEC were still elevated in the Bh-, Bs1- and Bs2-horizon (Johnson et al., 1997) compared with pre-harvest conditions. However, the observed differences in the Oa- and E-horizon after three years (see above) were not present any longer. During the eight years study period the ratio between CEC and organic matter increased by 25%, which together with changes in pH could explain the observed changes.

Short- and medium-term increases in mineral soil CEC were also found up to fifteen years after stem-only and whole-tree harvest of a mixed oak forest growing on an Acrisol (Johnson and Todd, 1998). Compared with pre-harvest conditions, the CEC had increased significantly after both stem-only (A- horizon) and whole-tree harvest (A- and E- horizons). The Bt-horizon was unaffected by harvest in contrast to Johnson et al. (1991b) and Johnson et al. (1997). The increases in CEC were not explained, but losses during storage could have been responsible for the results. When comparing treatment differences (WTH↔CH) no differences were found for the A-, E- or Bt-horizons.

Similar results were found by Ranger et al. (2008) who studied changes in CEC following whole-tree harvest of a Douglas-fir stand growing on a Cambisol in France. No change in CEC was found the first year after cutting. However, during the second year a small increase in effective cation exchange capacity (CEC_{eff}) in the upper mineral soil layers (0-5 cm) was observed. Changes further down in the mineral soil (down to 60 cm) were observed the third year after harvest in the 30-45 cm soil layer. No changes were observed for the middle or bottom soil layers compared with pre-harvest conditions. CEC and organic carbon exhibited similar patterns of variation. Despite these changes, Ranger et al. (2008) concluded that CEC was not affected by harvesting.

Opposite to this, Belleau et al. (2006) found a significant short-term (2 years) CEC increase in the forest floor, but not the mineral soil (0-10 cm), compared with a control forest following stem-only harvest of an Aspen stand growing on a Luvisol. The forest floor CEC increase and vegetation increment led to nutrient immobilization, partly explaining the lack of CEC effects in the mineral soil.

In contrast, Brais et al. (1995) did not measure any short- to medium term (5-12 years) changes in CEC in the forest floor or mineral soil (0-10 and 10-20 cm) after whole-tree harvest of a dry to fresh site in Canada compared with a growing uncut forest. However, decreases in CEC and base saturation were found on a moist site, both in the forest floor and the mineral soil while the acidity increased. These results are in agreement with McLaughlin & Phillips who could not find any significant long-term (16-18 years) treatment effects on CEC in the forest floor following whole-tree harvest compared with control and pre-harvest conditions. However, just like Johnson et al. (1991b) and Johnson et al. (1997) an increase was observed in the mineral soil where CEC_{eff} doubled compared to pre-harvest conditions and control forest. The increase in CEC_{eff} may have increased the retention of Mg thus counterbalancing increased leaching losses. The increase in base cation retention also increased pH, Ca^{2+} and decreased the solubility of inorganic aluminium. Using a chronosequence approach, Katzensteiner (2003) did not find any significant changes in CEC_{eff} after stem-only harvest in the H-layer compared with an uncut 140 years old forest. In the regeneration stand, which was 10 years old, CEC_{eff} were significantly higher. Katzensteiner (2003) believed that the input of base cations from decomposed logging residues and/or changes in chemical composition might be responsible for the high CEC_{eff} in the forest floor.

Post-harvest effects on CEC between different treatments (WTH↔CH) are often non-significant. For example, Bélanger et al. (2003) did not find any significant short-term effects in the forest floor (F & H) and mineral soil (0-20 cm) three years after whole-tree harvest of a black spruce forest in Canada. Belleau et al. (2003) got similar results. Medium to long-term (6-21) differences is described by Nykvist & Rosén (1985) who could not find any differences in CEC in the humus layer between WTH and CH after harvest of nine Swedish coniferous sites. These results are in agreement with Johnson & Todd (1998). Olsson et al. (1996b) did not measure any long-term (15-16 years) significant effects between WTH and CH at four different coniferous sites in Sweden. However, when the results from all four sites were combined lower CEC_{eff} per unit area were found after whole-tree harvest compared with CH. Lower base cation pools were given as an explanation. In a single study by Thiffault et al. (2006) treatment differences between WTH and CH have been found. They report long-term lower CEC both in the forest floor and the mineral soil (0-20 cm) fifteen to twenty years after whole-tree harvest compared with stem-only (Table 13). The lower CEC following whole-tree harvest at one of the three sites were attributed to lower input of soil organic matter resulting in lower organic C concentration in the mineral soil. In addition, changes in the quality of soil organic matter as explained by Johnson et al. (1991b) could have been responsible.

Johnson et al. (1997) reported effects on the CEC pools. A short-term (3 years) decrease in the forest floor CEC pool, but not mineral soil, was observed after whole-tree harvest compared with pre-cutting values. The effect had disappeared eight years later. Olsson et al. (1996b) observed lower forest floor CEC following WTH compared with CH 15-16 years after final felling. However, for most sites no treatment differences could be found.

In summary, most sites show no change in C and CEC concentrations and pools following harvest. However, when measurable, the concentrations and pools of soil C appear to decrease in the forest floor and increase in the mineral soil. These changes occur regardless of cutting method, indicating that the effect of harvest per se is more important than differences in harvest intensities since most difference between WTH and CH were non-significant. However, a larger dataset should be considered since the review of Johnson & Curtis (2001) suggest otherwise. In soils where organic matter is the major source of CEC, changes in decomposition rates (via increased water tables, mechanical mixing, and differences in litter input) could affect the amount of CEC. In addition, pH changes affect CEC. However, the effect of harvest on pH is considered small and insignificant (chapter 8.2). An increase in pH following cutting (or due to reduced acid deposition) would increase CEC and limit the base cation leaching after harvest. With time, CEC (and soil C) is expected to increase due to litter input from the new stand and understory vegetation. The results by Johnson et al., (1991b) and Johnson et al. (1997) suggests that harvesting may lead to a long-term increase in CEC in the spodic B-horizon of Podzols, which also may reduce the base cation leaching from the soil. No clear pattern between soil type, climate, tree species and treatment has been found.

Processes responsible for the observed trends in total C include:

- Decomposition
- Leaching
- Litter input
- Weed invasion
- Retention

And in CEC:

- Decrease in the amount of organic matter
- Increase or decrease in pH
- Increased humification
- Weathering (increasing the amount of clay minerals)
- Fewer water bridges between clay and organic matter

8.4 Nitrogen (N)

Clear-cutting effects on total N concentrations and pools show similarities with those observed for soil C; i.e. forest floor decreases due to decomposition and mineral soil increases due to retention. Other N effects include forest floor increases due to e.g. plant uptake from deeper soil horizons, which accumulates in the forest floor via litter input, as well as decreases in the mineral soil as a result of leaching losses. Thus, the soil N effects of clear-cutting appear to be much more site-specific compared with the effects on soil C.

For example, Mroz et al. (1985) measured short-term (2.5 years) changes following whole-tree harvest of three hardwood forests of varying site quality in the US. Compared with pre-harvest, the total N concentrations decreased in the forest floor at the two rich sites, while it increased at the poor site. However, the forest floor N pools decreased at all three sites between -846 to -87 kg per hectare. Large decreases in total N concentrations and pools also occurred in the mineral soil (E, Bs and BC) varying between -0.1 to -0.4 mg per gram (-484 to -1255 kg per hectare) at all three sites except for the Bh-horizon where total N concentrations increased between +0.1 and +0.2 mg per gram. Thus, some of the leached N was retained in the Bh-horizon. The decreased concentrations were most likely due to increased leaching of total N (Mroz et al., 1985). These large losses of up to 1300 kg N per hectare was later questioned by Binkley (1987) as being unrealistically high for this area and commented on by Mroz et al. (1987).

Similar results were found Ranger et al. (2008) who measured decreases in total N pools in the forest floor and mineral soil three years after whole-tree harvest compared with pre-harvest conditions attributed to decomposition and leaching losses. In Sweden, Nykvist (1977a) documented a short-term (4 years) decrease in total N pools in the humus layer after stem-only (-560 kg per hectare) and whole-tree harvest (-346 kg per hectare) at a site situated in the southern part of Sweden. However, at the northern site, forest floor N pools increased by +116 (CH) and +2 (WTH) kg per hectare. No explanation was given, but factors such as nitrogen deposition, leaching from logging residues, symbiotic N-fixation and/or redistribution of soil N from the mineral soil to the forest floor via vegetation may be of importance for the observed changes.

These results are partly in agreement with Hendrickson et al. (1989) who did not find any short-term (3 years) treatment effects on total N concentration in the forest floor or mineral soil after stem-only harvest compared with a control forest. However, in the whole-tree harvested plots, significantly lower concentrations (-1.9 mg per gram) were observed in the forest floor (but not the mineral soil) compared with the control. Reasons for this were not discussed but decomposition is likely to be responsible for the lower N concentrations.

Long-term (15-16 years) changes were studied by Olsson et al. (1996a) who measured decreases in total N pools in the forest floor (FH) at four coniferous sites regardless of harvest method (WTH and CH). However, the largest decreases were found at whole-tree harvested sites, which are not in agreement with the results of Nykvist (1977a). In contrast, mineral soil (0-20 cm) pools increased at most sites. Plant uptake, leaching to lower lying horizons, denitrification and immobilization by decomposing roots was discussed as possible mechanisms.

Katzensteiner (2003) did not observe any N concentration differences in the forest floor between a clear-cut and a 10-year old regenerating stand and a 140-year old mature forest, respectively, after stem-only harvest. In addition, Smethurst & Nambiar (1990) did not observe any significant short-term (3.25 years) harvest effects on mineral soil pools after stem-only and whole-tree harvest. Simard et al. (2001) did not find any short-term (5 years) harvest effects on total N concentrations in stem-only plots compared with control plots. On the other hand, higher total N concentrations (+1.4 mg per gram) and pools (+840 kg per hectare) were observed 17-22 years after whole-tree harvest. No explanation was given but forest floor dry mass and carbon content increased as a result of input from logging residues, root litter and from the new forest stand. Input from logging residues seems, however, unlikely since clear-cutting was performed as whole-tree harvest. Increases was also reported by Belleau et al. (2006) who measured a short-term increase (2 years) in total N concentration of +2.68 mg per gram in the forest floor after stem-only cutting of an aspen stand. No effects were seen in the mineral soil (0-10 cm). Leaching from logging residues left on site could explain the increased N concentrations.

No treatment effects were found on total N concentrations in the forest floor or mineral soil 16-18 years after whole-tree harvest compared with pre-harvest conditions in a Canadian study (McLaughlin & Phillips, 2006). However, higher forest floor N pools were found both in the control forest (+275 kg per hectare) and in the regenerating forest (+237 kg per hectare) compared with pre-harvest conditions but the differences between the regenerating area and control were non-significant in terms of concentrations and pools. Thus, soil N pools increased with time but not as a result of cutting. Instead, the increase was explained by a combination of factors such as atmospheric deposition, plant uptake of N from the mineral soil, redistribution to the forest floor via litterfall and decomposition and N fixation by feather moss and cyanobacteria. These results are in agreement with those of Johnson & Todd (1998) who measured long-term (15 years) increases in mineral soil (A, E and Bt) total N concentrations after WTH and CH of a mixed oak forest compared to pre-cutting values. Forest floor data is missing. Similar factors to those identified by McLaughlin & Phillips (2006) were discussed, but the input via these sources was considered too small. Symbiotic fixation of N by black locust trees and legumes seemed unlikely. Instead N_2-fixation of free-living bacteria was put forward as the most likely explanation. Thus, with time, total N pools may increase as a result of natural factors similar to soil C.

Short- and long-term treatment differences between WTH and CH are usually non-significant (Johnson & Todd, 1989; Belleau et al., 2006). But, Hendrickson et al. (1989) reported higher N total concentrations in the forest floor but not mineral soil in WTH plots compared with CH, three years after harvest. In contrast, Wall (2008), Olsson et al. (1996a) and Nykvist (1977a) observed lower forest floor pools varying between -509 to -77 kg per hectare. These treatment differences were still measurable after 15-16 years (Olsson et al., 1996a) but not after 26-28 years (Olsson et al., 2004). Treatment differences in the mineral soil are usually non-significant (Wall, 2008; Olsson et al., 1996a; Olsson et al., 2004). The short- and medium-term differences are normally related to the removal of logging residues.

To summarize, the concentration and pools of total N seem to decrease following clear-cutting as a result of decomposition, leaching losses, reduced plant uptake, denitrification and immobilization by decomposing roots. Mechanical disturbances during harvest and increases in soil temperature and moisture may accelerate decomposition rates thereby enhancing losses. In addition, elevated groundwater tables or snow accumulation may increase leaching losses. Short-term increases in the forest floor may be associated with accumulation of logging residuals or invasion of weeds and in the mineral soil due to retention in podzolic B-horizons. With time, as the forest floor begins to build up and the new forest is established, total N concentrations and pools starts to increase as a result of deposition, N-fixation and plant uptake from deeper soil horizons which accumulates in the forest floor via litter input. Treatment differences between WTH and CH are usually non-significant but the removal of logging residues may decrease forest floor N pools in the short- and medium term.

Treatment effects on total N include:
- N deposition
- N fixation
- Litter input
- Decomposition
- Leaching
- Plant uptake
- Retention
- Immobilization by decomposing roots
- Redistribution of N via plant uptake from deeper soil horizons

8.5 Base cations (Ca^{2+}, Mg^{2+} and K^{+})

The base cation results are complicated to evaluate and summarize due to the large variation of analytical methods used especially regarding the soil matrix. In our opinion, it is important to separate between concentrations and pools; or fluxes regarding stream water. A concentration decrease does not necessarily lead to a reduction in soil pools, since an increased soil density might compensate for the lower concentration. Mixing of the humus layer with mineral soil, a common disturbance during harvest is an example of this. Additionally, the exchangeable ions constitute a share of the total presence of an element, which means that harvest can affect the two fractions differently. In order to minimize the possibilities of false conclusions, we will separately discuss concentrations and pools of exchangeable and total fractions in soils as well as concentrations and fluxes in stream water.

The sources of base cations are primarily mineral weathering, atmospheric deposition and redistribution within the ecosystem due to internal nutrient cycling. Besides being important nutrients, base cations buffer against soil and surface water acidification (chapter 4). Nutrient losses via whole-tree harvesting have historically been addressed as a method by which soil nutrient pools are depleted thereby risking a loss in productivity (e.g. Rennie, 1955; White, 1974; Kimmins, 1977; Hornbeck & Kropelin, 1982). The loss of base cations has also raised a concern in terms of surface water acidification (the Swedish Environmental Protection Agency, 2007a). In areas subjected to acid deposition and loss of buffering capacity, whole-tree harvest may superimpose the negative effects of acid deposition. However, according to a review by Binkley & Högberg (1997), mineral weathering rates “appear to be greater than expected” suggesting that the loss of base cations may be partially or fully compensated for via weathering.

8.5.1 Decomposition and mineralization effects

After clear-cutting, nutrients continues to be released via decomposition of organic matter. Decomposition and nutrients release from various logging residues (stems, roots, branches, needles and leaves) after cutting are well-studied (e.g. Hyvönen et al., 2000; Arthur et al., 1993; Fahey et al., 1988; Fahey et al., 1991; Fahey & Arthur 1994; Palvainen et al., 2004b). The results show that nutrient rich material (leaves and needles) is a primary source of nutrients followed by more woody material (branches and roots) (e.g. Hyvönen et al., 2000). Potassium is quickly leached from the residues compared with other nutrients (e.g. Fahey et al., 1991; Fahey & Arthur 1994; Palvainen et al., 2004b).

It is scientifically debated whether decomposition of organic matter occurs more rapidly after clear-felling as a result of increased soil temperature and moisture conditions as put forward by e.g. Bormann & Likens, 1979, Martin & Pierce, 1980 and Martin et al., 2000. A number of studies have shown that decomposition and mineralization rates can decrease or remain indifferent compared with mature control forests (Yin et al., 1989; Prescott, 1997; Palvainen et al., 2004a; Cortina & Vallejo, 1994; Belleau et al., 2006). Prescott et al. (2000) studied temperature and moisture effects on the litter and forest floor decomposition rates in clear-cuts compared with mature forest across 21 sites in British Columbia, Canada. After four years, they did not find any evidence of increased decomposition rates on the clear-cuts. Instead a significant slower decomposition rate was found for pine needles on the clear-cuts compared with the uncut forest. Similar result was found in Catalonia, Spain after clear-cutting a *Pinus radiata* stand as a result of increased soil temperatures and decreased moisture content of the L-layer (Cortina & Vallejo, 1994). Thus, the drying out of the topmost soil layer as a result of increased sun exposure and wind speeds may in fact reduce decomposition rates after harvest. Prescott et al. (2000) concluded that “the suggestion that decomposition should be faster in clear-cuts appear to have its origin in speculation”.

