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PATTERNS OF THROUGHFALL DEPOSITION, NITRATE SEEPAGE, AND SOIL ACIDIFICATION IN CONTRASTING FOREST EDGES

Thesis submitted in fulfillment of the requirements
for the degree of Doctor (PhD) in Applied Biological Sciences:
Land and Forest Management

Dutch translation of the title:
Patronen van doorvaldepositie, nitraatuitspoeling en bodemverzuring in contrasterende bosranden

Illustrations on the cover:
Front: gradual forest edge at Neigembos - Belgium (Sept. 2006)
Back: naturally developed gradual forest edge near the seashore at Haringzelle - Nord-Pas-de-Calais - France (May 2008)

Citation: Wuyts K. 2009. Patterns of throughfall deposition, nitrate seepage, and soil acidification in contrasting forest edges. Ph.D. thesis, Ghent University, Belgium, 202p.

ISBN-number: 978-90-5989-283-5

Contents

Woord vooraf

Het wordt hoog tijd om iets te bekennen wat me al enige tijd van het hart moet: eigenlijk hoort het niet dat mijn naam alleen op de voorzijde van dit doctoraat prijkt, want - ere wie ere toekomt - ik heb het niet alleen geschreven! Ik heb het geluk gehad dat ik de voorbije vier jaren omringd ben geweest door een schare aan collega's, vrienden en familieleden die me met veel raad en daad hebben bijgestaan. Ook zij horen dus eigenlijk op de voorzijde van dit doctoraat vermeld te staan!

Mijn voorliefde voor bosranden ontkiemde toen ik de kans kreeg om mijn thesis te schrijven onder de begeleiding van An en met deze buitenbeentjes van bossen van dichtbij kennis te maken. Dat was eveneens het startschot voor een jarenlange stroom van steun: eender wanneer, waar, waarvoor, An, je stond steeds voor me klaar. Van het uitdenken van de onderzoeksvragen over het selecteren van bestanden, het zeulen van water op terrein, het breken van de hersenen op onverwachte resultaten, het nalezen van mijn schrijfsels van de eerste tot de laatste zin en het inspreken van moed tot het strooiselgraven à la limite! Dikke dikke merci!! En zeker weten: zonder jou was dit doctoraat er nooit geweest!

Kris, je bezieling voor het wetenschappelijk onderzoek werkt aanstekelijk! Zelfs de meest bizarre, wenkbrauwfronsende resultaten worden door jou onthaald met interesse en enthousiasme. Je hebt me door dik en dun gesteund en aangemoedigd. Ik ben je heel dankbaar voor je verweer tegen het dreigende voortijdige einde van dit doctoraat en dat ik de kans heb gekregen om de non-believers hun ongelijk te bewijzen!

Luc, zonder jou zat ik nu nog waterstalen te verzamelen, lysimeters te steken, boomhoogtes te meten, plastieken kerstboompjes te schudden, graduele randen te kappen en in kronen te knippen, in een eeuwige strijd met verstikkende bramen, broeken verslindende prikkeldraad, zelfdesintegrerend materiaal-met-pootjes, het vroeger donker worden dan gepland en muggen, dazen en teken die je fluks naar Deet doen grijpen! Tussen Jabbeke en Netersel hebben we 13553 waterstalen verzameld, memorabele avonturen beleefd en bijgeleerd over Marokko, luchtballonnen, de grenzen van de terreinwagen, de vrijgevigheid van apothekers en de snelheid waarmee laarzen vol regenwater lopen! Je stond altijd vol naarstigheid klaar, hoe lastig, verkleumend en vooral lang het werk ook mocht zijn. Superbedankt voor de bergen werk die je hebt verzet op terrein en in het chemisch labo!

Het Labo voor Bosbouw draagt samenwerking hoog in het vaandel en daar heb ik ook ten volle van kunnen genieten. Mijn speciale dank gaat dan ook uit naar de medeaanhangers van de biogeochemie: Lotte, Sandy en vooral Jeroen, die steeds bereid was zijn geweldige kritische zelve te laten opborrelen! Ook de oud-leden Jan, Leen en Guy wil ik bedanken voor hun inspiratie gebruikt in de uitwerking van dit doctoraat. Greet, bedankt om de onophoudelijk lijkende stroom van water- en bodemstalen de baas te kunnen en op 1-2-3 meetresultaten te voorschijn te kunnen ‘toveren’! Op terrein, in het labo en in en om de windtunnel heb ik zeer gewaardeerde hulp gekregen van onder andere Jeroen Vandenbruwane, Margot, Frederic, Robbie, Ive, An, Lander, Jeroen, Guy, Lotte, Christel, Leen, Evy, Kris Ceunen, Jurgen en mijn papa. Aan iedereen van het Labo, bedankt voor de leuke sfeer gedurende die vier jaren en de aanmoedigende woorden tijdens het afhandelen van de laatste loodjes. Ook Roeland, bedankt voor de enthousiaste aanmoedigingen!

