2 ECOLOGICAL FEASIBILITY OF FOREST CONVERSION: Can conversion of secondary coniferous into deciduous forest enhance environmental sustainability?

2.1 A meta-analysis of forest type effects on element deposition and leaching

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2.1.1 Introduction

Currently, Norway spruce and Scots pine forests are distributed far beyond their assumed natural ranges in Europe. Conversion of secondary spruce or pine forests on sites naturally more suited to broadleaves is generally agreed on in Europe (Spiecker *et al* 2004). Such conversion mainly comprehends an increase in the share of broadleaves. This can range from the complete replacement of a coniferous stand by a new deciduous stand to the gradual establishment of an admixture of broadleaves in a conifer dominated stand in the long term (Gartner & Reif 2004).

The rationale for forest conversion is mainly sylvicultural. Pure spruce stands are generally considered to be highly unstable and prospects are that this will persevere in the light of climate change, especially/particularly outside the natural distribution range of Norway spruce. A decrease in profitability as compared to a pure Norway spruce forest is expected in a mixed forest, e.g. of Norway spruce and beech, but for risk-averse investors this will be reimbursed by the risk abatement (Knoke *et al* 2005). Nature conservation and protection of soil and water are further important arguments for conversion, because environmental consequences of this forest conversion are expected to be positive (Gartner & Reif 2004, De Schrijver *et al* 2004, Von Wilpert *et al* 2000). Conifers have the bad reputation of speeding up forest soil acidification (lower soil pH and lower base saturation) compared to deciduous trees because coniferous litter is more resistant to biological degradation (see Section 2.4) and more organic acids leach from it (Ranger & Nys 1994, Howard & Howard 1990, Johansson 1995). Soil biological activity and therefore nutrient turnover is reported to be lower under conifers (Binkley 1996, Saetre 1998). Furthermore, coniferous canopies more efficiently scavenge atmospheric pollution, enhancing acid deposition through higher dry deposition of SO_4^{2-}, NH_4^+ and NO_3^- (De Schrijver *et al* 2004). Also the uptake and retention of N is reported to be higher in deciduous forest (Cole & Rapp 1981). An overall N retention of 96% in a deciduous plot and 85% in a coniferous plot was found after a nine-year period of chronic N addition (Magill *et al* 2000). From the combination of higher input and lower retention of N in a coniferous forest ecosystem, higher percolation of NO_3^- to the groundwater and associated base cation losses are expected (Schulze 2000).

Contrary to this are the results of recent forest type effect studies on European and national level (Kristensen *et al* 2004, Borken & Matzner 2004). Higher NO_3^- percolation from deciduous forests was found when comparing nutrient cycling in the Level II monitoring plots of the ICP network. The cause of this phenomenon is most likely that deciduous forests, that naturally dominate a broad range of sites, were replaced at large scale during the last centuries by conifers on the poorer sites across Europe, often carrying degraded forests, heath lands or marginal farmlands (Johann *et al* 2004). The sampling scheme of the Level II sites, intended to represent the European forest situation, reflects this distribution (Fischer *et al* 2003). In Denmark, higher NO_3^- concentrations were indeed found under forests on fine textured soils as compared to coarse textured soils, irrespective of tree species (Callesen *et al* 1999).

These contrary results lead us to the question which environmental effect can be expected from converting coniferous forest to deciduous forest. The only correct approach to answer this question is by comparing deciduous and coniferous forest stands under comparable local site conditions. Climate (Erisman & Draaijers 2003), soil type (Silva *et al* 2005, Boumans *et al* 2004), land use history (Matson *et al* 2002), pollutant emission level (Matson *et al* 2002), succession phase (Aber *et al* 1989, Agren & Bosatta 1988, Cairns & Lajtha 2005) and experimental set-up (Bleeker *et al* 2003) are all known to have a significant impact on the element fluxes observed. The aim of this paper is to analyse existing research results of element stand (throughfall + stemflow) deposition flux and percolation flux to groundwater in coniferous and deciduous forests that meet the prerequisite of comparability of forest stands.