Slower decomposition rates on clear-cuts may be consistent with the observed changes in concentration and leaching after harvest. For example, Prescott (1997) studied the effect of different harvesting techniques (clear-cut, patch cut etc.) on decomposition rates and N mineralization in Canada. During the first couple of years, decomposition rates were slower on clear-cut compared with uncut forest attributed to more moist soils during summer and perhaps a more undisturbed soil fauna and microflora. However, the concentration of NO_3^- and N mineralization rates were the greatest on the clear-cuts and successively slowed down in areas subjected to patch cut, shelter wood etc. and mature forest. According to these results, NO_3^- leaching from traditionally clear-cut areas would have the highest potential for nitrate leaching. Prescott (1997) explained this somewhat contradictory result by a reduction of available C, resulting from reduced litter inputs, leading to a reduction in soil microbial biomass and a build-up of ammonia susceptible to nitrification. Additionally, the microbes would assimilate less NO_3^-, favouring NO_3^- leaching.

8.5.2 Exchangeable concentrations in soils

Katzensteiner (2003) observed a short-term (≤5 years) increase in exchangeable Ca^{2+} in the forest floor in a chronosequence study between a clear-cut stand and a 10-year old regenerating stand in France. The increase was explained by mechanical and biological mixing of surface rich calcareous debris with the forest floor during stem-only harvest, a situation not likely to appear for most forest soils in Sweden. Medium- and long-term (≥15 years) decreases in Ca^{2+} at the same site between the regenerating stand and a 140-year old mature forest were not explained.

Mroz et al. (1985) reported increases in exchangeable base cation concentrations in the forest floor after whole-tree harvest of three podzolic hardwood sites in the USA. Compared with pre-harvest conditions, the concentration of Ca^{2+}, Mg^{2+} and K^+ increased by +105 to +299, +16 to +41 and +5 to +10 $cmol_c$ per kg, respectively, two and a half year after cutting. Further down in the mineral soil (E, Bh, Bs and BC), the concentration of exchangeable K^+ decreased, while data for Ca^{2+} and Mg^{2+} are not included. Reasons for the increased respectively decreased concentrations were not given. However, forest floor and mineral soil pools of K^+ decreased after whole-tree harvest, which was explained by cation exchange processes with H^+, released during nitrification and substituting K^+ on the exchange sites. Changes in Ca^{2+} and Mg^{2+} pools showed no consistent pattern but in general, the mineral soil pools decreased as a result of leaching losses and soil disturbances.

Belleau et al. (2006) compared post-treatment differences between a whole-tree harvested plot and a control forest (WTH↔CTR) and observed higher concentrations of Ca^{2+} and Mg^{2+} in the forest floor (but not K^+), two years after cutting of an aspen hardwood forest. No differences were found in the mineral soil. According to Belleau et al. (2006) the slash reduced the acidity of the forest floor and increased the availability of Ca^{2+} and Mg^{2+}. In contrast to these studies, Johnson et al. (1991b) observed decreased concentrations of exchangeable base cations both in the forest floor (Oa) and in the upper part of the mineral soil (E-horizon) three years after whole-tree harvest. The base cations were retained in the lower spodic B-horizon. According to Johnson et al. (1991b), podzols may thus limit the nutrient leaching following harvesting. The concentration increase in the mineral soil was explained by an increase in CEC by 30-60% (chapter 8.3). Despite these changes, Johnson et al. (1991b) concluded that the total effect of whole-tree harvesting on soil base cation pools was little, despite the large removal of nutrients in harvest and via stream water leaching. However, whole-tree harvest did lead to a redistribution of exchangeable cations in the soil.

Several other authors also report increases in mineral soil exchangeable concentrations based on results from pre- and postharvest studies. For example, Ranger et al. (2008) reported short-term increases in exchangeable concentrations of Ca^{2+} (+0.71 $cmol_c$ per kg), Mg^{2+} (+0.12 $cmol_c$ per kg) and K^+ (+0.09-0.16 $cmol_c$ per kg) in the 0-5 cm mineral soil layer up to three years after WTH. Decomposition of the forest floor and downward transport of organic matter could explain this increase. Further down in the mineral soil (5-10, 10-15, 15-30, 30-45 and 45-60 cm) statistically significant decreases in Ca^{2+}-, Mg^{2+}- and K^+ concentration were often seen, especially in the lowest lying soil layers. It was suggested that H^+ released during nitrification, although the production of NO_3^- decreased after cutting, was partly responsible for the decrease via cation exchange. The soil system (forest floor plus 0-60 cm mineral soil) lost -155, -44 and -162 kg per hectare of Ca, Mg and K, respectively.

Johnson & Todd (1998) measured a short- and medium-term (up to 15 years) increase of exchangeable Ca^{2+} (+1.18 and +0.36 $cmol_c$ per kg) and Mg^{2+} (+0.14 $cmol_c$ per kg) in the A- and E-horizon (only Ca^{2+}) compared with pre-cutting values after stem-only harvest of a mixed oak forest. The concentrations of K^+ remained unaffected. In contrast, the Ca^{2+}- and Mg^{2+} concentrations did not change after whole-tree harvest while the concentration of exchangeable K^+ increased (+0.03 $cmol_c$ per kg) in the Bt-horizon compared with pre-harvest data. Johnson & Todd (1998) explained the increase in Ca^{2+} concentrations in the mineral soil with time as a result of an accumulation from decomposing logging residues left on site after stem-only harvesting since the increase in Ca^{2+} were equal to the amount of Ca^{2+} released by logging residues. Increases in Mg^{2+} and K^+ were not explained. Based on these results, Johnson & Todd (op.cit) concluded that WTH does not lead to a decrease in site productivity and that Ca^{2+} is efficiently retained in the soil.

Studies describing long-term effects are rare but available from some studies where both short- and long-term differences are described. For example, Simard et al. (2001) measured short-term (3-5 years) post-treatment effect of CPRS (Cut with Protection of Regeneration and Soils) with data from a control plot (CH↔CTR). In general, higher base cation concentrations were found in the clear-cut area. For example, the concentration of exchangeable Ca^{2+} in the forest floor was +2.4 $cmol_c$ per kg higher in the clear-cut compared with the control forest but no difference was found for Mg^{2+} and K^+. However, in the mineral soil (0-10 cm) all three base cations were higher by +0.18 (Ca^{2+}), +0.05 (Mg^{2+}) and +0.03 (K^+) $cmol_c$ per kg. Simard et al. (2001) also studied the long-term effect (11-16 years and 17-22 years) of whole-tree harvesting compared with a control forest (WTH↔CTR). Higher concentrations of Ca^{2+} and Mg^{2+} were found both in the forest floor and mineral soil while forest floor K^+ concentration was lower compared with the control plot. Simard et al. (2001) explained the increased base cation concentrations by mineralization and decomposition of fresh organic matter demonstrated by higher dry mass and organic C content in the forest floor. This would certainly explain concentration increases in the stem-only but not the whole-tree harvested area.

In contrast to these results, Knoepp & Swank (1997) measured a short-term (<3 year) increase in exchangeable Ca^{2+} concentration in the 0-10 cm (but not the 10-30 cm) mineral soil layer after stem-only harvest. The concentration of Mg^{2+} and K^+ also increased immediately after cutting both in the 0-10 and 10-30 cm layer and were still greater compared with pre-harvest conditions 17-20 years after harvest. The short-term increases in cation concentrations (Ca^{2+}, Mg^{2+} and K^+) immediately following harvest were probably the result of leaching from fresh easily decomposable organic material in the logging debris. With time larger and woodier debris (branches and roots) become a nutrient source.

In a Canadian study by McLaughlin & Phillips (2006), long-term soil chemical changes in a 16-18 year old regenerating stand were studied. The results were compared with pre-harvest conditions and an uncut 77-85 year old reference stand. Forest floor concentrations of exchangeable Ca^{2+} and K^+ did not change after whole-tree harvest compared with preharvest values. However, the concentration of exchangeable Mg^{2+} decreased significantly by -3.76 $cmol_c$ per kg. Mineral soil concentrations of exchangeable Mg^{2+} and K^+ were largely unaffected by harvesting while the concentration of Ca^{2+} increased with +0.73 $cmol_c$ per kg. When comparing the data with the uncut reference stand no differences could be seen in forest floor Ca^{2+} concentration but the concentration of Mg^{2+} and K^{2+} were lower in the regenerating stand by -0.59 and -0.49 $cmol_c$ per kg, respectively. In the mineral soil, exchangeable concentrations of Ca^{2+} and Mg^{2+} were higher by +0.69 and +0.12 $cmol_c$ per kg, respectively, while K^+ remained unaffected. The higher

Ca^{2+} concentration found in the 16-18 year old stand were explained by differences in weathering and decomposition rates, higher nutrient uptake and a greater circulation of Ca^{2+} between the trees and soil. The loss of Mg^{2+} with time at both the regenerating stand and reference stand were explained by acid atmospheric deposition, which induced Mg^{2+} depletion (but not Ca^{2+}) at these sites. Differences in sensitivity between the sites could explain why the Mg^{2+} concentrations were higher in the regenerating stand compared with the reference stand, despite the decrease with time. In spite of these changes, the exchangeable base cations pools remained unaffected by whole-tree harvesting (see below).

In the following text, treatment differences between harvest intensities (WTH and CH) are reported. In general, the differences between whole-tree and stem-only harvest (WTH↔CH) are non-significant in terms of exchangeable base cation concentrations (Table 13). However, lower concentrations of Ca^{2+}, Mg^{2+} and K^+ have frequently been observed both in the forest floor and in the mineral soil (0-10 and ≥10 cm) after whole-tree harvest compared with stem-only harvest (e.g. Johnson & Todd, 1998; Bélanger et al., 2003; Thiffault et al., 2006; Nykvist & Rosén, 1985; Belleau et al., 2006) for periods ranging between 3-21 years. The mechanism responsible for the observed differences is a relocation of base cations from decomposing logging residues to the soil exchange complex in stem-only harvested plots, which creates a measurable difference with whole-tree harvested plots (e.g. Nykvist & Rosén, 1985; Belleau et al., 2006).

However, the individual base cations may respond differently to the treatments. For example, Bélanger et al. (2003) measured significantly lower exchangeable concentrations of Mg^{2+} (-0.35 $cmol_c\ kg^{-1}$) in the forest floor three years after whole-tree harvest compared with stem-only harvest. Similar differences, although not statistically significant, were found also for Ca^{2+} and K^+. In the mineral soil (0-20 cm, ~B-horizon) a statistically significant difference was found for exchangeable K^+ (-0.013 $cmol_c\ kg^{-1}$) but not for Ca^{2+} or Mg^{2+}. The explanations for this Ca^{2+} and Mg^{2+} lack of response include differences in the release of nutrients from decomposing logging residues, lower ion mobility in the soil and a low contribution of Ca^{2+} and Mg^{2+} to the soil solution ionic activity.

These results are in agreement with Johnson & Todd (1998) who measured lower concentrations of Ca^{2+}, Mg^{2+} and K^+ in the mineral soil, 15 years after whole-tree harvest compared with stem-only, attributed to release of base cations from decomposing slash. Thiffault et al. (2006) compared long-term (15-20 years) treatment differences between CH and WTH at three sites in Canada. After whole-tree harvesting, lower concentrations of Ca^{2+} were found in the forest floor but not the mineral soil (0-20 cm) at all the three study sites. Lower concentrations of K^+ in the mineral soil were also found at one of the sites, while the concentration of Mg^{2+} remained unaffected at all three sites. In addition to these results, lower CEC in the forest floor and mineral soil was found at the whole-tree harvested sites compared with the stem-only harvested. Thiffault et al. (2006) concluded that the slash left on site after stem-only harvest was a source of Ca^{2+}, which was easily retained in the forest floor because of slow decomposition and low mobility. In contrast, K^+ was easily leached from the system and not retained in the soil, which explained why no differences were seen in the forest floor concentrations between WTH and CH plots. Some of the leached K^+ was however retained further down in the mineral soil.

8.5.3 Total concentrations in soils

Harvest effects on the total concentrations of base cations are less well described in the literature. Nevertheless, just like the concentrations of exchangeable base cations, total concentrations seem to increase in the soils following harvest. For example, in the chronosequence study by Katzensteiner (2003), the total concentration of Ca^{2+}, Mg^{2+} and K^+ in the LF-layer increased by +6.3, +0.4 and +0.5 mg per gram, respectively, between a clear-cut and a 10-year-old plantation following stem-only harvest. In addition, the concentrations of Ca^{2+} also increased in the humus layer by +15.6 mg per gram. Reasons for this were not discussed. However, input from decomposing litter and decaying roots could be responsible as thickness and dry mass of the LF-horizon decreased during this time. In addition, input via weathering could have been a potential source as proposed by McLaughlin & Phillips (2006).

Hendrickson et al. (1989) and Simard et al. (2001) also observed higher BC concentrations following cutting compared with control plots. For example, Hendrickson et al. (1989) measured higher concentrations of Ca^{2+}, Mg^{2+} and K^{+}, especially in the upper part of the soil, three years after whole-tree and stem-only harvest using an uncut forest stand as a control. No reasons were given but the total soil organic matter content increased during the same time, which could explain the higher values. Simard et al. (2001) studied the short- and long-term effects of clear-cutting (WTH and CH) on soil chemical properties at three study sites in Canada. Following cutting, the total concentration of Ca^{2+} was higher at all three sites regardless of harvest intensity compared with control plots. In addition, the total concentrations of Mg^{2+} were higher at two of the sites while K^{+} remained unaffected. The exchangeable concentrations (see above) and pools of total Ca^{2+} and Mg^{2+} in the forest floor increased in a similar way. No explanation was given.

When data from a 10 years old stand were compared with a 140 years old mature forest Katzensteiner (2003), the total concentrations of Ca^{2+} and Mg^{2+} (but not K^{+}) decreased with time. No explanation was given for this difference. In contrast, Belleau et al. (2006) studied soil chemical changes following stem-only harvest of an aspen stand in Canada. Compared with a control plot of growing forest, no differences were observed between the two plots except for a small decrease in total K^{+} concentration in the forest floor by -0.1 mg per gram during the first year after harvest. During the second year, this difference had disappeared. In the same study, the concentration of exchangeable Ca^{2+} and Mg^{2+} increased (see above). A decrease in total concentration of Ca^{2+} (-46%) was also reported by Brais et al. (1995), 5-12 years after whole-tree harvest in moist forest sites compared with control plots. At the same time, the pools of total Ca^{2+}, Mg^{2+} and K^{+} in the forest floor decreased significantly by 67% (-900 kg per hectare), 48% (-143 kg per hectare) and 40% (-136 kg per hectare), respectively. On dry to fresh soils, the total pool of Ca^{2+} decreased by 61% (-459 kg per hectare) while no significant change was seen for total Mg^{2+} and K^{+}. The decreases in total pools of base cations were mainly attributed to a loss of substrate since the mean forest floor ash free dry weight decreased by 44% (36.4 tonnes per hectare) at moist sites.

Post treatment differences between whole-tree and stem-only harvest (WTH↔CH) in terms of total base cation concentrations are seldom reported in the literature. However, according to Belleau et al. (2006) the total concentration of Ca^{2+}, Mg^{2+} and K^{+} in the forest floor and in the upper mineral soil (0-10 cm) did not differ between whole-tree and stem-only harvested plots during the first and second year after cutting of a hardwood forest. In contrast, Hendrickson et al. (1989) reported lower total concentrations of Ca^{2+}, Mg^{2+} and K^{+} in the forest floor three years after harvest (WTH↔CH). Further down in the mineral soil (0-5, 5-10 and 10-20 cm), the total K^{+} concentrations were lower in the 10-20 cm soil layer. No other treatment differences could be found.

8.5.4 Exchangeable pools in soils

In general, the harvest effects on forest floor and mineral soil exchangeable base cation pools are non-significant compared with pre-harvest conditions and uncut control plots (e.g. Belleau et al., 2006). But, there are reports of both increased and decreased pools after harvest for individual soil horizons and cations. In a study by Mroz et al. (1985), the nutrient content in forest floor and mineral soil (down to 100 cm) were compared prior to and after whole-tree harvest of three hardwood sites in the US. Compared with preharvest conditions and two and a half years after cutting, the pools of K^+ decreased throughout the soil profile at all three sites between -1114 to -883 and -26 to -6 kg per hectare in the forest floor and mineral soil, respectively. The concentrations of exchangeable K^+ changed in a similar way (see above) and it was assumed that the losses were due to a displacement of K^+ with H^+ (released during nitrification) on the exchange complex. Some losses were also seen in the pools of exchangeable Ca^{2+} and Mg^{2+} associated with leaching and mechanical disturbance of the forest floor during harvest. The great losses reported by Mroz et al. (1985) have been questioned by Binkley (1987) and commented on by Mroz et al. (1987).

Romanowicz et al. (1996) measured increases in mineral soil (down to the C-horizon) exchangeable pools of K^+ during the first three years following WTH of a high elevation spruce fir forest in the USA. An increase was also seen in the forest floor (Oa-horizon) although not statistically significant. Increases in soil pools, compared with pre-harvest conditions, were attributed to an increased retention of K^+ on the exchange complex. Eight years after harvest, these differences had disappeared. Increases in mineral soil pools of K^+ were also seen in the lower parts of the catchment although not significant. At one site, a decrease in the forest floor K^+ pool was documented between the third and eight year after harvest, but the pools were still higher compared with pre-harvest conditions. The results of Romanowicz et al. (1996) indicate that mineral soil pools of K^+ increases following cutting but seem to return to pre-harvest conditions with time.

Johnson et al. (1991b) also reported increased mineral soil pools of K^+ three years after whole-tree harvest. However, the pools had returned to pre-harvest conditions eight years after harvest (Johnson et al., 1997). The increase was consistent with increased concentrations of exchangeable K^+ and CEC_{eff}. The pools of Ca^{2+} and Mg^{2+} did not change during this time. Using a chronosequence approach, Snyder & Harter (1984) measured initial increases in mineral soil pools of K^+ throughout the soil profile (E, Bh, Bs1 and Bs2) after stem-only harvest. Increased pools of Ca^{2+} and Mg^{2+} were also seen in the upper part of the soil. These increases were associated with decomposition and mineralization of logging residues, which were leached down into the mineral soil where they were retained. Although not statistically significant, both Ca^{2+} and Mg^{2+} appeared to increase with time (up to 30 years after harvest) whereas K^+ started to decrease between 10-30 years after harvest.

Nykvist (1977b) measured differences in the pools of exchangeable K^+ with time after stem-only and whole-tree harvested plots in Sweden. In general, the pools of K^+ decreased four years after cutting both in the forest floor and mineral soil (0-20 cm) although the results were not statistically tested. Also, the pools became lower following whole-tree harvest compared with stem-only. The release of K^+ from logging residues could account for the observed differences. Brais et al. (1995) also reported significant declines in base cation exchangeable pools 5-12 years after whole-tree harvest associated with a substrate loss. McLaughlin & Phillips (2006) assessed long-term (16-18 years) differences in soil exchangeable pools in Canada. Compared with pre-harvest conditions, the forest floor pools of exchangeable Mg^{2+} decreased by 37 kg per hectare after whole-tree harvest. When tested alone, no differences were found in the mineral soil but when the results were combined with forest floor data a significant decrease was found for the total soil (forest floor+mineral soil) pools of Mg^{2+} (-7.2 kg per hectare). A similar trend in the uncut control indicated a loss of Mg^{2+} due to the acid deposition and not to the biomass harvest per se. In contrast to these results, the pool of exchangeable Ca^{2+} increased by 603 kg per hectare in the mineral soil compared with pre-cutting values. Compared with the uncut control stands, higher pools were also found in the mineral soil (+546 kg per hectare). McLaughlin & Phillips (2006) suggested that increased mineral weathering due to whole-tree harvest was responsible for this increase. Apparently, whole-tree harvesting does not necessarily lead to losses in the exchangeable base cation pools but gains can also occur.

Generally, there is no statistically significant difference in exchangeable base cation soil pools between whole-tree and stem-only harvested plots (WTH↔CH). However, decreased pools have frequently been reported, both in the forest floor and in the mineral soil associated with the removal of logging residues. For example, Olsson et al. (1996b) reported significantly lower pools of exchangeable Ca^{2+}, Mg^{2+} and K^{+} in the forest floor and mineral soil (down to 20 cm) in whole-tree harvested plots compared with stem-only, 15-16 years after cutting. The largest differences took place in the uppermost soil layers. A relative large recovery of Ca^{2+} in the soil from decomposing logging residues in stem-only harvested plots explained the difference between WTH and CH. In contrast, K was easily leached from the system. In a follow-up study by Olsson et al. (2004), 26-30 years after cutting, the treatment differences could still be observed.

Wall (2008), Nykvist (1977b) and Nykvist (1990) also reported lower exchangeable soil pools. According to Wall (2008), removal of logging residues decreased the pools of base cations, especially K^{+}, in the L, FH and 0-10 cm soil layer four years after harvest. Nykvist (1990) reported long-term exchangeable K^{+} pool differences, 10 and 16 years after final felling, at two sites in Sweden. Compared with stem-only harvested plots, whole-tree harvested plots had lower pools of K^{+} in the humus layer at both sites. Lower mineral soil pools (0-20 cm) were also found at the southern site compared with the northern site. However, these differences decreased with time.

8.5.5 Total pools in soils

Ranger et al (2008) described short-term treatment effects on the total base cation pools following whole-tree harvest of a Douglas fir plantation in France. During the first three years after cutting the pools of total Ca^{2+}, Mg^{2+} and K^{+} decreased by -144, -28 and -47 kg per hectare, respectively, in the forest floor and by -11, -16, -115 kg per hectare, respectively, in the mineral soil. These changes were due to a loss of substrate in the forest floor and acidification in the mineral soil. In Sweden, Nykvist (1977a) observed a short-term decrease in the forest floor pools of total K^{+} and an increase in total Ca^{2+}, both after whole-tree and stem-only cutting compared with pre-harvest data. In contrast, the pools of total Mg^{2+} were indifferent following cutting. Despite these varying results, caused by humus layer heterogeneity, Nykvist (19977a) concluded that the general effect of harvest is decreased base cation pools. On the contrary, Simard et al. (2001) observed higher total pools of Ca^{2+} (+270 kg per hectare) and Mg^{2+} (+35 kg per hectare) in the forest floor, three to five years after stem-only harvest compared with an uncut control as a result of higher dry mass caused by litter input. Similar results were found when comparing whole-tree harvested plots and control plots 11-16 and 17-22 years after harvest at two other sites.

Post treatment differences between harvest intensities (WTH↔CH) on total pools of base cations are reported by Wall (2008), who measured lower pools of total Ca^{2+} (-387 to -64 kg per hectare), total Mg^{2+} (-35 to -6 kg per hectare) and total K^{+} (-34 to -6 kg per hectare) in the forest floor (litter layer) after WTH compared with stem-only, 4 years after final felling. No treatment differences were found in the humus layer (F+H) or mineral soil (0-10 cm). Nykvist (1977a) also reported lower total pools in the forest floor. During the first year after harvest the pools of base cations were lower following WTH compared with CH ranging between -213 to -207 (Ca^{2+}), -25 to -16 (Mg^{2+}) and -52 to -32 (K^+) kg per hectare. During the fourth year the differences were less; -150 to -37 (Ca^{2+}), -7 to -1 (Mg^{2+}) and -13 to 0 (K^+). Hence, the largest losses seemed to occur for Ca^{2+}.

8.5.6 Concentrations in soil water, groundwater and stream water

Short-term (≤5 years) treatment effects on base cation concentrations in soil water and stream water are well described in the literature while data from groundwater studies are largely lacking. In general, the concentrations of Ca^{2+}, Mg^{2+} and K^+ increase in response to harvest (CH and WTH) in soil water (Hornbeck & Kropelin, 1982; Dahlgren & Driscoll, 1994; Romanowicz et al., 1996; Titus et al., 1997; Hendrickson et al., 1989; Briggs et al., 2000; Katzensteiner, 2003; Piirainen et al., 2004) ground water (Wiklander, 1974) and stream water (Bormann et al., 1968; Likens et al., 1970; Wiklander, 1974; Martin & Pierce, 1980; Grip, 1982; Tiedemann et al., 1988; Mann et al., 1988; Adamson & Hornung, 1990; Jewett et al., 1995; Kubin, 1995; Stevens et al., 1995; Martin et al., 2000; Neal et al., 2004; Wang, 2006; Löfgren et al., 2009b; Tremblay et al., 2009) compared with pre-harvest conditions or controls.

Changes in base cation concentrations may result from disturbances of the internal nutrient cycling or via changes in the weathering rates as well as in the atmospheric deposition of base cations and mineral acid anions. According to the mobile anion theory (Reuss & Johnson, 1986), changes in SO_4^{2-}, NO_3^- and Cl^- concentrations change the leaching of base cations. These processes may work independently or they may interact with each other. For example, during mineralization of organic N and nitrification of ammonia to NO_3^-, H^+ is produced. An increase in H^+ may increase weathering rates or they may participate in cation exchange processes, causing increased BC concentrations in the soil solution. Together with NO_3^-, base cations are easily leached from the system (Bormann et al., 1968; Grip, 1982; Löfgren et al., 2009b). Dahlgren & Driscoll (1994) measured higher soil water NO_3^- concentrations both in the forest floor (Oa) and in the mineral soil (Bhs and Bs2) five to six years after whole-tree harvest compared with uncut control plots. The H^+ released during nitrification were largely neutralized via aluminium buffering and cation exchange processes releasing Al^{3+}, Ca^{2+}, Mg^{2+} and K^+ to the soil solution. Measurements at the catchment outlet showed that the released cations where leached together with NO_3^- from the soil to the stream water. Ten years later, the concentration of K^+ in soil water and stream water remained elevated as a result of decomposition, mineralization and weathering (Romanowicz et al., 1996).

In comparison, Wiklander (1974) measured increased concentrations of Ca^{2+} and Mg^{2+} but not K^+ in a spring, two years after stem-only cutting of a softwood stand in Sweden. No reasons were given but the concentration of NO_3^- increased during the same time, probably affecting the Ca^{2+} and Mg^{2+} mobility. The lack of response in K^+ concentrations is more difficult to explain since increased stream water K^+ concentrations were found in another clear-cut area as a result of litter leaching (op. cit.). It is thus possible that K^+ was leached from the forest floor but was retained further down in the mineral soil. With time as revegetation takes place, NO_3^- concentrations and associated base cation leaching decrease (Hornbeck & Kropelin, 1982; Hendrickson et al., 1989; Briggs et al., 2000; Katzensteiner, 2003).

Disruptions in the internal nutrient cycling following harvest include reduced uptake by vegetation and microbial community of nutrients from fresh litter, roots and woody debris (Bormann et al., 1968; Rosén et al., 1996; Titus et al., 1997). Lawrence et al. (1987) explained increased stream water concentrations of Ca^{2+}, Mg^{2+} and K^+ by reduced plant uptake and nitrification, one year after whole-tree harvest compared with an uncut control forest. Since K^+ is loosely held in plant tissue it is readily leached from fresh litter compared with Mg^{2+} and Ca^{2+} (Staaf & Berg, 1982; Titus & Malcolm, 1999). For example, Stevens et al. (1995) studied the effects of stem-only and whole-tree harvest in soil water and stream water five years after cutting. Following felling, the concentrations of K^+ increased in soil water and stream water in the stem-only harvested plots as a result of leaching from the logging residues. However, not all of the released K^+ reached the stream, indicating soil retention. Moreover, the concentration of NO_3^- increased which induced K^+ leaching. On plots where logging residues had been collected, no K^+ increase occurred.

Increased stream water concentrations of K^+ as a result of litter leaching have also been documented by Wiklander (1974), Adamson et al. (1987), Adamson & Hornung (1990), Tiedemann et al. (1988), Kubin (1995) and Löfgren et al. (2009b). In Canada, Tremblay et al. (2009) monitored stream water concentrations before and after (2 years) stem-only cutting of five catchments. The results were compared with unharvested control forests. Following clear-cutting, the concentrations of Mg^{2+} and K^+ increased relative to the control. During the same time, the concentration of NO_3^- increased while pH decreased slightly. Tremblay et al. (2009) believed that nitrification caused some of the increased cation concentrations and mobility, but reduced plant uptake was a more likely factor since the concentration of Mg^{2+} also increased. Short-term losses of NO_3^- and Ca^{2+} as a result of decomposition, reduced uptake and leaching from the system has also been reported by Martin and Pierce (1980) following clear-cutting of seven catchments in the USA. In general, the largest losses occurred for entirely clear-cut catchments compared with partial clear-cut areas. After circa five years, the concentrations returned to pre-harvest conditions.

Increased soil water concentrations may not necessarily lead to leaching losses from the system. For example, Piirainen et al. (2004) observed increased concentrations of Ca^{2+}, Mg^{2+} and K^+ in the O-horizon three years after stem-only harvest compared with a control forest as a result of litter leaching and decomposition. However, most of this was retained in the mineral soil via adsorption. Jewett et al. (1995) and Romanowicz et al. (1996) also discussed ion exchange processes and mineral fixation as factors explaining decreased K^+ concentrations. These results are in agreement with groundwater studies from Finland where base cation concentrations in two study areas remained unaffected five years after stem-only harvest when compared with controls (Mannerkoski et al., 2005). Reasons for the lack of response were not discussed. However, only 10-30 % of the catchments were clear-cut and most wells were located inside the aquifer but outside the harvested areas. In another Finnish study by Rusanen et al. (2004), no long-term (15-22 years) effects on base cation concentrations were seen following clear-cutting of a large aquifer compared with pre-harvest conditions. On the other hand, the stream water concentrations may still increase despite decreasing soil water trends if e.g. the weathering rates increase as indicated by results from McLaughlin & Phillips (2006).

Swank et al. (2001) used a paired catchment study approach for monitoring the long-term (20 years) BC changes following stem-only cutting of a northern hardwood forest in the US. Compared with pre-harvest conditions and the control, higher concentrations of base cations and NO_3^- were found after cutting, especially during the third year. With time, as the new forest established, the concentrations of Ca^{2+} remained elevated. In contrast, the concentration of K^+ only increased during the first five years before declining to similar values as the control. Prior to cutting the roads were seeded with grass, limed and fertilized (NPK), which increased the concentrations of Ca^{2+} and K^+. After cutting, the concentration of Ca^{2+} increased further. In contrast to these ions, the concentrations of NO_3^- did not increase following N fertilization and prior to cutting due to in-stream processes, but increased after harvest and tree mortality 10-15 years after harvest. The high mortality reduced plant uptake and produced easily decomposable litter. However, a similar pulse was not seen for Ca^{2+} and K^+.

In another long-term (up to 27 years) study by Martin et al. (2000), significant increases in Ca^{2+}, K^+, NO_3^- and H^+ concentrations was observed immediately following stem-only and whole-tree harvest relative to a control forest. After three to five years, the concentrations returned to pre-harvest values. However, the concentrations of Ca^{2+} and K^+ remained elevated throughout the study period. Also, a second pulse of $NO_3^-$was observed in the whole-tree harvested catchment after the loss of short-lived pioneering trees. In the stem-only cut area no second pulse of NO_3^- was seen owing to the fact that measurements begun later resulting in a shorter (14 years) study period and consequently younger trees. Martin et al. (2000) did not discuss possible reasons for the elevated Ca^{2+} and NO_3^- concentrations. However, the increased NO_3^- concentrations during the first and second pulse (only in the WTH area) were probably causing some of the elevated Ca^{2+} and K^+ concentrations in the stream. The SO_4^{2-} concentrations, on the other hand, decreased immediately following cutting as a result of increased soil retention and returned to pre-harvest conditions by the 4th year after cutting.

Long-term (16-18 years) trends in soil water (collected at 50 cm depth) and stream water chemistry is also available from a whole-tree harvest experiment by McLaughlin & Phillips (2006). Compared with an uncut control catchment, short-term (0-3 years) increases in Ca^{2+} and NO_3^- concentrations were seen in the soil water at 25 and 50 cm depth immediately after harvesting (1981), after which the concentrations returned to pre-harvest conditions (see Briggs et al., 2000). However, during the 90's, the concentrations of Ca^{2+} and Mg^{2+} showed a decrease in the soil solution but an increase in stream water in the regenerating stand. Higher stream water concentrations may be a result of increased weathering and increased fluxes from the mineral soil. The decreasing trends in soil solution may be due to higher adsorption in the upper mineral soil (McLaughlin& Phillips, 2006).

Decreased base cation concentrations following harvest have also been shown for various other reasons. In a study by Ranger et al. (2007), the concentration of Ca^{2+}, Mg^{2+} and K^+ decreased following whole-tree harvest of a Douglas-fir forest. Decreases were observed throughout the soil profile (forest floor and mineral soil down to 120 cm) but the results varied slightly between the individual horizons depending on the different lysimeter equipments (zero-tension plate lysimeters versus tension-cup ceramic lysimeters). The decreased BC concentrations coincided with decreased concentrations of NO_3^-. Ranger et al. (2007) proposed that the close relationship between Douglas-fir trees and nitrifiers prior to cutting were disrupted following tree removal. This led to a decreased nitrification and subsequently a decreased NO_3^- production. Mann et al. (1988) also showed the importance of tree species and their response to cutting on the BC concentrations. Based on soil water data (≥50 cm) from eleven research sites, they showed that the concentrations of Ca^{2+} and K^+ increased as a result of increased NO_3^- concentrations. The increases took place both in softwood and hardwood forests regardless of cutting method (whole-tree and stem-only). However, harvested red alder stands were an exception to this since the NO_3^- concentrations decreased as the N-fixation ceased.