2.1.2 Methodology

Data collection

A literature survey was performed starting on the Web of Science (ISI Web of Knowledge v3.0) to obtain published studies comparing values of stand deposition and percolation fluxes of elements in pairs of forest stands solely differing in tree species composition. Afterwards the cited references of these studies were searched for other peer-reviewed publications. The forest stands in each pair had to be identical concerning mesoclimate, soil type, land use history, pollutant emission level and forest age as well as experimental set-up.

We considered 19 comparative studies analysing 25 pairs of forest stands for element stand deposition flux and 8 stand pairs for element percolation flux (List available from the authors). In these pairs, specific element fluxes were determined (i) in throughfall, sometimes (ii) in stemflow, and (iii) in wet-only or bulk precipitation in an adjacent open space. Element percolation in the soil profile was measured by different types of lysimeters at a depth assumed to be underneath the main rooting zone. In the studies considered, the sampling depth ranged between 45 and 175 cm (List available from the authors). The collected data summaries (mean annual fluxes) were recalculated in the SI-unit mol ha^{-1} y^{-1} and entered in a database.

Data analysis

To compare the results of the different studies, we performed a meta-analysis as described in Gurevitch & Hedges (2001). Because the site conditions between the stand pairs differed considerably, the parameter used from each pair was the ratio between the element flux in the coniferous forest stand and the element flux in the deciduous forest stand. We calculated these ratios both for stand deposition and for percolation fluxes.

We considered this forest type effect as random over all the stand pairs. Random variation within one stand has two possible sources: (i) space, the number and the choice of the location of collectors (e.g. Van Ek & Draaijers 1994) and (ii) time, variation between a number of measuring years (e.g. Houle *et al* 1999). Between stand pairs, the sampling effort, i.e. number of collectors and number of years of observation, can influence the observed mean. Moreover, nutrient fluxes vary considerably between years with varying meteorological conditions (Erisman & Draaijers 2003). Analytical accuracy of neither chemical analyses nor errors in water flux calculations were considered, because rarely any study provides information on these.

The lack of reporting on sample standard deviation in most studies negates the use of the response ratio metric (R_x) over the d-index for a meta-analysis (Hedges *et al* 1999). We calculated a weighted mean ratio for each element (using data from those studies that provide data on the specific element) when at least four stand pairs were available for the meta-analysis. The flux ratio of each stand pair was log-transformed and a weighted mean over all pairs was calculated using the number of sampling years as weight, since it was the only source of variation consistently reported in all studies (List available from the authors). After back-transformation, a 95% bootstrap confidence interval of this weighted mean was calculated (Gurevitch & Hedges 2001). This approach is less powerful and less accurate than the standard meta-analysis statistical procedures (Gurevitch & Hedges 2001). Calculating the weighted means without logarithmic transformation resulted in comparable results, but with broader confidence intervals (not reported).

The level of atmospheric pollution differed very much between the studies. Especially forest plots in heavily urbanised regions, near industrial plants or near intensive animal breeding farms received much more N deposition than sites in remote, semi-natural areas. To forest ecosystems, an open field deposition level of 10 kg ha^{-1} y^{-1} of total inorganic N (NH_4^+-N + NO_3^--N) seems to be a threshold above which N cycling changes (Kristensen *et al* 2004). In the studies considered, the atmospheric N pollution level remained fairly constant during the measuring period. Exceptions to this are the long-term studies at the Solling site (Matzner & Meiwes 1994) and the Šerlich site (Lochman *et al* 2004, Lochman & Mares 1995), although the open field deposition flux remained above 10 kg N ha^{-1} y^{-1}.

2.1.3 Results

Pairs of mean throughfall deposition fluxes of the following elements were reported in the studies considered: NH_4^+, NO_3^-, total inorganic N, H^+, SO_4^{2-}, Na^+, K^+, Ca^{2+}, and Mg^{2+}. Too few data pairs were available for Al(III) and HCO_3^- to draw meaningful conclusions. Mean percolation fluxes were reported for the NH_4^+, NO_3^-, inorganic N, SO_4^{2-}, K^+, Ca^{2+}, Mg^{2+} and Al(III), while H^+ and Na^+ fluxes were not reported frequently enough to be considered for meta-analysis.