Robertson et al. (2000) obtained similar results when comparing soil water concentrations below clear-cut alder stands with harvested oak, spruce and pine stands. In New Zealand, Parfitt et al. (2002) measured decreased concentration of Ca^{2+}, Mg^{2+} and K^+ in the upper mineral soil solution (10 cm) three years after stem-only cutting of a softwood forest compared with pre-harvest conditions. No significant changes were seen in the lysimeters at a soil depth of 60 cm. Just like Ranger et al. (2007) and Mann et al. (1988), the decrease in base cation concentrations appeared at the same time as the concentration of NO_3^- decreased. A rapid colonization of grasses and weeds together with microbial immobilization were responsible for the decreased NO_3^- concentrations. In addition, reduced dry deposition primarily affecting the concentration of Cl^- but also NH_4^+ and NO_3^- could have been responsible for the decrease. Measurements in spring water and stream water within the same catchment showed that the concentrations of major anions and cations remained fairly constant except for a small decrease in Ca^{2+} and NO_3^- (relative to the spring) probably due to denitrification and vegetation uptake. Similar results were found by Titus et al. (1997) who studied clear-cutting effects in soil water at 50 cm depth at sites of varying site quality. Contrary to what would be expected, the greatest increase in NO_3^- and base cation concentrations did not take place at the rich site since this site was moist, favouring vegetation uptake and denitrification. Hence, processes either preventing nitrification or limiting the build-up of produced NO_3^- may limit the base cation leaching.

Decreased dry deposition and dilution may also be responsible for decreased base cation concentrations after harvest. For instance, Robertson et al. (2000) observed clear differences in soil water Ca^{2+} and Mg^{2+} concentrations in the L-, A- and C-horizons before and three years after stem-only harvest, attributed to a reduction in atmospherically derived ions and to some extent dilution. One would except that the largest reductions would have taken place on the coniferous sites compared with hardwood sites due to higher dry deposition. However, two years after harvest, soil water concentrations differed little between the sites (Robertson et al., 2000). Compared with Ca^{2+} and Mg^{2+}, the concentrations of K^+ in the mineral soil increased as a result of litter leaching and reduced plant uptake. The increase was however limited to the coniferous sites since the leaves were missing in the broadleaved stands at the time of felling.
Decreased sea salt deposition was responsible for decreased Ca^{2+} and Mg^{2+} concentrations in groundwater and stream water six years after stem-only harvest (Neal et al., 2004). Harvesting effects in terms of increased NO_3^- concentrations were also observed both in groundwater and stream water during the first couple of years after felling, thereafter declining. In addition, decomposition and reduced plant uptake also affected the stream water Ca^{2+}, Mg^{2+} and K^+ concentrations. Two years after clear-felling, the stream water Ca^{2+} and Mg^{2+} concentrations started to decline while the K^+ concentrations remained elevated compared with pre-harvest conditions and control. The decline corresponded to a gradual decrease in sea-salt derived ions.

Soil water, groundwater and stream water studies describing treatment differences between WTH and CH are scarce. In general, the leaching of cations is lower following whole-tree harvest compared with conventionally harvested areas where the logging residues are left on site. The presence or absence of slash most often explains the treatment differences between WTH and CH. For example, Staaf & Olsson (1994) measured lower soil water concentrations of K^+ and NO_3^- (collected at 30 cm) after whole-tree harvesting compared with stem-only plots. Litter leaching of K^+ and stimulated mineralization and nitrification releasing NO_3^- below piles of logging residues in stem-only harvested plots explained these differences as shown by Rosén & Lundmark-Thelin (1987). Four years after treatment, however, the differences between CH and WTH were no longer seen. Stevens et al. (1995) observed a short-term treatment difference following final felling of a Sitka spruce stand in Wales. Five years after harvest the soil water BC concentrations in the forest floor (L-layer) and mineral soil (C-horizon, 70 cm) after whole-tree harvest were lower compared with stem-only as a result of lower litter leaching and NO_3^- concentrations. Based on Swedish sites, Westling et al. (2004) observed lower concentrations of Ca^{2+}, Mg^{2+} and K^+ in the soil solution at 50 cm depth, 9-11 years after whole-tree harvest compared with stem-only harvest. Also based on Swedish data, Zetterberg et al. (2008) showed that such treatment effects might persist up to 27-31 years.

In contrast to these results, Wall et al. (2008) did not detect any significant treatment differences in soil water BC concentrations in the O-horizon four years after harvest, while Hendrickson et al. (1989) measured higher Ca^{2+} leaching in the forest floor and in the mineral soil (100 cm) from whole-tree harvested areas compared with conventionally harvested, two years after cutting. Higher concentrations of K^+ and Mg^{2+} were also observed in the forest floor and mineral soil, respectively. The cations were most probably released from the forest floor and leached to lower lying horizons. According to Hendrickson et al. (1989) it is also possible that plant uptake of fine roots in the forest floor was offset by the whole-tree harvesting. These differences had disappeared during the third year. Depending on site quality, Titus et al. (1997) measured, three years after final felling, a varied response in the soil water concentrations of Ca^{2+}, Mg^{2+} and K^+. In general, stem-only harvest increased soil water concentrations of Ca^{2+} at 50 cm depth as a result of litter leaching, nitrification and reduced plant uptake. However, stem-only harvesting resulted in lower K^+ respective Mg^{2+} concentrations at one out of three sites each. No explanation was given for these differences.

8.5.7 Fluxes in soil water and stream water

Soil water fluxes have been shown to both increase and decrease following cutting. For example, Ranger et al. (2007) measured short-term (3 years) increased fluxes of Ca^{2+}, Mg^{2+} and K^+ of +3.2, +1.4 and +6.1 kg per hectare, respectively, from the mineral soil at 60 cm, but not 120 cm, as a result of increased water fluxes. Further down in the mineral soil, base cation fluxes decreased as a result of decreased concentrations despite increased water fluxes. Stevens et al. (1995) observed decreased concentrations and fluxes of Ca^{2+} and K^+ in the O- and L-layers five years after whole-tree harvest compared with stem-only harvest. Increased plant uptake was suggested causing the decreased Ca^{2+} fluxes, while no explanation was given for the decreased K^+ fluxes. In Finland, Piirainen et al. (2004) measured a doubling in base cation fluxes from below the O-horizon to the E-horizon three years after stem-only cutting of a coniferous forest. The leached ions were effectively retained in the mineral soil (below B-horizon). According to Piirainen et al. (2004) decomposition and mineralization of slash (and K^+ from litter leaching) explained the flux changes. To our knowledge, data on groundwater fluxes are missing in the literature.

Base cation fluxes often increase in stream water (Likens et al., 1970; Haveraaen, 1981; Hornbeck et al., 1990; Ahtiainen, 1992; Stevens et al., 1995; Rosén et al., 1996; Ahtiainen & Huttunen, 1999; Swank et al., 2001; Löfgren et al., 2009b) following cutting as a result of increased runoff. As the new forest stand grows older evapotranspiration increases and fluxes return to pre-harvest levels. Increases in Ca^{2+}, Mg^{2+} and K^+ fluxes may last up to at least eight years after cutting. However, Adamson et al. (1987) observed increased fluxes of Ca^{2+}, Mg^{2+} and K^+ during the first year following stem-only harvest compared with an unharvested control. In the second year, the BC fluxes decreased which Adamson et al. (1987) attributed to decreased fluxes of SO_4^{2-} and Cl^-. In order to maintain electro neutrality, any change in the anion flux has the potential to affect the BC flux. The reduced sulphate deposition since the 1980’s has reduced the BC fluxes in boreal, Swedish streams (Löfgren et al. 2009a).

8.5.8 Summary of harvest effects on base cations in soils and water

Generally, the exchangeable base cation (Ca^{2+}, Mg^{2+} and K^+) concentrations in the forest floor and upper mineral soil increase following clear-cutting. Decomposition of fresh litter and leaching from logging residues left on ground is an important source. Increased soil moisture facilitates a downward movement of organic matter and base cations in soil solution. Biological and/or mechanical mixing may accelerate the decomposition rates although drying out of the top most layers may slow it down. Increased exchangeable base cation concentrations in deeper mineral soil layers are usually caused by an increased CEC, offering more exchange sites to which base cations can absorb. However, decreased exchangeable base cation concentrations have frequently been measured in soils with extensive nitrification and cation exchange with H^+. Clear-cutting increases the total base cation concentrations for the same reasons as given above, while decreases result from loss of soil organic matter.

The treatment effects on the pools of exchangeable and total base cations are usually non-significant although there have been some reports of either increasing or decreasing amounts. In the forest floor and upper mineral soil, diminishing pools are generally explained by decomposition and leaching. Deeper in the mineral soil, the pools seem to increase as a result of increased retention and weathering.

In general, the differences between whole-tree and stem-only harvest (WTH↔CH) are non-significant in terms of exchangeable base cation concentrations. However, lower exchangeable (and sometimes total) base cation concentrations have frequently been reported following WTH compared with CH. The mechanism responsible for the observed differences is a relocation of base cations from decomposing logging residues to the soil exchange complex in stem-only harvested plots, which creates a measurable difference with whole-tree harvested plots. Similar differences have been observed regarding the soil BC pools.

The soil chemical studies generally agree with the observations made in soil water, groundwater and stream water. In the aqueous media, the base cation concentrations and fluxes seem to increase after clear-felling and regardless of harvest intensity. Decomposition, mineralization, litter leaching, weathering and reduced plant uptake may cause increased soil water base cation concentrations. In addition, hydrogen ions released during nitrification participate in cation exchange processes transferring BC from the soil matrix to soil water. Processes responsible for decreased BC concentrations include dilution, leaching, cation exchange, mineral fixation of K^+, reduced deposition of mineral acid anions and on rare occasions, decreased nitrification after clear-cutting of nitrogen fixating stands e.g. alder.

Treatment differences between whole-tree and conventional harvest on the BC leaching to soil water, groundwater and stream water are scarce. If observed, the leaching of base cations generally decrease following WTH compared with CH. The presence or absence of logging residues, stimulating mineralization and nitrification, most often explains the treatment differences between WTH and CH.

The reported changes in base cation concentrations, pools and fluxes show that there is no single mechanism which can explain the observed treatment differences but rather a combination of factors which includes:

- Plant uptake (reduced or increased)
- Litter leaching
- Decomposition of logging residues
- Mineralization and nitrification
- Denitrification
- Mobile anion associated leaching (in particular NO_3^-)
- Changed atmospheric deposition (primarily SO_4^{2-}, Cl^-, NO_3^-)
- Dilution due to increased water fluxes (reduced evapotranspiration).

It appears that nitrification is a key process, which can increase base cation concentrations and fluxes in soil water, groundwater and stream water. Thus, by identifying nitrifying forest soils measures can be taken to limit the nutrient leaching. Removal of logging residues may also decrease NO_3^- and base cation leaching but usually the effect of cutting per se is greater than short-term treatment differences between WTH and CH.

8.6 Anions (NO_3^-, SO_4^{2-}, Cl^- and DOC)

8.6.1 NO_3^-

Nitrogen losses from boreal forests are generally small (Hedin et al., 1995; Löfgren & Olsson, 1990; Binkley & Högberg, 1997; Nettelbladt et al., 2006) since tree growth is limited by the nitrogen availability (Tamm, 1991). However, increases do occur as a result of fire (Bladon et al., 2008), wind-throw (Mellert et al., 1996; Hellsten et al., 2009), insect outbreak (Swank et al., 1981) and human activities (Vitousek & Melillo, 1979; Binkley & Brown, 1993; Gundersen et al., 2006). In areas receiving high levels of N deposition, careful management has been called for not to enhance NO_3^- leaching due to N saturation (Rothe & Mellert, 2004). The term “N saturation” is not well defined and is used inconsistently in the literature (Binkley & Högberg, 1997). In general, most forested sites in Sweden are not N saturated using the definition “output of N is equal to or higher than the input” except for local sites on the southwest coast (Binkley & Högberg, 1997; Nettelbladt et al., 2006). Branches and needles contain higher concentration of nutrients (especially of N) compared with stems (e.g. Rosén, 1991). Therefore, in areas where the N deposition is high, whole-tree harvesting has been suggested as a countermeasure to lower the N accumulation and thereby the risk for N saturation. Harvest of logging residues in three thinnings and clear-felled areas would remove about 500-700 kg N per hectare, assuming 15% logging residues left on ground, and theoretically reduce the risk of N leaching during the clear-cut phase (Lundborg, 1997).

In Sweden, Wiklander (1983) found a positive correlation between groundwater NO_3^- concentrations and site quality. Additionally, soil water studies in clear-cuts along an N deposition gradient showed a positive correlation between NO_3^- concentrations and deposition (Akselsson et al., 2004). An increased mineralization due to increased N-input was proposed to cause this effect (op.cit.). Gundersen et al. (2006) found the same positive relationship between atmospheric N-input and NO_3^- leaching for sites in the US and Europe (Figure 11). In contrast to what is generally believed (e.g. Lundborg, 1997), they did not see any differences between stem-only and whole-tree harvesting. Instead, excess NO_3^- leaching following clear-cutting is a result of pre-existing site conditions. In addition, tree species, stand age and deposition may influence excess NO_3^- leaching (Binkley & Högberg, 1997).

The results from this review also show that the concentration and fluxes of NO_3^- in soil water (Hornbeck & Kropelin, 1982; Stevens & Hornung, 1988; Hendrickson et al., 1989; Mitchell et al., 1989; Dahlgren & Driscoll, 1994; Romanowicz et al., 1996; Titus et al., 1997; Piirainen et al., 2002; Rothe & Mellert, 2004; Briggs et al., 2004; Westling et al., 2004; Ranger et al., 2007), groundwater (Wiklander, 1974; Tamm & Wiklander, 1972; Tamm et al., 1974; Kubin, 1998; Parfitt et al., 2002; Neal et al., 2004; Rusanen et al., 2004; Mannerkoski et al., 2005) and stream water (Bormann et al., 1968; Likens et al., 1970; Martin & Pierce, 1980; Grip, 1982; Adamson et al., 1987; Mann et al., 1988; Tiedemann et al., 1988; Harr & Fredriksen, 1988; Adamson & Hornung, 1990; Ahtiainen, 1992; Jewett et al., 1995; Kubin, 1995; Rosén et al., 1996; Ahtiainen & Huttunen, 1999; Martin et al., 2000; Swank et al, 2001; Neal et al., 2004; Tremblay et al., 2009; Löfgren et al., 2009) usually increase after harvest regardless of cutting method (CH and WTH) compared with pre-harvest conditions and uncut controls.

Increased NO_3^- concentrations after clear-cutting are probably the most distinct chemical response and also the "*simplest emerging pattern*" according to Neal et al. (1998). Ring (2007) draw some generalized conclusions about the short-term effects of harvesting (stem-only and whole-tree harvesting) based on data from eight sites situated in Sweden. The conclusions are schematically illustrated in Figure 25 and follow those of Roth & Mellert (2004) and Gundersen et al. (2006):

- The concentration of NH_4^+ and NO_3^- generally increases following harvesting.
- The highest concentration is found at high quality sites.
- Elevated NO_3^- concentrations last approximately 5 years.
- The mean soil solution NO_3^- concentration is linearly correlated to the C/N-ratio and site quality class.
- There is a lag time before elevated concentrations are observed.
- The duration of this lag time differs between rich and poor sites. At poor sites (3.1 m^3 per hectare and year) the lag time may be as long as 5 years compared with rich sites (8.4-12.6 m^3 per hectare and year) where the lag time is only one year.

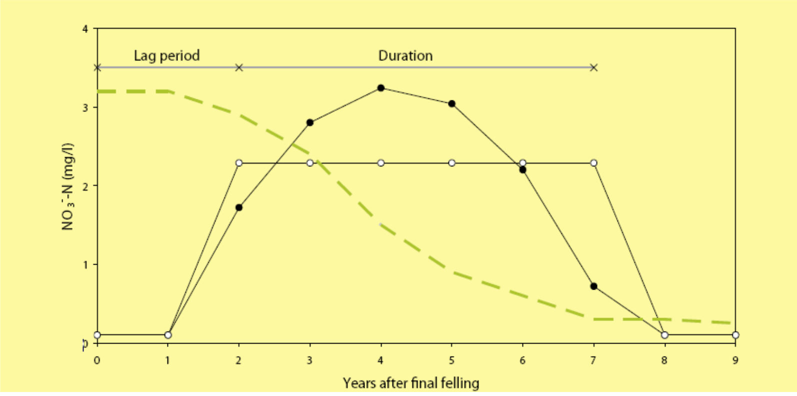


Figure 25. A schematic illustration of the short-term response and duration in NO_3^- concentration (solid line) and runoff (broken line) after cutting. The increase can either be expressed as mean concentration (white circles) or as a polynomial equation (black circles). From Ring (2007).

The NO_3^- concentration increase is most often attributed to increased mineralization of organic N and nitrification in the forest floor and mineral soil. It often coincides with increased losses of K, decreased pH (e.g. Romanowicz et al., 1996; Staaf & Olsson, 1994; Dahlgren & Driscoll, 1994: Rosén et al., 1996; Neal et al., 2004; Adamson & Hornung, 1990) and SO_4^{2-} concentrations (Hornbeck & Kropelin, 1982), while the concentrations of inorganic aluminium increase (Emmett et al., 1991). The sudden release of nutrients are believed to be the result of higher soil temperatures positively affecting decomposition, increased water fluxes, mixing of the forest floor with the mineral soil, reduced vegetation uptake and a decrease in microbial immobilization. In addition, herbicide-treatment retarding growth has also been shown to increase NO_3^- concentration in soil water (Briggs et al., 2000) and surface water (Likens et al., 1970). Usually, the increase is most pronounced in the topsoil (Jewett et al., 1995). Differences in silvicultural practices such as the use of buffer strips (Martin et al., 1984; Ahtiainen 1992) partially clear-cut (Martin & Pierce, 1980; Feller et al., 2005) or strip-cut (Martin et al., 2000) harvest can reduce enhanced NO_3^- leaching to stream water.

Most studies report a short-term increase in NO_3^--, usually less than five years (e. g. Staaf & Olsson, 1994; Hornbeck & Kropelin, 1982; Mitchell et al., 1989; Rusanen et al., 2004) linked to the increasing nutrient demand of the new forest stand. As a result, NO_3^-- leaching is small in mature forests (Katzensteiner 2003). However, there are a few reports of long-term increases. For example, Kubin (1998) observed elevated NO_3^- concentrations in groundwater beginning in the first year following stem-only and whole-tree harvest of two upland forested sites in Finland, relative to an uncut control. Nitrate concentrations continued to increase and reached a maximum 4-5 years after harvest. However, the concentrations remained elevated throughout the study period, which lasted up to ten years after harvest. In a follow-up study, NO_3^- concentrations were still elevated, 20 years after harvest (Kubin & Krecek, 2009). This was partly explained by slow regeneration of the new forest stand. Martin et al. (2000) measured long-term changes in NO_3^-- concentrations following stem-only (0-14 years) and whole-tree (0-27 years) harvest of a hardwood forest. In the stem-only strip cut area, the NO_3^- concentrations increased up to 7 years after harvest. Thereafter (8-14 years), the NO_3^- concentrations were less than during the pre-harvest conditions due to the regenerating tree stands. In the whole-tree harvested stand, NO_3^-- concentrations followed the same pattern. However, the concentrations of NO_3^- increased again after 20 years most likely due to high mortality of short-lived pioneer trees and/or inefficient root activity of the new tree stand. Swank et al. (2001) also found a second NO_3^--pulse due to increased tree mortality circa 10 years after stem-only harvest of a mixed deciduous forest.

Some sites showed a short-term decrease in NO_3^- concentration (and often K^+) in soil water following cutting (CH or WTH) (e.g. Mann et al., 1988; Robertson et al., 2000). Decreasing soil water NO_3^- concentrations in the organic layer has also been observed by Stevens & Hornung (1988) after stem-only harvest compared with an uncut control. By removing the trees, transpiration rates and interception decreased which in turn increased soil wetness causing a “dilution effect”. Denitrification via increased groundwater levels (Neal et al., 1992) or via stream water running through wetland areas (Spoelstra et al., 2010) could also lower excess NO_3^- concentrations.

Ranger et al., (2007) observed large decreases in soil water NO_3^- concentrations ranging between -200 to -530 µeq per litre (-2.8 to -7.4 mg per litre) . The decrease occurred throughout the soil profile (organic layer down to 120 cm) following whole-tree harvesting. This is in contrast to Stevens & Hornung (1998) who observed a decrease in the organic layer, no change in the E- and Bs-horizon and an increase in the C-horizon. The mechanism proposed to explain these results was that the Douglas fir stimulated the nitrifier population and by removing the trees their activity would decrease. Soil water concentrations in the mineral soil also decreased following clear-cutting in a pine plantation of high natural N status in New Zeeland (Parfitt et al., 2002). Suggested mechanisms include increased uptake by weeds, an increase in soil microbial biomass and a slow turnover of N in the microbial pools. Hence, final felling, with or without logging residues, can limit excess NO_3^- leaching under special circumstances.

Treatment differences between whole-tree and stem-only harvest are usually non-significant although there are some reports of lower NO_3^- concentrations following whole-tree harvest. For example, Staaf and Olsson (1994) reported increased NO_3^- concentrations in soil water (30 cm) following both stem-only and whole-tree harvest of a spruce stand in southern Sweden. However, higher leaching of NO_3^- (and K^+) was found under piles of slash (CH) compared with sites where logging residues had been collected (WTH). In addition, excess leaching of NO_3^- was effectively reduced by *Deschampsia flexuosa* growing on WTH plots. Lower soil water NO_3^- concentrations following WTH was also found by Rosén & Lundmark-Thelin (1987) at a coniferous site in Sweden. These results are in agreement with Emmett et al. (1991) who observed higher NO_3^--fluxes under piles of slash (CH) due to stimulation of the microbial activity, transforming organic N to NO_3^-. According to Emmett et al. (1991) logging of residues can reduce the leaching of NO_3^- by 10%. In Canada, Titus et al. (1997) measured lower soil water concentrations on WTH plots compared with CH. The higher concentrations found on CH plots were attributed to leaching from the slash itself and increased microbial activity and mineralization beneath the slash. After three years no differences were found between the treatments. However, Titus et al. (1987) pointed out that lower soil solution concentrations on WTH-plots should not be used exclusively as an indicator for less impact since this does not take into consideration increased runoff and potentially larger NO_3^--fluxes.

In summary, clear-cutting regardless of intensity generally leads to a short-term increase in the NO_3^- concentration and fluxes in soil water, groundwater and stream water. This coincides with a base cation concentration increase (especially K^+) and a decrease in pH due to H^+ production by the nitrification process. The pH change is buffered by aluminium oxyhydroxides in the soils, releasing inorganic Al^{n+}. The magnitude of these changes and the duration of the increased leaching vary from site to site. The mechanisms most important for the observed changes are reduced vegetation uptake and increased mineralization and nitrification. The use of buffer zones and harvesting techniques such as partial or patch cut have the potential to reduce the NO_3^--leaching. The NO_3^- concentration increases are shorter following WTH compared with CH, which can leach for more than 10 years (Kubin, 1998). Tree mortality can induce a NO_3^- pulse.

Processes responsible for increasing or decreasing NO_3^- concentrations and fluxes include:

- Mineralization and nitrification
- Abiotic site condition changes (temperature and moisture)
- Immobilization
- Denitrification
- Silviculture practices (harvest techniques, buffer zones, fertilization etc.).
- Lower rates of N fixation (e.g. alder tree harvesting)
- Dilution effects
- Increased groundwater tables and runoff
- Tree mortality
- Revegetation

8.6.2 SO_4^{2-}

There are a number of studies where lower concentrations of SO_4^{2-} has been observed in soil water (Hornbeck & Kropelin, 1982; Westling et al., 2004; Piirainen et al., 2004; Robertson et al., 2000; Welsch et al., 2004; McLaughlin & Phillips, 2006; Zetterberg et al., 2008), groundwater (Neal et al., 2004; Welsch et al., 2004) and stream water (Bormann et al., 1968; Likens et al., 1970; Hornbeck & Kropelin, 1982; Adamson et al., 1987; Lawrence et al., 1987; Mitchell et al., 1989; Mann et al., 1989; Adamson & Hornung, 1990; Dahlgren & Driscoll, 1994; Martin et al., 2000; Welsch et al., 2004; Neal et al., 2004; McLaughlin & Phillips, 2006; Löfgren et al., 2009) following stem-only and whole-tree harvest.

Hornbeck & Kropelin (1982) observed in the beginning of the 1980’s lower soil water and surface water SO_4^- concentrations following whole-tree harvest compared with an uncut control. They proposed dilution as primary cause for this change since runoff increased, but speculated that S oxidizing inhibition and decrease dry S deposition could cause lower concentrations. Removal of the forest cover during times of high atmospheric S deposition would lead to decreased dry deposition and consequently a decrease in stream water SO_4^- concentrations (Hultberg, 1985).

In southwest Sweden, decreased concentrations of SO_4^- were observed in soil water (50 cm depth) during the first four years following stem-only harvest as a result of reduced S dry deposition (Westling et al, 2004). Measurements were carried out during the 1990's, a period when non-marine S deposition in throughfall was greater than 14 kg per hectare in this region (Nettelbladt et al., 2006). In the UK, Robertson et al. (2000) measured lower soil water concentrations after stem-only harvest of four different tree species as a result of reduced dry deposition and/or dilution. Adamson et al. (1987) studied short-term (0-2 years) stream water changes following stem-only harvest of a spruce stand in 1983. Compared with a control catchment, SO_4^{2-} concentrations decreased as a result of decreased dry deposition, and possibly, decreased plant uptake. Similar results were found by Adamson & Hornung (1990) who carried out studies in the same area but including results from more sample plots. The results showed that SO_4^{2-} concentrations continued to decrease up to four years after harvest (Adamson & Hornung, 1990).

As a result of the S emission reductions, decreased concentrations of SO_4^{2-} in surface waters have been observed across Europe and North America (Skjelkvåle et al., 2001ab; Stoddard et al., 1999). The relative importance of dry deposition for decreasing SO_4^{2-} concentrations during the clear-cut phase must therefore be questioned and other mechanisms be looked for. For instance, Löfgren et al. (2009) conducted a study in the northern part of Sweden where current and historically S deposition has been low. The forest was clear-cut in 2006 using conventional harvest methods leaving logging residues left on ground. During the two years post-cutting period, the concentration of SO_4^{2-} decreased in stream water relative to an uncut area. A decrease in dry deposition was ruled out since it only amounted to circa 1.3 kg per hectare and year. Instead, higher groundwater levels could have led to dilution and anaerobic conditions and subsequently, less oxidation of SO_4^{2-}. Thus, in areas where current rates of S deposition are low, a reduction in dry deposition is not likely to be responsible for decreased SO_4^{2-} concentrations. Present day deposition rates of total S vary between 0-2 kg per hectare in the northern parts of Sweden and 3-6 kg per hectare in the southern part (Figure 2). However, some areas on the southwest coast still receive more than 6 kg per hectare and year. Thus, in these areas a reduction in dry deposition via clear-cutting could lead to decreased soil water, ground water and stream water SO_4^{2-} concentrations.

In areas where S deposition historically has been high or continues to be high, an accumulation of SO_4^{2-} has taken place in the soils, which may delay the recovery process in streams and lakes, despite reduced S emissions. Input-output budgets for acidified forest catchments in southwest Sweden show larger output than input of SO_4^{2-} as a result of decreased SO_4^{2-} deposition (Moldan, 1999). Suggested mechanism for these observations has been desorption of SO_4^{2-} in the mineral soil. However, soil water studies below the O-horizon (Löfgren et al., 2001) and isotopic studies on extractable SO_4^{2-} (Mörth et al., 2005) have also shown a net release of organic S via mineralization in the forest floor. In these systems, clear-cutting may delay the natural recovery process even further by affecting decomposition rates and SO_4^{2-} retention.

Increased S retention in the soil as a result of acidification has been suggested for decreased soil water and stream water concentrations (Fuller et al., 1987; Nodvin et al., 1988; Mitchell et al., 1989; Dahlgren & Driscoll, 1994; Martin et al., 2000; Welsch et al., 2004). For soil colloids with variable charge, the concentration of SO_4^{2-} in the soil and soil solution depends on soil solution pH, ionic strength (Sokolova & Alekseeva, 2007) and anion adsorption processes

(Johnson & Cole, 1980). If the pH in the soil solution increases, the number of negative charges on the soil surfaces will increase. Alternatively, if the pH decreases, the positive charge on soil surfaces increases and SO_4^{2-} ions can absorb. Nodvin et al. (1986) studied the effects of pH and ionic strength on SO_4^{2-} adsorption by using soil samples from a podzolic B-horizon. Depending on SO_4^{2-} concentration and pH in the solution, the net removal or release of SO_4^{2-} varied according to Figure 26. A small change in pH can thus significantly increase the SO_4^{2-} retention at low pH-values.

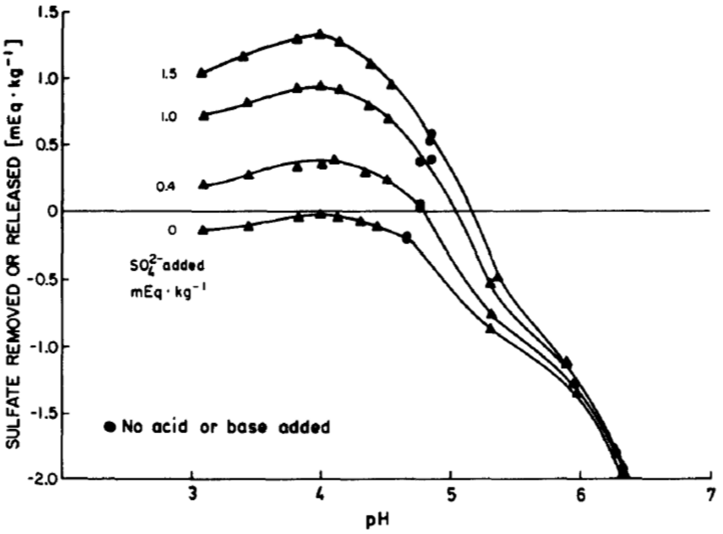


Figure 26. The net removal (positive values) or release (negative values) of SO_4^{2-} as a function of soil solution concentration and pH. From Nodvin et al., 1986.

In the Hubbard Brook Experimental Forest, stem-only harvest and herbicide treatment caused stream water SO_4^{2-} concentrations to decrease relative to a control watershed (Likens et al., 1970). Nodvin et al. (1988) showed that the decline in SO_4^{2-} concentrations was caused by increased soil retention as a result of acidification induced by nitrification. These results are in agreement with Mitchell et al. (1989) and Zhang et al., (1999) who studied sulfur dynamics two and eight years, respectively, after whole-tree harvest in a nearby catchment. By analysing soil samples it was found that the amount of adsorbed SO_4^{2-} increased significantly after cutting in the lower mineral soil (E- and Bh-horizon) (Mitchell et al., 1989). In contrast, no change was observed in the organic S-pool. Higher SO_4^{2-} retention led to lower soil water (Bs2) and stream water SO_4^{2-} concentrations, relative to an uncut control, five months after harvest (Fuller et al., 1987). In contrast, forest floor (Oa) concentrations remained constant or increased as a result of increased mineralization (Fuller et al., 1987). Flux calculations showed that 88% of the decline in stream water concentrations was accounted for by increased S adsorption in the E- and Bh-horizons (Mitchell et al., 1989). A few years later, the stream water SO_4^{2-} concentrations increased relative to the control, probably as a result of a pH-induced desorption (Dahlgren & Driscoll, 1994). Fifteen years later, stream water SO_4^{2-} concentrations were still elevated relative to the control catchment (Martin et al., 2000). A similar short-term (4 years) decrease in stream water SO_4^{2-} concentration was also observed in a strip-cut forest in Hubbard Brook before returning to pre-harvest conditions (Martin et al., 2000).

The response of SO_4^{2-} concentrations to stem-only cutting of a northern hardwood forest, was studied in the Catskill Mountains, New York by combining the results from bulk deposition, soil water, groundwater, stream water concentrations and stream water fluxes and buried soil bags (Welsch et al., 2004). Following clear-cutting, short-term decreases in SO_4^{2-} concentrations were evident in all matrixes, especially in the B-horizon soil solution where pH also dropped by 0.5 units (from 4.5 to 4.0) compared with pre-harvest conditions. The decrease in soil water SO_4^{2-} concentrations was positively correlated with H^+ concentration (r^2=0.71) in the B-horizon but not in the O-horizon. By removing the tree canopy, dry deposition of S decreased but only accounted for a decrease in stream water fluxes of between 0 to 43% during the first two years. The strong relationship between pH and SO_4^{2-} concentration indicated increased adsorption of SO_4^{2-} to the mineral surfaces as the primary cause of the decreasing SO_4^{2-}-trends in stream water concentrations. In addition, no changes were seen in the O-horizon soil water or in buried soil bags (25-30 cm), strengthening the hypothesis of S retention in deeper soil.

In Finland, Piirainen et al. (2004) observed a general decrease in SO_4^{2-} leaching from below the O-horizon over a three-year period following stem-only harvesting as a result of decreased atmospheric S inputs. However, variations between years and horizons were observed. The greatest leaching from below the O-horizon occurred during the first year following harvest but decreased during the second and third year. These results were explained by the release of SO_4^{2-} from logging residues via mineralization, direct leaching of S from dead organic matter, increased uptake of S by the understory vegetation, microbial immobilization and possibly reduced mineralization of S. Below the E- and B-horizon, the retention of SO_4^{2-} decreased after harvest probably as a result of decreased plant uptake in the E-horizon and a net release of earlier adsorbed SO_4^{2-} from the B-horizon itself due to an increase in soil water pH. Despite these changes, the net flux of SO_4^{2-} from the B-horizon did not change following harvesting.