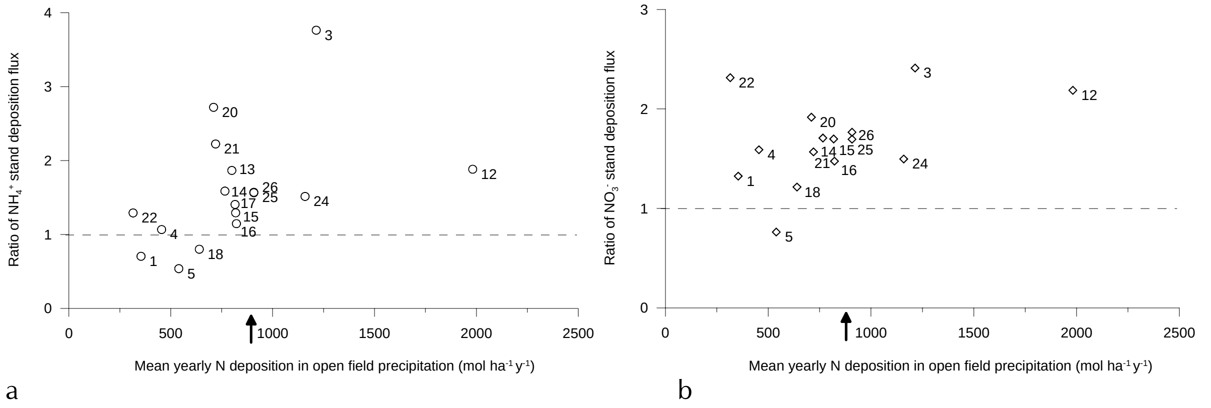
Stand deposition

For the global dataset used for meta-analysis, the ratio of stand deposition under coniferous and deciduous canopies was significantly higher than 1 for all ions other than K^+, for which the 95% bootstrap confidence interval of the mean included 1 (Table 2-1). Considering all available data, the ratio of stand deposition flux under coniferous and deciduous canopy is significantly higher than 1 for both NH_4^+ and NO_3^- and also for their sum (Table 2-1). The mean ratio is higher for NO_3^- (1.50) than for NH_4^+ (1.35), but not significantly. Distinguishing into stand pairs with a low (<10 kg N ha^{-1} y^{-1}) and a high inorganic N deposition in the open field (>10 kg N ha^{-1} y^{-1}) reveals that in the low deposition plots mean stand deposition of NH_4^+ and NO_3^- are not significantly different between the two forest types. However, there exists a clear tendency to a lower NH_4^+ stand deposition in the coniferous stands (mean ratio = 0.83) (Figure 2-1a). On the contrary, for the stand pairs with high open field N deposition, the average stand deposition flux of NH_4^+ is significantly higher in the coniferous stands (mean ratio = 1.66) (Figure 2-1a). For NO_3^-, the variability of the ratios is higher than for NH_4^+. The average ratio (1.27) indicates a higher deposition under coniferous stand, irrespective of the level of open field N deposition (Figure 2-1b). The ratios of the stand deposition fluxes of NO_3^- and of total inorganic N are significantly higher than 1 in the plots with high open field N deposition, but not for the paired stands with lower wet N deposition..

SO_4^{2-} deposition is significantly higher under coniferous canopy than under deciduous canopy (mean ratio = 1.69). H^+ deposition is on average 2.5 times higher in coniferous stands as compared to deciduous stands, but the mean ratio is very variable (confidence interval 1.77-3.57). The mean ratio of stand deposition fluxes under coniferous and deciduous canopy is significantly higher than 1 for all base cations, except for K^+. The ratios for Ca^{2+} and Na^+ are of the same magnitude (1.40 and 1.38 respectively) and both elements have a comparable confidence interval, although the number of replicates for Na^+ is only half of the number for Ca^{2+}. The mean ratio for Mg^{2+} is a little lower (1.26), while for K^+ it amounts 1.01.

Table 2-1: Weighted mean ratio between element stand deposition fluxes under coniferous forest and under deciduous forest. The 95% bootstrap confidence interval is given between brackets. n = number of studies. Results are given for: (1) all available data, (2) data with open field inorganic N deposition flux ≤ 10 kg ha^{-1} y^{-1} and (3) > 10 kg ha^{-1} y^{-1}.

Element	*n*	Mean Ratio (1)	*n*	Mean Ratio (2)	*n*	Mean Ratio (3)
NH_4^+	*17*	1.35 [1.06 - 1.70]	*5*	0.83 [0.56 - 1.09]	*12*	1.66 [1.36 - 2.06]
NO_3^-	*17*	1.50 [1.36 - 2.06]	*5*	1.27 [0.88 - 1.91]	*12*	1.87 [1.51 - 2.37]
NH_4^+ + NO_3^-	*20*	1.61 [1.31 - 1.99]	6	1.08 [0.78 - 1.35]	*14*	1.86 [1.51 - 2.35]
H^+	*15*	2.47 [1.77 - 3.57]				
SO_4^{2-}	*24*	1.69 [1.45 - 1.99]				
Na^+	*10*	1.38 [1.16 - 1.66]				
K^+	*16*	1.01 [0.83 - 1.27]				
Ca^{2+}	*20*	1.40 [1.22 - 1.62]				
Mg^{2+}	*12*	1.26 1.03 - 1.56]				



a

b

Figure 2-1: Scatter plot of (a) the ratio of NH_4^+ deposition flux and (b) the ratio of NO_3^- deposition flux in coniferous paired deciduous stands for increasing mean yearly inorganic N deposition flux in open field precipitation (mol ha^{-1} y^{-1}) adjacent to each stand pair. The arrow indicates 10 kg N ha^{-1} y^{-1} open field deposition. The labels refer to the index numbers of the list of studies (available from the authors).

Percolation

The ratio of ion percolation under coniferous and deciduous stands was significantly ($p < 0.05$) different from 1 for NO_3^-, total inorganic N, and K^+, but not for the other elements considered (Table 2-2. Percolation of NO_3^- was found to be significantly and on average almost three times higher under coniferous forest stands than under deciduous forest stands. The NO_3^- percolation ratio varied strongly between stand pairs, and ranged from 0.78 to 37.5. We could not distinguish between regions with low and high atmospheric N pollution level, because (not all studies reporting on N percolation fluxes included open field N deposition data). Anyhow the number of replicates would become too low for a meaningful analysis and almost all stand pairs were located in a region where wet deposition exceeded 10 kg N ha^{-1} yr^{-1}.

Percolation of H^+ and Al(III) is not significantly different between coniferous and deciduous forest stands (Table 2-2) because the ratios for both elements show a high variability between stand pairs. The mean ratio of 1.71 for both elements may indicate higher percolation under coniferous stands. Also for SO_4^{2-}, Ca^{2+} and Mg^{2+} percolation, no significant differences were found between the two stand types and the ratios close to 1 indicate only small differences between coniferous and deciduous forest for these elements. Of the base cations, only K^+ percolation was significantly higher under coniferous stands than under deciduous stands with a mean ratio of 1.5.

Table 2-2: Weighted mean ratio between element fluxes in percolation water under coniferous forest (C) and under deciduous forest (D). N = number of studies. See Table 2 for calculations of mean ratio and 95% confidence interval

Element	*n*	Mean ratio
NH_4^+	*5*	0.50 [0.11 - 1.22]
NO_3^-	*5*	2.76 [1.17 - 6.49]
inorganic N	*8*	3.05 [1.68 - 5.44]
SO_4^{2-}	*6*	1.11 [0.85 - 1.44]
K^+	*6*	1.51 [1.20 - 1.92]
Ca^{2+}	*6*	1.15 [0.49 - 2.32]
Mg^{2+}	*6*	1.18 [0.89 - 1.59]
Al (III)	*5*	1.71 [0.88 - 4.22]