Decreasing SO_4^{2-} concentrations appear to be of short duration before returning to pre-harvest conditions or values similar to an uncut control forest. However, in a study by McLaughlin & Phillips (2006) lower SO_4^{2-} concentrations were still observed in soil water (4 times lower) and stream water, 17 years after whole-tree harvest, compared with a mature forest. It was hypothesized that the “internal acidity generation” was greater in the control catchment resulting in lower pH and higher SO_4^{2-} concentrations. However, as shown above, a lower pH would result in lower SO_4^{2-} concentrations and not vice versa. An alternative explanation would be that the internal acidity generation was equal in the two watersheds but that buffering mechanisms in the cut area more effectively neutralized the acidity.

In contrast to these results, Mannerkoski et al. (2004) and Rusanen et al. (2004) did not observe any significant short-term effects on groundwater concentrations following stem-only harvest compared with an uncut control forest and pre-harvest conditions, respectively. Also, Parfitt et al., (2002) observed no changes in soil water concentrations following stem-only harvest. These results are in agreement with stream water studies by Grip (1982) and Patric & Aubertin (1977). Reasons for the lack of treatment response are usually not discussed. Ranger et al. (2007) reported increased soil water SO_4^{2-} concentrations at 60 cm depth after whole-tree harvest of a Douglas fir stand in France. Desorption of SO_4^{2-} from mineral soil surfaces were not considered likely. Instead it was hypothesized that mineralization of organic S could be a reason.

In terms of soil water fluxes, SO_4^{2-} leaching usually increases after harvest (Ranger et al., 2007) and stream water (Haveraaen, 1981; Adamson & Hornung, 1990; Rosén et al., 1996; Hornbeck et al., 1990; Swank et al., 2001; Löfgren et al., 2009). Increased leaching is usually coupled with increased groundwater tables and stream runoff.

Studies describing treatment differences between whole-tree and stem-only harvesting are rare. According to soil water studies from below 50 cm in the mineral soil, Westling et al. (2004) observed lower concentrations 9-11 years after whole-tree harvest. No explanation was given.

Soil pools were not affected by harvest differences (Örlander et al., 1997). In another study by Zetterberg et al. (2008) lower concentrations were still measurable after 28-31 years.

In summary, the concentration of SO_4^{2-} usually decreases after cutting, regardless of harvest method (CH and WTH). The effect is usually short-term since much of the increase can be explained by the “clear-cut” effect e.g. increased nitrification and subsequent release of H^+ leading to increased retention of SO_4^{2-} ions on mineral surfaces. Processes responsible for the observed treatment effects include:

- Reduced dry deposition
- Dilution via increased groundwater tables and stream runoff
- Inhibition of S-oxidizing bacteria
- Anaerobic condition leading to S reduction
- Increased soil retention
- Changes in plant uptake

8.6.3 Cl^-

Chloride is, together with SO_4^{2-} and NO_3^-, a major component of the strong acid anion sum (equation 7). However, chlorine (including all organic forms of chlorine and inorganic Cl^-) is seldom studied in forested systems (Kauffman et al., 2003, Lovett et al., 2005) probably since the acidifying effect of Cl^- after harvest is less pronounced compared with the other two major anions (Neal et al., 1998). It originates primarily from atmospheric deposition (dry or wet) of sea salt why the highest soil and water concentrations are found in coastal regions (Öberg, 1998). During heavy storm events, sea salt deposition can cause episodic acidification in streams and lakes (e.g. Wigington et al., 1996a; Wright et al., 1988) and in the most severe cases lead to fish death (Hindar et al., 1994). An increase in neutral salt concentration is a powerful variable for displacing cations, aluminium and H^+ on the soil exchange complex. This “sea-salt” effect is well recognized and has been documented over the years in Europe (e.g. Wiklander, 1975; Harriman et al., 1995; Hindar et al., 2005) and America (Wigington et al., 1996b). Input via weathering is considered small. For example, Lovett et al. (2005) estimated that only 1-2% of stream water output originated from weathering in Hubbard Brook, USA.

In general, the concentration of Cl^- decrease following harvest in soil water (Dahlgren & Driscoll, 1994; McLaughlin & Phillips, 2006; Westling et al., 2004; Parfitt et al., 2002; Robertson et al., 2000), groundwater (Neal et al., 2004) and stream water (Adamson et al., 1987; Neal et al., 2004) compared with pre-harvest conditions and uncut controls. In most studies this decrease is attributed to a reduction in dry deposition following the removal of “atmospheric scavenging” trees (Westling et al., 2004; Robertson et al., 2000; Neal et al., 2004; Adamson et al., 1987; Neal et al., 2004).

For example, Robertson et al. (2000) studied the effect of stem-only harvest in throughfall and soil water chemistry under four different tree species (spruce, pine, oak and alder) in the UK. Compared with pre-cutting values the concentration of Cl^- in the L-, A-, and C-horizons decreased during a three years study period immediately following harvest (especially below conifer trees) as a result of decreased atmospheric deposition and/or dilution. The most dramatic decrease (-715 µeq per litre) occurred in the A-horizon below pine trees. Robertson et al. (2000) suggested that higher surface leaf area of conifers and the fact that conifer trees don’t shed their needles during winter time is likely to have caused the species differences. Neal et al. (2002) found similar results in a paired catchment study. Following stem-only harvest the concentration of Cl^- in groundwater and stream water decreased during a six years study period, compared with an uncut control forest and pre-harvest conditions. Adamson et al., (1987) reported decreased stream water Cl^- concentrations two years after stem-only harvest, relative to a control catchment. The number of long-term observations is few. However, McLaughlin & Phillips (2006), measured lower soil water Cl^- concentrations (-14.4 µeq per litre) in the mineral soil (total 25-50 cm) 16-18 years after whole-tree harvest compared with a control forest. However, no treatment differences could be seen in the stream water concentrations. No explanation was given, but it is likely that the crown structure of the regenerating stand has not fully developed which would lead to lower dry deposition compared with a mature forest. Also, the lack of response in stream water could be the result of internal sources releasing Cl^-, see below.

Changes in stream water chemistry may not always mirror those in soil water and groundwater. For example, Parfitt et al. (2002) measured decreased soil water concentrations in the mineral soil (10 and 60 cm) up to three years after stem-only cutting of a *Pinus radiata* stand in New Zeeland, similar to the results above. However, spring water and stream water concentrations did not show any response to cutting (or decreased slightly), which suggested internal sources of Cl^- (rhyolite and pumice) released deeper down in the soil via weathering processes. Internal sources of Cl^- were also believed to be the reason for increased stream water concentrations six months after stem-only cutting in the Hubbard Brook Experimental Forest since output were larger than atmospheric input (Likens et al., 1970). However, some of the increase could also be attributed to the addition of a Br^- containing herbicide, not separated from Cl^- during analysis.

These results are in agreement with those of Dahlgren & Driscoll (1994) who studied the short-term effect (5-6 years) of whole-tree harvest on soil water and stream water chemistry at three study sites. In general, soil water concentrations were small and relatively constant throughout the soil profile. Also, highest concentrations were found below coniferous stands compared with hardwood stands, similar to the results of Robertson et al. (2000). Clear-cutting resulted in a small short-term decrease in soil water concentrations in the Oa-horizon at two of the three sites. However, no changes were seen further down in the mineral soil (Bhs- and Bs2-horizon) whereas stream water concentrations increased slightly before returning to values similar to the control. Although not discussed this would indicate an internal source of Cl^-. Öberg (1998) showed in a review that organic matter contains chlorine in amounts similar to phosphorous, which may be released via mineralization. In addition, the concentration of organic chlorine is 2-4 times larger than the concentration of inorganic Cl^- in the O-horizon (Johansson et al., 2003). In a later study by Öberg et al. (2005) it was shown that the pool of chlorine in organic matter is much larger than what is being lost from a catchment. Similarly, Rodstedth et al. (2003) found that the pools of chlorine were four times larger than the pools of Cl^-. A small change in this pool may have an effect on Cl^- budgets.

Higher stream water concentrations after stem-only cutting was also observed by Löfgren et al. (2009b), who discussed the possibility of decomposition of organic matter, roots and litter (cf. Kauffman et al., 2003; Svensson, 2006). In a Finnish study, Mannerkoski et al. (2005) also observed increased groundwater concentrations of Cl^-. They found a significant increase in Cl^- concentration in groundwater wells five years after stem-only harvest compared with control plots and pre-harvest values. The increase appeared in every groundwater well regardless of soil type. According to Mannerkoski et al. (2005) the reason for this increase is unclear, but release via decomposition was not considered in their study.

In contrast to these results, Hornbeck & Kropelin (1982) did not see any significant changes in soil water (30 cm) or stream water Cl^- concentrations, three years after whole-tree harvest compared with a reference area. Similarly, Rusanen et al. (2004) did not observe any significant treatment effects on groundwater Cl^- concentrations following stem-only harvest in a large aquifer. In Sweden, Grip (1982) did not see any changes in stream water concentrations of Cl^- three years after stem-only cutting compared with a growing control forest. Reasons for the lack of effects are seldom discussed.

Soil water and stream water fluxes of Cl^- generally increase after harvest in the range of 7.6-8.3 kg per hectare in soil water (Ranger et al., 2007) and 0.1-19 kg per hectare in stream water (Likens et al., 1970; Haveraaen, 1981; Hornbeck et al., 1990; Swank et al., 2001; Rosén et al., 1996; Löfgren et al., 2009) for periods lasting up to eight years (Rosén et al., 1996). As groundwater tables rise following cutting, runoff increases which leads to a greater export of Cl^- from the soil to the stream.

Treatment differences between whole-tree and stem-only harvest for groundwater and stream water are not described in the literature. However, results from a soil water study by Westling et al. (2004) suggest that the Cl^- concentration following whole-tree harvest is lower compared with stem-only at a soil depth of 50 cm. However, at another research site included in the same study, no differences could be seen. Additionally, the soil water fluxes did not show any treatment effects (Örlander et al., 2004). Thus, treatment differences appear to be small and insignificant between stem-only and whole-tree harvest.

Chloride is generally believed to act as a conservative ion in the soil with an output largely balanced by the input via deposition (Lovett et al., 2005). However, sandbox experiments in Hubbard Brook, USA, have shown evidence of a built up of inorganic Cl^- in the soil (Kauffman et al., 2003). During harvest this inorganic Cl^- could be released from roots, litter and organic matter and be responsible for as much as 50% of the leached Cl^- (op. cit.). In addition, reduced vegetation uptake and decomposition of organic chlorine to Cl^- could occur. In Sweden, Svensson (2006) showed that the biogeochemistry of Cl^- is much more complex than previously believed involving organically bound chlorine released to the aqueous solution via decomposition. As a result, stream water Cl^- could be as old as several hundreds of years. Svensson (2006) proposed that the upper soil layers would act as a sink where inorganic Cl^- is transformed into organic chlorine. The lower soil layers would thus act as a source to which organic chlorine is transported to and released via decomposition.

In summary, clear-cutting generally leads to decreased Cl^- concentrations and increased fluxes in soil water, groundwater and stream water regardless of harvest methods. However, increased concentrations have frequently been reported in groundwater and stream water. Mechanisms responsible for the observed changes include:

- Decreased dry deposition
- Reduced plant uptake
- Increased decomposition
- Weathering
- Dilution

8.6.4 DOC

The concentration and export of DOC largely depend on catchment characteristics and seasonality (Ågren et al., 2007). For example, higher concentrations of DOC can usually be found in soil- and stream water draining wetlands and peat land areas compared with forest soils. The DOC flux from these types of catchments can be 4-8% of the annual net primary production (Kolka et al., 2008). Besides being an important source of nutrient and energy (Jansson et al., 2000; Ågren et al., 2008), dissolved organic acids are also strong complexing agents for metals such as aluminium (Cory et al., 2006) and mercury (Garcia et al., 2007) thus reducing their bioavailability. DOC is also a natural source of acidity (Driscoll et al., 1989). For instance, in an American study by Dijkstra et al. (2001) organic acids accounted for 25% and 43% of the negative charge depending on tree species. This negative charge is balanced by positive ions (protons, base cations and metals) and contributes to increased base cation leaching losses from the soil (Dijkstra et al., 2001). In the northern part of Sweden, DOC is the driving force of episodic stream water acidification during spring flow (Laudon et al., 2000) and autumn storms (Laudon & Bishop, 2002).

Recently, a number of studies have reported increasing concentrations of DOC in surface waters (lakes and streams) across Europe and North America (e.g. Skjelkvåle et al., 2001ab, 2005). Different processes have been proposed to explain these trends such as increased runoff (Hongve et al., 2004), increased temperature (Freeman et al., 2001), climate change (Worrall et al., 2003), N deposition (Pregitzer et al., 2004) leading to a shift in microbial community and function (Zak et al., 2008), land use changes (Garnett et al., 2000), increasing CO_2-levels (Freeman et al., 2004), increased litter production (Roulet & Moore, 2006), decreasing acid deposition (Evans et al., 2006) and changes in atmospheric deposition chemistry and acid sensitivity of catchments (Monteith et al., 2007). In Sweden, Weyhenmeyer & Karlsson (2009) showed that DOC increased in a nonlinear manner along a temperature gradient (longitudinal and latitudinal), best explained by the number of days when air temperature reached above 0°C. These results suggest that climatic factors are responsible for the observed increases. On the other hand, looking more closely at the climatic variables, Zhang et al. (2010) did not find any evidence of either increasing or decreasing trends in DOC concentrations in lakes in Canada during the last 21 years, except for one study area (Experimental Lakes Area in central Canada) where increasing concentrations correlated with increasing summer precipitation and decreasing summer total solar radiation. In a recent review, Clark et al (2010) have discussed possible mechanisms for the DOC dynamics.

In contrast to the surface waters, Zetterberg & Löfgren (2008) reported decreasing soil water DOC concentrations at 50 cm depth in southern part of Sweden during the period 1986-2008, which suggest that different processes are responsible for the increasing trends in surface water and the decreasing trend in soil water. Similarly, Wu et al. (2009) observed decreasing soil water DOC concentrations in Norway.

According to Hope et al. (1994) the main sources of DOC is input from vegetation (leaching from live vegetation and litter) and from the soil (microbial metabolism, root exudation and leaching of soil organic matter). Clear-cutting will have an effect on these cycles, but little is known about the effects of logging on the leaching of DOC in the boreal forest (Piirainen et al., 2002).

This review shows that clear-cutting often increases the concentrations of DOC in soil water, ground water and stream water following stem-only harvest. For example, Robertson et al. (2000) reported increased soil water DOC concentrations in the forest floor (L-layer) three years after stem-only cutting of a mixed hardwood/softwood forest in the UK. Smaller, but still significant increases were seen in the mineral soil (A- and C-horizon). These increases were attributed to increased decomposition and possibly leaching from logging debris. Similarly, Dahlgren & Driscoll (1994) observed higher DOC concentrations in the Oa-, Bhs- and Bs2-horizon at high and low elevation, after whole-tree harvest compared with an uncut forest. In comparison, at middle elevation no change was observed in the Oa- and Bhs-horizon whereas the concentration decreased in the Bs2-horizon. Stream water measurements showed that whole-tree harvest had little effect on the DOC concentrations. Thus, DOC was leached from the forest floor to the lower mineral soil where it was largely retained (op. cit.). However, a couple of years later, Romanowicz et al. (1996) observed increased stream water DOC concentrations in the outlet of the watershed. The reason for this was not discussed.

Neal et al. (2004) reported increased DOC concentrations in groundwater and stream water following stem-only harvest of a softwood forest. Groundwater measurements carried out in the paired catchment showed that DOC concentrations increased by +0.05 mg per litre compared with pre-harvest conditions and by +0.19 mg per litre compared with the uncut reference during the first six years. Similarly, stream water concentrations increased by +0.21 (0-2 years) and +0.32 (2-6 years) mg per litre compared with pre-harvest data and by +0.58 (0-2 years) and +0.69 (2-6 years) compared with the uncut reference. Reasons for these increases were not discussed. In a Finnish study by Piirainen et al. (2002), the concentration of DOC from below the O-horizon increased 1-3 years after stem-only harvest, but the retention of DOC increased in the mineral soil (both the E- and B-horizons). The overall flux from below the B-horizon did however not change after cutting. Cutting of the trees removed an important source of DOC but the overall result was an increase in concentration (twice as high in plot 1 and five times as high in plot 2) most likely as a result from increased decomposition and leaching from logging residues and the organic layer. The podzolization process explains an accumulation of DOC in the B-horizon. However, increased retention in the E-horizon could not be explained. During the three-year study period, the increased leaching did not show any signs of levelling off to pre-harvest conditions.