2.1.4 Discussion

Stand deposition

Atmospheric deposition can be considered as an additional source of nutrients to the 'internal' element cycles in forest ecosystems. Stand deposition flux, calculated as the sum of throughfall and stemflow deposition (fluxes), was assumed to be an adequate parameter for studying the input with the water flux to the forest floor, i.e. the net interaction between atmosphere and forest canopy. Compared to precipitation in the open field, the chemical composition of throughfall and stemflow water is generally altered, and it is widely acknowledged that this transformation results from (i) dry deposition of aerosols and gases on leaves, branches and stems, as well as (ii) canopy exchange processes such as diffusion and/or exchange between the water layer covering leaves/needles and the underlying apoplast and uptake of gases through stomata (Draaijers *et al* 1997).

In general, Na^+ is assumed to be inert with respect to the canopy, i.e. no uptake and no leakage occur (Ulrich 1983). Therefore, the stand deposition flux of Na^+ is often used as an indicator of the dry deposition capacity of a forest. Based on the meta-analysis of data on Na^+ stand deposition flux from 10 stand pairs, it can be concluded that the dry deposition capacity of coniferous forests is significantly higher than that of deciduous forests (about 1.4 times higher), implying that a coniferous stand annually intercepts more atmospheric pollutants than a deciduous stand at the same site. Several studies also consider SO_4^{2-} to be more or less inert with respect to the canopy (Stachurski & Zimka 2000, Ukonmaanaho & Starr 2002). Based on the meta-analysis of SO_4^{2-} stand deposition flux data from 20 stand pair comparisons, coniferous forest has a 1.7 times higher dry deposition capacity than deciduous forest. Important factors for lower dry deposition in deciduous than in coniferous stands are differences in vegetation structure, such as the generally lower LAI, stand and crown density, and the summergreen character of deciduous stands (Alcock & Morton 1985, Draaijers 1993, Houle *et al* 1999, Erisman & Draaijers 2003). IN addition, the leaf shape affects the amount of elements deposited. Leaves with long narrow shape (needles) are more efficient in salt accumulation than circular ones (Woodcock 1953 in Smith 1981). Nevertheless, during the growing season, higher dry deposition fluxes to deciduous forest in comparison to coniferous forests have been observed (Cappellato *et al* 1993).

Opposite to Na^+ and SO_4^{2-}, stand deposition fluxes of the base cations K^+, Ca^{2+} and Mg^{2+} are affected by canopy exchange processes. The effect of forest type can be direct, through differences in canopy characteristics such as aboveground biomass, nutrient amount, species specific concentrations in leaves (Lovett *et al* 1989) and evergreen character (Smith 1981). Nutrient concentrations in broadleaves are higher compared to needles (Johansson 1995). Also the age distribution of leaves affects the magnitude of leaching to a large extent: young immature leaves/needles tend to lose more nutrients compared to older ones except when they are hydrophobic (Parker 1990). The type of forest can also indirectly influence canopy exchange through forest soil characteristics such as extractable amount of base cations in the soil solution (Lovett & Schaefer 1992).

In general, stand deposition flux of K^+ originates over 70% from canopy leaching (Houle *et al* 1999, Van Ek & Draaijers 1994, Parker 1983, Ragsdale *et al* 1992). The fact that K^+ stand deposition does not differ significantly between the two forest types in the meta-analysis may indicate that the higher dry deposition from the atmosphere (soil dust and sea salt) to coniferous forest stands compensates for the higher canopy exchange in the deciduous forest stands (Alcock & Morton 1985, Houle *et al* 1999, Van Ek & Draaijers 1994). Stand deposition fluxes of Ca^{2+} and Mg^{2+} are significantly higher in the coniferous forests considered in the meta-analysis. There is considerably less canopy exchange of these two elements than of K^+ (Ragsdale *et al* 1992), so this again reflects the higher dry deposition capacity of coniferous canopies.