In the Appalachian Mountains, Qualls et al. (2000) measured soil water concentration and fluxes of DOC from a stem-only harvested hardwood forest. The results were compared with an uncut control forest. During the two-year sampling period, the concentration and fluxes of DOC was higher in clear-cut plots compared with uncut plots for all horizons. The largest difference was noticed in the leaching from slash, which was 2.6 times higher than from throughfall. Thus, logging residues were the main DOC source in the cut plots, while most of the DOC originated from the forest floor in the growing forest. Also, a greater proportion of DOC was retained in the A-horizon from the cut plot, but increased runoff and higher concentrations resulted in higher DOC fluxes. In accordance with the results of Piirainen et al. (2002) and McLaughlin & Phillips (2006), they attributed the decrease in DOC to adsorption to Fe and Al-oxyhydroxides. In terms of fluxes, Katzensteiner (2003) measured higher soil water DOC fluxes from the O-horizon in a clear-cut compared with a 10-year old regenerating stand and a mature forest (140 years) during a three-year period. These differences were attributed to increased decomposition in the organic layer and leaching of soluble DOC from the logging debris. However, at all three sites, DOC fluxes decreased with time.

In contrast to these results, Mannerskoski et al. (2005) did not find any significant treatment differences in groundwater DOC concentrations five years after stem-only harvest compared with growing uncut forest. Also, Rusanen et al. (2004) measured no short-term change in DOC concentration in spring water from large aquifers. The lack of response could be due to the large volumes of water stored in the aquifers, which would “buffer” small changes in groundwater chemistry. In addition, only 17% of the catchment was clear-cut over a period of three years.

In Sweden, Lundin (1999) observed short-term increases in stream water DOC concentrations after stem-only cutting varying between 1-21 mg per litre compared with pre-harvest data. In contrast, Löfgren et al. (2009b) reported no significant stream water changes after stem-only cutting, except for one site where higher DOC concentrations were found two years after cutting compared with a control area. Increases in DOC were positively correlated with stream runoff, which showed the largest increases during base flow (lower transpiration rates) and peak flow (larger snow accumulation during winter resulting in larger snowmelt) (Sørensen et al., 2009). Thus, significant increases in DOC occurred during base flow and snowmelt.

Following clear-cutting, DOC concentrations have also been shown to decrease, especially following whole-tree harvest. For example, Ranger et al. (2007) observed short-term (0-3 years) decreasing soil water DOC concentrations (and fluxes) both in the forest floor and in the mineral soil (measured at 15, 60 and 120 cm) after whole-tree harvest compared with pre-harvest conditions. No explanation was given. Long-term 16-18 years) decreases were reported by McLaughlin & Phillips (2006) who measured lower concentrations of DOC in soil water at 25-50 cm depth and in stream water after whole-tree harvesting compared with an uncut control forest. In addition, the concentration of DOC decreased (70%) with increasing soil depth in the clear-cut area, which indicated a high adsorption to Fe- and Al oxyhydroxides in the B-horizon, similar to the findings of Piirainen et al. (2002). McLaughlin & Phillips (2006) suggested that the observed differences between the clear-cut and uncut forest could have been the result of differences in litter input and residual effects from harvesting i.e. mixing of humus and mineral soils. Compared with the reference stand, the DOC flux was lower from the clear-cut stand.

There is only one study by Zetterberg et al. (2008) comparing DOC concentrations between WTH and CH. In general no long-term treatment differences were seen after 27-31 years except for one site where lower concentrations were found.

Generally, clear-cutting seems to increase DOC concentrations and fluxes. In some cases, decreased concentrations have been observed as a result of increased retention in the mineral soil. Processes responsible for the observed changes include:

- Increased decomposition
- Leaching from logging litter and organic horizons
- Increased retention in the mineral soil
- Increased runoff
- Groundwater levels and flow paths

9 Assessment in relation to previous reviews

9.1 Hydrology

The conclusions from this review are in agreement with those of Hibbert (1967), Bosch & Hewlett (1982) and Brown et al., (2005). Hydrological changes following clear-cutting are well documented since the 1960’s. The soil water and groundwater tables usually increase immediately following harvest as a result of reduced evapotranspiration, leading to increased annual stream runoff. As new forest stands and understory vegetation establish, water tables and runoff return to normal levels or even becomes lower compared with pre-harvest conditions due to higher evapotranspiration. Tree mortality might cause long-term temporary increase in runoff. Generally, there is a close relationship between the reduction in forest cover and increase in runoff. Clear-cutting can increase spring flood as a result or larger snow packs during wintertime, but generally the runoff increases are most pronounced during the vegetation period due to the decreased evapotranspiration. Only small changes occur wintertime. Harvesting may also change the magnitude and duration of peak (storm) flow.

A temporary increase in groundwater tables affects decomposition, mineralization and nitrification rates in the soil as it becomes more anaerobic. In addition, redox reactions might occur. Together these processes may lead to decreased concentrations of base cations, NO_3^- and H^+, which in turn have an effect on other soil processes. In addition, stream runoff increases also allows for a greater transport of ions from the system. In conclusion, increased water tables and runoff due to harvesting have a short-term effect on the base cation concentrations (decrease) and fluxes (increase) by affecting decomposition and mineralization. In addition, reduced nitrification and subsequent release of NO_3^- ions may reduce the base cation fluxes even further.

9.2 pH

In general harvest effects on soil exchangeable pools (H^+) are small and mostly non-significant. However, the short-term effect on soil pH can be high. For example, there are a number of studies where forest floor and mineral soil pH have been shown to increase during the first five years in the range of +0.3 to +0.9 (n=3) and +0.1 to +0.4 (n=9) units, respectively, following whole-tree and stem-only cutting. These short-term increases are mainly a result of decomposition, mineralization and litter leaching, causing the concentrations of Ca^{2+}, Mg^{2+} and K^+ to increase in the soil solution. Base cations replace H^+ at the soil exchange complex leading to an increased soil pH. It appears that H^+ is most often exchanged for Ca^{2+} (Katzensteiner, 2003; Hendrickson et al., 1989; Simard et al., 2001). Lower rates of nitrification would operate in a similar way due to a lower production of H^+. After a few years, pH returns to pre-harvest conditions or values similar to uncut control forests. However, at some sites medium- to long-term pH increases have been observed in the forest floor (+0.2 to +1.2 units, n=3) and mineral soil (+0.1 to +0.6 units, n=6) up to seventeen years after cutting. These long-term changes can also be explained by base cation concentration changes (Simard et al., 2001), but also by increased CEC due to the build-up of soil organic matter (McLaughlin & Phillips, 2006).

Short- to medium-term pH decreases in the forest floor (-0.5 units, n=2) and mineral soil (-0.6 to -0.2 units, n=8) have been found in a few studies. In soils with high nitrification rates, pH usually decreases. Also a decrease in decomposition and mineralization, e.g. as a result of higher groundwater tables, may also lower pH by accumulation of organic matter. Long-term pH decreases occur as a result of natural biological acidification via plant uptake.

The conclusions regarding pH are partly in agreement with previous reviews (Sinclair et al., 1992; Egnell et al., 1998; Swedish Energy Agency, 2006). Egnell et al. (1998) concluded that "*there is a temporary (10-20 years) increase in soil pH after clear-cutting which is 0-0.4 units lower in the humus layer following whole-tree harvest compared with stem-only. There are no observed pH effects in the mineral soil.* However, this review shows that mineral soil pH increases cannot be ruled out.

Differences in pH between whole-tree and stem-only harvest are usually too small to be observed in the soil. Thus, in most studies, treatment differences are non-significant except for some Swedish studies (Staaf & Olsson, 1991; Olsson et al., 1996b and Olsson et al., 2004) where lower forest floor pH (but not in the mineral soil) has been reported after whole-tree harvest compared with stem-only. With time, these differences have disappeared. Sinclair et al. (1992) concluded that "*the increase in pH after felling is smaller after whole-tree harvest compared with stem-only"* and the Swedish Energy Agency (2006); "*whole-tree harvest temporary* *lowers the concentration of exchangeable H^+ in the humus layer while the mineral soil remains unaffected"* which is in agreement with this review.

Clear-cutting effects on pH in soil water, groundwater and stream water are more conclusive. Generally, pH decreases are consistent with increased NO_3^- concentrations, indicating net-nitrification, and the release of organic acids from logging debris (e.g. Tremblay et al., 2009). Reported pH decreases in soil water varies between –0.3 to -0.6 units (forest floor) and -0.07 to -0.2 units (mineral soil ≥50cm). Decreases in soil water pH in the 0-50 cm mineral soil have not been specified. Groundwater and surface water decreases varies between -0.14 units and -0.03 to -0.8 units, respectively. Although not as common, there are also reports of higher soil water pH as a result of decreased nitrification (Ranger et al., 2007), decomposition and mineralization (Piirainen et al., 2004) and on rare occasion, bicarbonate production (Parfitt et al., 2002). Long-term increases in pH may results from increased CEC and weathering processes releasing buffering base cations to the soil solution (McLaughlin & Phillips). These results agree with those of Feller (2005) for soil water and stream water where clear-cutting "*lowers the pH in soil water and stream water in the order of magnitude of 0.2-0.5 pH-units as a result of nitrification and release of organic acids*". However, increases in pH were not discussed.

In conclusion, clear-cutting most often causes temporarily higher pH in soil, but lower pH in soil water, ground water and surface water. Generally, there are no differences in pH effects between whole-tree and stem-only harvest.

9.3 Carbon (C) and CEC

Organic matter plays an important role for the cation-exchange capacity of soils. Hence, a reduction in organic matter content may decrease the buffering capacity of the soil and subsequently increase the sensitivity to acid inputs. In this assessment, based on a limited number of C studies, total C concentrations appear to be largely unaffected by tree harvest although there have been some records of decreased concentrations in the forest floor and small concentration increases in the mineral soil. Total C pools seem to decrease in the humus layer and increase in the mineral soil as a result of accelerated decomposition and a downward transport and retention. With time, forest floor C pool rebuilds as a result of long-term input of litter. Most treatment differences are non-significant between whole-tree and stem-only harvest.

The data suggest that the overall effects of cutting on total C concentration are small, which is in agreement with the review of Kreutzweiser et al. (2008). Out of the 14 sites studied, 76 % showed no change in soil C concentrations following clear-cutting (forest floor and mineral soil horizons combined). Thus, only 24 % of the sites showed increases or decreases in soil C concentrations. The decreases were attributed to lower litter inputs while increases were not explained.

The assessment indicates decreasing forest floor C pools after clear-cutting. The effects on this soil layer are seldom discussed in other reviews. However, Sinclair et al. (1992) suggested "*the volume of humus will decrease"*. Increased mineral soil pools of C after clear-cutting are in agreement with the review of Sanchez et al. (2006) who used data from the Long-Term Soil Productivity (LTSP) database following organic matter removal, including stem-only and whole-tree harvest treatments. During the first five years after clear-cutting, increases in mineral soil C pools were observed at most sites attributed to rapid decomposition of dead roots and root exudates from the new forest stand. The results are partly in agreement with Johnson & Curtis (2001) who found that the pools of mineral soil C increased following stem-only harvest with 18 %. However, following whole-tree harvest, mineral soil C pools decreased with 6 %. Their conclusion is based on a much larger material and probably more conclusive. In contrast to these results, Egnell et al. (1998) concluded "*there is no short-term (1-16 years) effect on soil C pools".*

Differences between harvest intensities (WTH versus CH) were reviewed by Rosén (1991) who concluded that *"the risk of a significant decrease of soil organic matter following whole-tree harvesting cannot be verified"* and *"thus, it appears that losses in soil C content are restricted to the forest floor and is of short duration. In Sweden, where harvest of logging residues is mainly carried out in coniferous forests with higher biomass production, treatment effects could appear after harvest*". The result from this report verifies these conclusions based on a small number of data.

Due to the strong interrelation between C and CEC, the short- and long-term treatment effects on CEC are usually small and non-significant, regardless of harvest method. However, just like soil C, individual observations of increases and decreases in CEC have been documented. A decrease in the amount of organic matter or pH in the upper part of the soil leads to a decreased CEC, while increased humification, weathering and pH contribute to an increased CEC. Post-harvest effects between different treatments (WTH↔CH) are often insignificant. An increased CEC may counterbalance base cation leaching.

9.4 Nitrogen (N) and major anions (NO_3^-, SO_4^{2-}, Cl^-, DOC)

Generally, the NO_3^- concentrations increase after harvest compared with pre-harvest conditions and uncut controls, regardless of harvest method. In this review, the short-term (≤5 years) increases in soil water vary between +9-355 (forest floor), +23-6640 (≤50 cm) and +2.4-355 (≥50 cm) µeq per litre. The NO_3^- concentrations increase in groundwater and stream varies +36-560 and +0.05-5790 µeq per litre, respectively. During the clear-cut phase, nitrification continues releasing NO_3^- and H^+ to the soil solution. Low plant uptake means that the ions can be leached to deeper soil horizons and ultimately the stream water. Increased NO_3^- concentrations most often coincide with increased base cation concentrations especially K^+. Microbial immobilization and mechanical mixing of the forest floor and mineral soil also influence the rate of nitrification. In some cases temporary long-term changes has been observed as a result of tree mortality. In contrast, decreased NO_3^- concentrations (and K^+) have been reported as a result of cutting nitrogen fixating stands e.g. alder, dilution, denitrification, rapid revegetation etc. Also, excess NO_3^- leaching may be affected by the choice of silvicultural practices e.g. buffer zones, partial cutting etc. Soil water studies between whole-tree and stem-only harvest indicate that removing the logging residues can reduce the NO_3^- leaching. Whether or not this is an effective method for reducing excess NO_3^- leaching to streams cannot be determined since there are no stream water studies uniquely comparing the treatment differences.

Increased NO_3^- losses are consistent with decreased total N concentrations and pools in the forest floor and mineral soil. The total N decreases are usually associated with decomposition (smaller soil pools) and increased runoff (loss of total N via leaching). Increased soil temperatures and moisture accelerates the N decomposition. In contrast, increases in total N concentrations, pools and fluxes may result from litter input and forest floor mass increases, retention deeper down in the soil, N-deposition, N-fixation and plant uptake (N recirculation). According to Egnell et al. (1998) the clear-cutting *"effect on N pools is not clear although there are some reports of lower soil pools in the humus layer and mineral soil".* This is only partly true since soil N pools have been shown to increase in some studies due to retention. Kreutzweiser et al. (2008) reviewed clear-cutting effects in the boreal forests of North America. In terms of total N, *“the effect on total soil N (pools and concentrations) is variable and difficult to predict and highly site-specific. With respect to the 13 studies reviewed, 73 % of the soils showed no change. Compared with the mineral soil, changes (both increases and decreases) are more likely to occur in the forest floor”.* In a meta-analysis, Johnson & Curtis (2001) also found that soil N (and C) increased following stem-only cutting by +18% but decreased with -6% after whole-tree harvest, in coniferous forests only. Despite these changes Johnson & Curtis (2001) concluded that the effect of harvesting is small, which is in agreement with the earlier review by Johnson (2002).

There are a number of reviews assessing clear-cutting effects on NO_3^- concentrations and fluxes in groundwater and stream water (e.g. Vitousek et al., 1979; Vitousek & Melillo, 1979; Martin et al., 1984; Binkley & Brown, 1993; Brown & Binkley, 1994; Feller, 2005; Gundersen et al., 2006). In general, the results from these assessments are in agreement with those in this assessment. However, in an old Swedish review, Rosén et al. (1991) concluded that *“mineralization of nutrients (N, P, K and Mg) and nitrification in the forest floor decrease after whole-tree harvesting”,* and *“in terms of nutrient leaching, whole-tree harvesting only have a minor effect”.* Clearly, this is not true since increased NO_3^- concentrations and fluxes have been observed following both stem-only and whole-tree harvesting. But, Rosén et al. (1991) pointed out *“leaching below piles of slash may be substantial”.* Leaching from the slash and increased mineralization below the piles partly explains the observed differences between stem-only and whole-tree harvested plots. Egnell et al. (1998) on the other hand stated, “*clear-cutting increases nutrient leaching. Compared with stem-only harvest, nutrient leaching is lower following whole-tree harvesting”* which is in agreement with this assessment. Later on, the Swedish Environmental Agency (2006) concluded that *“whole-tree harvesting reduces the concentration of* NO_3^- *in the humus layer of the soil and sometimes in the soil water, but the effects are not clear”.* These results can be questioned since lower soil concentrations have been found in a number of studies (Rosén Lundmark-Thelin, 1987: Emmett et al., 1991; Staaf & Olsson, 1994; Titus et al., 1997). Thus, increased nitrification will lead to short-term acidification effects and NO_3- associated leaching of base cations from the system. The potential for nitrification can sometimes be reduced via whole-tree harvesting.