The higher dry deposition capacity can explain the higher stand deposition of inorganic N in coniferous forest. However, this general conclusion is inconsistent with the observed lower NH_4^+ stand deposition under conifers in regions with low atmospheric N pollution. This apparent contradiction can be explained by the fact that forest canopies can take up considerable amounts of N out of the atmosphere. If only wet deposited inorganic N is considered, uptake can already range from 100 to 900 mol N ha^{-1} y^{-1} (Lovett & Lindberg 1992, Harrison *et al* 2000). Previous research indicates that N uptake rates are higher for deciduous leaves of beech and silver birch than for Norway spruce (Brumme *et al* 1992, Harrison *et al* 1991), which is attributed to a combination of thinner cuticles and higher wettability of foliage and bark of the deciduous species. However, this higher uptake rate may be counterbalanced because deciduous trees have foliage for only part of the year (Harrison *et al* 2000) and because coniferous forests have a higher leaf area index. So we hypothesize that, when open field N deposition is low, higher yearly canopy uptake of NH_4^+ by coniferous forests may keep pace with the higher dry deposition as compared to deciduous forest, resulting in the observed lower NH_4^+ stand deposition flux under conifers. In regions with high atmospheric N pollution, the canopy uptake of NH_4^+ may be obscured by elevated dry deposition, with a higher stand deposition flux of NH_4^+ in coniferous forest as a result. This hypothesis can however not be corroborated, since still no conclusive evidence exists that higher N deposition results in a higher canopy uptake of N (Bleeker & Draaijers 2002) or whether the N uptake process becomes saturated with increasing N deposition. Since NH_4^+ is taken up at a significantly faster rate than NO_3^- (Bowden *et al* 1998, Boyce *et al* 1996, Garten & Hanson 1990, Potter *et al* 1991, Schulze & Gebauer 1989, Stachurski & Zimka 2000), no such compensation process is observed for NO_3^-, and the higher dry deposition of NO_3^- onto coniferous forests therefore results in higher stand deposition flux, also in regions with lower atmospheric N pollution.

Stand deposition of protons is the resultant of on the one hand supply by dry deposited acids (NO_3^- and SO_4^{2-}) and on the other hand consumption in the canopy by canopy exchange processes, dry deposition of ammonia and bicarbonate (Ulrich 1983). The higher proton input in the coniferous forest indicates both the higher dry deposition capacity and the lower canopy exchange of potassium in coniferous stands. In addition, the higher proton input in the coniferous plots can be due to the canopy exchange of NH_4^+ for H^+ (Neary & Gizyn 1994), as coniferous forest probably retains more NH_4^+ in the canopy.

Percolation

The significant higher stand deposition flux of inorganic N in coniferous forests is reflected in an almost threefold higher percolation flux of NO_3^-. In all stand comparisons included in the meta-analysis, NO_3^- percolation under coniferous forest is higher than under deciduous forest, although the differences are highly variable. Several studies have found a close relationship between the level of N deposition and level of N percolation, at least for those sites where N saturation is evident (Matson *et al* 2002, Macdonald *et al* 2002). Apart from stand deposition fluxes, meteorological conditions explain, to a large extent, the variation in concentrations of major ions in the soil solution (De Vries *et al* 2003). Besides a higher stand deposition flux, the higher NO_3^- percolation flux under coniferous forest may also be related to species-specific N uptake and N retention capacity. For certain broadleaved species, such as ash (*Fraxinus* sp.) and oak (*Quercus* sp.), high NH_4^+ concentrations are a stimulus for NO_3^- uptake (Stadler *et al* 1993), while NO_3^- uptake by roots of coniferous trees is known to be strongly reduced in the presence of NH_4^+ (Marschner *et al* 1991, Rennenberg *et al* 1996). Also Gebauer *et al* (2000) found that young and mature conifer stands retained less than 10% of throughfall N deposition, while broadleaf trees were able to retain 70% and more.

The higher deposition load of SO_4^{2-} in coniferous forest stands was not reflected in a significant higher percolation to groundwater. The ability of many soils to retain significant quantities of SO_4^{2-} by inorganic adsorption mechanisms (Harrison & Johnson 1992, Martinson *et al* 2005) may underlie this conclusion. E.g. adsorption typically increases as the soil pH and the amount of organic matter decrease (Harrison & Johnson 1992), both parameters being influenced by the forest type.

Different mechanisms might be responsible for the higher mean percolation flux of base cations and Al(III) under coniferous stands in the meta-analysis (Meesenburg *et al* 1995, Wesselink *et al* 1995). First, a higher input of base cations through the stand deposition flux and through litterfall can result in a higher output. Secondly, differences in soil acidification due to the different proton input or the different input of base cations could be responsible for percolation differences under the forest types. Soil acidification will induce displacement of cations by protons, thereby reducing the base cation pool and increasing the proportion of Al(III) on the soil-exchange complex (Boxman *et al* 1998, Bredemeier *et al* 1998, Tietema *et al* 1998). A third possibility is that the higher percolation of anions under conifers involves a higher soluble cation fraction, making total cation percolation a function of total anion percolation (Johnson 1992). In forest soils with a base saturation over 25% or a pH-KCl level over 4.5, a strong positive relationship exists between concentrations of Ca^{2+} and strong acid anions in the subsoil (De Vries *et al* 2003), indicating that the percolation of anions in less acidified soils is mainly neutralised by the release of base cations. In forest soils with a base saturation below 25% or a pH-KCl below 4.5, the concentration of Al(III) in the subsoil is strongly related to the concentration of SO_4^{2-} and NO_3^- (De Vries *et al* 2003). Aluminium release is the dominant buffering process in acid forest soils. In a German study, a stronger acidity of the soil solution under coniferous forest than under deciduous forest was only found at the acid soil, but not at the less acidified soil (Rothe *et al* 2002). The high variability in soil conditions and the limited number of stand pairs reporting on percolation fluxes can explain why no univocal and significant differences between forest types were found in the present meta-analysis.

2.1.5 Conclusions

Anthropogenic N additions to ecosystems have been shown to affect a wide range of ecosystem properties and processes, especially when inputs are high and continuous (Matson *et al* 2002). Increased NO_3^- percolation from N saturated ecosystems drives acidification and eutrophication and alters biodiversity in downstream freshwater and marine systems (Rabalais 2002, Schindler 1994). From the comparison of pure coniferous and pure deciduous stands at comparable sites, it can be inferred that forest conversion has a good potential to reduce the N input to forest stands and N percolation and to decrease the loss of base and acid cations to the deeper soil, particularly in areas with high levels of atmospheric N pollution. As has also been shown in the NITREX and EXMAN clean rain experiments, reduction of the N input to forest soils results in a rapid and significant decrease in NO_3^- percolation, and consequently of base and acid cations (Boxman *et al* 1998, Bredemeier *et al* 1998, Tietema *et al* 1998). Therefore, the mitigation of the environmental problems of eutrophication and acidification of forest ecosystems is a valuable additional argument for forest managers to convert coniferous forests.

2.2 Effect of Norway spruce, oak and beech forest types on biogeochemical cycling

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2.2.1 Introduction

The development of human activities has drastically changed the natural aspect of temperate forest ecosystems. Since the 19th century in Belgium, open landscapes originating from ancient management practices were largely planted with easy growing coniferous tree species. Because of its high potential on poor soil and many wood qualities, Norway spruce (*Picea abies*) rapidly became the main production tree species in Wallonia, especially in the Ardenne, where Norway spruce stands are managed as even-aged monocultures. In the eighties, forest decline symptoms and concerns about ecological and environmental impact of Norway spruce monocultures have slowed down its extension in Wallonia, where Norway spruce remains the main production tree actually. Furthermore, Claessens *et al* (2001) have demonstrated that nearly half of the Norway spruce stands in Wallonia are not on adequate sites. These aspects were treated in the Section 1.6. An option to restore a balanced situation on unsuitable areas for Norway spruce stands is to convert them into pure or mixed broadleaved stands, adapted to local conditions (Spiecker *et* *al* 2004). Broadleaved trees and conifers could also be mixed at suitable places. Indeed, Cannell *et al* (1992) have shown that two or more species may use resources differently and be more resistant to stress factors if they coexist in a same stand.