There are many reports of short-term decreases in SO_4^{2-} concentrations in soil water following stem-only and whole-tree harvest. The decreases range between -250 to -6 (0-50 cm) and -26.5 (≥50 cm) µeq per litre. Decreases in the forest floor have also been observed but not quantified. Changes in groundwater and stream water vary between -241 to -84 and -217 to -10 µeq per litre, respectively. There are a number of processes responsible including dilutions, inhibition of S oxidizing bacteria, increased plant uptake, anaerobic conditions, microbial immobilization and less dry deposition. During periods of high atmospheric deposition a reduction in dry deposition would certainly have been responsible for much of the change. However, due to the obtained emission reductions, the total deposition of SO_4^{2-} has tangibly decreased while other factors become more important. In areas where SO_4^{2-} has accumulated in the soil, clear-cutting could delay the aquatic recovery process via the release of SO_4^{2-} during decomposition. Increased SO_4^{2-} concentrations have been reported in a number of studies as a result of increased decomposition and mineralization, leaching from logging residues and dead organic matter, decreased plant uptake and desorption. The chemical transformations, which individually or combined could explain decreased stream water concentrations were summarized in a review by Feller (2005). He concluded that the process most likely responsible for decreased stream water concentrations is increased SO_4^{2-} adsorption in the soil.

The Cl^- concentrations have been found to decrease following clear-cutting as a result of reduced dry deposition and/or dilution. Decreases in soil water are seldom quantified but has been shown to decrease in the range -72 to -32 (n=4) in groundwater µeq per litre and -66 to -35 (n=2) µeq per litre in stream water. Treatment differences between stem-only and whole-tree harvests appear to be small and insignificant. However, increased Cl^- concentrations have frequently been observed following both stem-only and whole-tree harvest. In most cases there seem to be an internal source of chlorine from which inorganic Cl^- is released via leaching or decomposition. Recent findings by Svensson et al. (2006) indicate that the soil pools are dominated by organic chlorine, while the fluxes are dominated by inorganic Cl^-. The amounts of organic chlorine stored in organic matter are of the same order of magnitude as the amounts phosphorous (P: 0.03- 0.2% and Cl: 0.01-0.5% dry matter) (Öberg, 1998). Accelerated decomposition after harvest could thus lead to the formation of inorganic Cl^-, which is easily leached from the system together with base cations. So far, this process has been largely overlooked in the literature. Except for a brief mentioning by Feller (2005), neither of the reviews mentioned in chapter 3 discussed increased Cl^- concentration and the potential acidification effects. As the SO_4^{2-} concentrations in soil water and surface waters have decreased, the relative importance of other mobile anions for the base cations leaching has increased.

Based on few studies, the DOC concentrations in soil water, groundwater and stream water have increased during the short-term, clear-cut phase. The long-term effects are largely unknown. Decomposition and leaching from the logging residues and organic horizons have been proposed as responsible for the increased DOC concentrations. Increased DOC concentrations in soil water does not necessary lead to increased stream water concentrations since DOC is effectively retained in the B-horizon of podzols. These results are in agreement with the review of Kreutzweiser et al. (2008) who found that "*disrupting the carbon cycle usually leads to increased DOC concentration both in soil water, streams and lakes attributed to a rise in water tables, leaching from slash and increased decomposition*". Studies linking soil water and groundwater changes to stream water are rare. In terms of DOC it has been shown that the concentration and export of DOC to a large extent depends on catchment characteristics and seasonality (Ågren et al., 2007). Bishop et al. (2004) have shown the importance of the near-stream zone and hydrological flow paths. These studies indicate that the stream water response following cutting may differ significantly from soil water and groundwater. This is in agreement with the review by Hope et al. (1994) who concluded *"differences in the export of DOC sometimes depend on hydrology, season and antecedent weather conditions and not the actual cutting".*

Changes in anion concentrations (NO_3^-, SO_4^{2-}, Cl^- and organic anions/DOC) have a large impact on the base cation leaching. Neal et al. (1998a) studied individual effects of SO_4^{2-}, NO_3^- and Cl^- on four acidification indices using multiple regression analysis (Figure 27). The general effect of increasing anion concentrations was an increase in Al^{n+} and Ca^{2+} and a decrease in alkalinity and ANC (Figure 27). Also, the influence of individual anions varied. For example, compared with Cl^- and SO_4^{2-}, the effect of NO_3^- on alkalinity and ANC was higher. In comparison, an increase in SO_4^{2-} had a greater effect on Ca^{2+} concentration than NO_3^-. Also, the effects take place at different soil depths. For example, the small effect on Al^{n+} by Cl^- indicates that the decrease in alkalinity and ANC occurs in the organic layers of the soil. Following the results from Neal et al. (1998) harvesting effects on individual anions should be discussed separately.

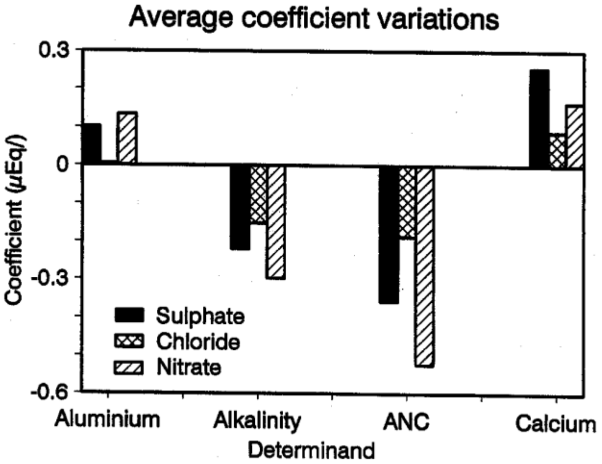


Figure 27. Based on nineteen sites, average coefficient variations could be determined for the acidification indices (Al, alkalinity, ANC and Ca^{2+}). If NO_3^- decreases by 1 µeq per litre, Gran alkalinity will decrease by 0.3 µeq per litre. The remaining 0.7 µeq per litre is consumed by weathering, during which aluminium is released, From Neal et al. (1998a).

9.5 Base cations (Ca^{2+}, Mg^{2+}, K^+)

Concern has been raised about forest productivity and risk of acidification when the loss of base cations exceeds input via weathering and deposition. In areas subjected to acid deposition and loss of buffering capacity, whole-tree harvest may exacerbate the negative effects of acid deposition. Following clear-cutting, short- and long-term increases in exchangeable and total base cation concentrations have been observed, both in the forest floor and mineral soil. Decomposition of fresh litter and leaching from logging residues left on ground is an immediate source of base cations. With time, decomposition of woody debris and roots becomes more important. An increase in soil moisture facilitates a downward movement of base cations in soil solution. Biological and/or mechanical mixing of organic and mineral soil layers accelerates the decomposition rates even further. Together these processes are responsible for the increases in exchangeable base cation concentrations in the forest floor and upper mineral soil. Increases deeper down in the mineral soil are usually explained by increased CEC offering more exchange sites to which base cations can absorb. In soils with extensive nitrification, exchangeable base cation concentrations in the mineral soil decreases via cation exchange processes with H^+. Generally, the base cation trends in soil water, groundwater and stream water mirror those in the soil.

Egnell et al. (1998) concluded that the soil base cation pools and base saturation decrease following whole-tree harvesting. This is only partly true according to the reviewed studies. Kreutzweiser et al. (2008) reviewed changes in stream water concentration and export after CH and WTH in the North America boreal forest and showed that the base cation fluxes are likely to increase following harvest as a result of increased water fluxes. However, the soil concentrations (total and exchangeable) do not always change.

10 Conclusion

Following the abrupt removal of trees in final cutting, soil properties will be affected resulting in changed concentrations, pools and fluxes of many constituents. The quantitative importance and duration of these changes differ between sites. Treatment differences between whole-tree and stem-only harvest are difficult to quantify, which suggests that the effect of cutting per se is more important than the differences in harvest intensity, especially during the clear-cut phase. In general, short-term effects (equal to the clear-cut phase) are well studied, especially in forest soils, soil water and stream water, since this is usually the time when the largest changes take place. Groundwater studies are largely lacking. Clear-cutting effects should also be viewed in a catchment perspective, covering the effects on the entire soil to stream water system. This type of studies should therefore be prioritized in future research.

In addition, long-term effects are rarely described due to the lack of long-term data. There are no studies describing treatment effects older than 31 years. Considering normal forest rotation periods (60-120 years) and that it may take decades for soil changes to become detectable, continued monitoring in already established research sites should be supported. This becomes even more apparent when considering the use of steady-state mass-balances and dynamic modelling for predicting long-term changes. According to the model simulations, clear-cutting, especially whole-tree harvest, has a long-term negative impact on soil nutrients which cannot always be compensated for by weathering processes and deposition. However, based on the empirical data assessed in this review, the short-term effect of clear-cutting final felling is relatively small even though there are sites where the effect of cutting has been considerable. The long-term effects are largely unknown. Thus, there is a discrepancy between the empirical data and model predictions. A summary of the results from this assessment is given below:

Hydrology:

- Following harvest, groundwater tables and runoff usually increases as a result of reduced evapotranspiration varying between -3 to +96 cm (groundwater), -52 to +103 mm (soil water) and +4 to +450 mm (annual stream runoff). In the Nordic countries, runoff increases between 90-371 mm (average 212 mm) have been observed.
- With time, groundwater tables and stream runoff returns to pre-harvest conditions or values similar to an uncut forest. High transpiration rates by young forest stands may occasionally lead to lower runoff than before harvest.

pH and H^+:

- In general, the short- and long-term effects of harvest on exchangeable pools of H^+ are small and non-significant, regardless of harvest intensity. However, the short-term effect on soil pH can be high, both after stem-only and whole-tree harvest. Decomposition of organic matter, mixing of organic matter and mineral soil, nitrification and changed base cation concentrations are plausible explanations.
- Short- to long-term pH decreases in the forest floor may take place after cutting as a result of biological acidification. pH changes in the mineral soil appear to be linked with increased nitrification or increased groundwater tables, slowing down the decomposition processes.
- Treatment differences between stem-only and whole-tree harvests are rare, but when measurable, whole-tree harvest leads to higher acidity.
- Soil water, groundwater and stream water studies frequently report decreased pH regardless of harvest intensity as a result of nitrification. This effect is of relatively short-duration as the new forest stand establishes and NO_3^- concentrations returns to pre-harvest conditions.

Carbon and CEC:

- Based on few observations, the harvest effect on the total C concentrations is small and non-significant regardless of harvest method.
- Clear-cutting may lead to short- and long-term decreases in the forest floor C pools and increases in the mineral soil C pools. With time, the forest floor C pool rebuilds as a result of long-term input of litter

- Most studies indicate that there is no difference between WTH and CH indicating that whole-tree harvest does not lead to soil C losses. However, a larger dataset should be considered since the review of Johnson & Curtis (2001) suggest otherwise.
- The treatment effect on CEC is often non-significant. However, increases and decreases have been observed both in the forest floor and mineral soil. The treatment effects on CEC are coupled to the organic matter quantity, decomposition rates, weathering rates, clay minerals and changes in pH.

Nitrogen:

- Harvest effects on the soil N concentration and pools are similar to those observed for soil C.
- Short- and long-term decreases in the forest floor soil N concentrations and pools have been registered following whole-tree and stem-only harvest as a result of decomposition.
- Short- and long-term increases in the forest floor soil N concentration and pools could also occur e.g. as a result of plant uptake from deeper soil horizons which accumulates in the forest floor via litter input.
- Short- and long-term increases in the mineral soil N concentration and pools have frequently been reported due to leaching and retention. In contrast, mineral soil N concentration and pools could also decrease as a result of leaching.
- Short- and long-term treatment differences between WTH and CH are usually non-significant but there are examples where WTH has lowered the concentrations and pools of total N during the first five years.
- The effects of cutting on soil N appear to be site-specific and difficult to predict.
- Decomposition of total N and nitrification are key processes for the observed increases in soil water, groundwater and stream water NO_3^- concentrations and leaching of Ca^{2+}, Mg^{2+} and K^+.

Anions (NO_3^-, SO_4^{2-}, Cl^- and DOC):

- NO_3^- concentrations and fluxes usually increase during the clear-cut phase, regardless of harvest method. Enhanced NO_3^--leaching is often associated with increased base cation, especially K^+, concentrations and fluxes. The magnitude of change and the duration of the increased leaching vary from site to site.
- SO_4^{2-} concentrations and fluxes usually decrease following harvest, regardless of cutting method. During times of high deposition, a reduction in dry deposition often explained much of the observed decreases. Total S deposition has decreased why other processes become more important such as S retention and SO_4^{2-} desorption.
- Decreases in Cl^- concentrations usually follows clear-cutting, especially in coastal areas, as a result of decreased dry deposition.
- Increases in Cl^- concentrations have frequently been reported suggesting an internal source of chlorine.
- DOC concentrations and fluxes often increase following clear-cutting.
- Following the law of electron neutrality principle, changes in anion concentrations will have an impact on the base cation concentrations.

Base cations (Ca^{2+}, Mg^{2+} and K^+):

- Clear-cutting effects on Ca^{2+}, Mg^{2+} and K^+ results are complicated to assess due to the large variation of analytical methods used for soils.
- Decomposition and nutrient release from logging residues and organic soil layers are the primary sources of Ca^{2+}, Mg^{2+} and K^+.

- Clear-cutting favours decomposition and mineralization. However, drying out of the top-most soil layer may slow down the rates.
- Compared with Ca^{2+} and Mg^{2+}, K^+ is quickly leached from the logging residues. During periods of nitrification, excess NO_3^- leaching increases the base cation losses, especially K^+. The base cations may be retained further down in the mineral soil.
- Short- term increases in the BC concentrations (total and exchangeable) in forest floor and mineral soil usually follows clear-cutting as a result of decomposition, deposition, weathering, litter leaching and mechanical/biological mixing.
- In the mineral soil, increases in BC concentrations (total and exchangeable) are explained by higher CEC or mineral fixation of K^+.
- BC concentration decreases (total and exchangeable) often results from cation exchange processes, which generally coincides with extensive nitrification and release of H^+. In addition, a loss of organic matter may lead to a decrease in concentrations and pools.
- Short- and long-term treatment effects on the base cations pools (exchangeable and total) are usually statistically insignificant.
- When measurable, decreasing base cation pools generally take place in the upper parts of the soil due to decomposition and leaching.
- Increases in mineral soil base cation pools results from weathering and adsorption of base cations leached from higher soil layers.
- In general, differences between whole-tree and stem-only harvest (WTH↔CH) are statistically insignificant in terms of exchangeable base cation concentrations.
- When measurable, lower exchangeable concentrations of Ca^{2+}, Mg^{2+} and K^+ in the forest floor and in the mineral soil (0-10 and ≥10 cm) have been reported after whole-tree harvest compared with stem-only harvest for periods ranging between 3-21 years. The mechanism responsible for the observed differences is a relocation of base cations from decomposing logging residues to the soil exchange complex in stem-only harvested plots.
- Soil water, groundwater and stream water base cation concentrations usually increase after clear-cutting regardless of harvest intensity.
- Base cation fluxes increase as a result of increased groundwater tables and stream runoff.

The short-term (decades) empirical results versus steady-state mass-balance models:

- In general, there is a discrepancy between the empirical data in this assessment and mass-balance budget calculations for base cations.
- According to the mass balances, weathering and deposition alone cannot compensate for losses of Ca^{2+} and Mg^{2+}, regardless of harvest method (CH and WTH). Based on Swedish mass-balances, losses of K^+ following CH in Scots pine stands are compensated for via weathering and deposition but not in Norway spruce stands following WTH.
- In this review, the long-term studies indicate that there is a relocation of BC from the forest floor and upper mineral soil to lower horizons. Additionally, there are indications on increased weathering rates following whole-tree harvest compensating for the BC-losses. Thus, long-term acidification effects due to BC-removal are still uncertain. Such feedback mechanisms are difficult to predict but necessary to include in the mass-balance models.
- In this review, it has been shown that BC-losses do occur via biomass removal and anion-associated leaching during the clear-cut phase. Regardless of harvest intensity, and thereby biomass BC-removal, it appears as if nitrification and NO_3^- leaching is a key process for surface water acidification.

- The data in this review cover only 0-30 years due to the lack of long-term experimental data. In addition, the studies may not be representative. It can therefore not be disregarded that acidification effects may appear later on in the succession, as indicated by the mass-balance simulations.

Implications for dynamic models such as MAGIC for simulating long-term biomass harvest acidification effects, the following issues are important:

- When modelling long-term acidification trends in first order streams following clear-cutting and aboveground biomass removal, different landscape units and hydrological flow paths within a catchment must be considered.
- Conceptually, processes in both recharge and discharge areas (including the riparian zone) should preferably be simulated. Thus, when implementing the model, a minimum of two horizontally compartments should be used.
- The zonal appearance and chemical characteristics of podzols commonly found in Swedish forest soils have an impact on concentrations, pools and fluxes. Accordingly, a two-box model should represent the recharge compartment (forest floor & mineral soil) while a one-box model is sufficient for discharge areas.
- With time the forest floor builds up affecting CEC. At present, CEC is considered constant in MAGIC but should vary with time.
- Nitrification and NO_3^- leaching are key processes in terms of base cation leaching, especially for K^+. The effect is usually short-term (≤5 years) but may remain elevated for periods up to 10 years. Hence, different scenarios on nitrification should be simulated.
- There are indications on increased weathering rates following harvest, which may compensate for the removed base cations. Hence, different scenarios on weathering should be simulated.

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