Whereas tree species composition and the choice of site-adapted species appear to be fundamental in new management practices, there are still many gaps in our knowledge of tree-soil interactions. Some effects of tree species are nevertheless well documented. Soil acidification under conifers and under Norway spruce stands in particular has been investigated in many studies (Noirfalise & Vanesse 1975, Bonneau *et al* 1979, Nys & Ranger 1985). However, comparisons of tree species effects under similar environmental conditions are very rare. Without this prerequisite, measured differences between stands can be attributed to former land use or to different pedological conditions rather than to tree species (Hagen-Thorn *et al* 2004). Homogenous conditions (except tree species) are difficult to combine and it could take a long time for trees to influence soil parameters (Menyailo *et al* 2002, Priha & Smolander 1999).

In forest stands, the availability of nutrients for plant growth and maintenance is dependent of the efficiency of the biogeochemical cycle. This nutrients turnover varies between different tree species (Binkley & Giardina 1998). For example, nutrients may be differentially accessed at depth and recycled after litterfall (see Section 2.4) (Washburn & Arthur 2003). Nitrogen (N) is frequently the most limiting nutrient in forests (Prescott 2002). N is mainly located in soils but a major part is in organic forms, not directly available for plant nutrition. N cycling is thus a critical process in forest productivity. Soil microorganisms control the release of inorganic N from decomposing litter.

The main objective of this research was a qualitative and quantitative comparison of the effects of three tree species (Sessile oak, European beech, Norway spruce) on soil fertility and biological soil N transformations. Key environmental parameters such as chemical composition of throughfall, water soil solution and leachates serve as indicators to describe nutrients dynamics under each stand.

2.2.2 Material and Methods

Study site

The experimental plots (Table 2-3) are localised in the public forest 'Hertogenwald' in the Membach locality, south-eastern Belgium. Mean annual temperature and rainfall are 7°C and 1300 mm, respectively. The three neighbouring pure stands (*Fagus sylvatica, Quercus petraea, Picea abies*) are situated on stony-loamy acid brown soil with moder humus, developed on a quartzitic bedrock (Revinian – 'la Venne-Coo' formation) (Laloux *et al* 1996).

Oak and beech stands (even-aged stands) originate from a coppice with standards. The Norway spruce stand is an even-aged secondary plantation, 75 years old. By the end of the 18th century, the three sites were covered by a broadleaved timber forest. Under the oak stand, a dense herbal layer mainly composed from *Molinia cearula* and, for a lesser part, from *Pteridium aquilinum* is present. *Deschampsia flexuosa* and *Vaccinium myrtillus* are also observed at some places. In the Norway spruce stand, the moss layer is developed and is composed of *Polytrichaceae* and *Dicranaceae*. At more open places, the same herbal species as in the oak stand are growing. Nearly no herbal layer is present under the beech stand. Only some *Deschampsia flexuosa* and *Vaccinium myrtillus* have been noted.

Table 2-3: Characteristics of the three forest stands

tree species	slope (m 100m^{-1})	stem number (ha^{-1})	DBH (cm)	stand area (ha)	age (yr)	basal area (m²)	forest type
Oak	0	150	41	2	75	19.6	Luzulo-Quercetum molinietosum
Beech	<5	137	49	1	70-130	28.7	Luzulo-Fagetum
Spruce	<5	325	44	1	75	23.1	Piceetum

DBH = diameter at breast height (130 cm)

Experimental plots

Six experimental plots were installed around six randomly selected trees in each stand. Soil solution was collected continuously in each plot by tension lysimeters (PRENART soil water sampler, Denmark) at 1.5 m from the stem of the selected tree, below the forest floor (about 10 cm). Because their interception surfaces are known and allow potential input-output budget measurements (without root uptake), zero tension lysimeters were used in addition to tension lysimeters for leaching measurements. Potential soil leaching losses were quantified *in situ* using zero tension lysimeters, consisting of a 20 cm soil column in a PVC tube (20 cm diameter) in connection with a funnel which is linked to a collector flask. As zero tension