Mijn dank gaat ook uit naar de voorzitter en de leden van de jury alsook de leden van de leescommissie voor hun constructieve opmerkingen op de eerste versie van dit doctoraat, met name de Professoren Marc Verloo, Stefaan De Neve, Wim Cornelis, Quentin Ponette en Kathy Steppe. I am very grateful to Prof. Per Gundersen who enthusiastically agreed on coming all the way from Denmark!

Ik ben professoren Wim Cornelis en Donald Gabriels dankbaar voor het vrij ter beschikking stellen van hun windtunnel en anemometers en de hulp bij het uitdenken en opstarten van onze experimenten. Dank ook aan Jerry Cox, Luc de Cat, Paul Meuleman, Patrick Engels, Danny Maddelein en Paul Vandenabeele van ANB, Wim de Jong van Brabants Landschap, Marc Beque van Stad Brugge, Lieven Kinds, de jachtwachters Mr. De Waele, Mr. Versteynen en Mr. Van der Steen en de privéboseigenaars Mr. Mison, Mr. Vande Vyvere en Mr. Gilles de Pelicht. Want allemaal dulden jullie die afzichtelijke buizen, potjes, darmpjes, trechters en flessen in jullie bossen. Speciale dank aan de bosarbeiders die onder leiding van Jerry Cox de helft van de prachtige graduele bosrand in Neigembos kapten.

Mama en papa, jullie hebben ons opgevoed met open geest en hebben ons alle kansen geboden die het mogelijk maakten ons eigen zelfje te ontwikkelen in eender welke richting we wilden. We zijn altijd welkom en jullie staan steeds met open armen klaar om ons praktisch en emotioneel te steunen en aan te moedigen! Er bestaan geen woorden om jullie daarvoor te bedanken. Papa, bedankt om je verwondering en liefde voor de natuur met me mee te geven. Tom & Bianca, oma & opa, Sarah & Johan, Jan & Lieve en Wesley: een dikke merci voor jullie interesse, enthousiasme en onuitputtelijke stroom aan aanmoedigingen! En een dikke knuffel voor de ‘speciallekes’ Flor en Wisse.

Lander en Margot, jullie delen in mijn brommerigheid en vreugde, tegenslagen en successen, pessimistische en optimistische momenten, leed en liefde. Ik ben jullie oneindig dankbaar voor jullie kleine en grote bijdrages aan dit doctoraat en mijn leven.

Tot slot wil ik iedereen bedanken die is komen supporteren tijdens de verdediging van mijn doctoraat! Dikke merci!

Binnenkort is het weer lente, geniet ervan en laat je verwondering de vrije loop!

List of abbreviations and symbols

Abbreviations

BA	Basal area (m^2 ha^{-1})
BC	Base cations ($Na^+ + K^+ + Ca^{2+} + Mg^{2+}$)
Bc	Base cations minus Na^+
Bp	Stand dominated by *Betula pendula* Roth
CD	Mean crown depth (m)
CE	Canopy exchange (equiv ha^{-1} $period^{-1}$)
CEC	Cation exchange capacity
CL	Critical load (equiv ha^{-1} $period^{-1}$)
DD	Dry deposition (equiv ha^{-1} $period^{-1}$)
DEI	Depth of edge influence (m)
DH	Site 'Dombergheide' near Turnhout
DIN	Dissolved inorganic nitrogen
DOC	Dissolved organic carbon
DON	Dissolved organic nitrogen
DOM	Dissolved organic matter
EF	Edge deposition flux
EGZ	Enhanced gust zone
FED	Forest edge distance (m)
FEE	Forest edge enhancement
GEV	Gradual edge vegetation
H	Mean tree height (m)
H_{dom}	Dominant tree height (m)
IF	Interior deposition flux
IFEE	Integrated forest edge enhancement
LAI	Leaf area index (m^2 m^{-2})
MEI	Magnitude of edge influence
N	Nitrogen
NB	Site 'Neigembos' near Ninove
NH	Site 'Neterselse heide' in the Netherlands
N+S	Potentially acidifying deposition flux (equiv ha^{-1} $period^{-1}$)
OF	Open-field bulk deposition flux (equiv ha^{-1} $period^{-1}$ or kg ha^{-1} $period^{-1}$)
Pn	Stand dominated by *Pinus nigra* ssp. *laricio* Maire or *P. nigra* ssp. *nigra* var. *nigra* Arnold
Ps	Stand dominated by *Pinus sylvestris* L.
Qr	Stand dominated by *Quercus robur* L.
S	Sulphur
SD	Standard deviation
SN	Stand density (ha^{-1})
SOM	Soil organic matter
TF	Throughfall deposition flux (equiv ha^{-1} $period^{-1}$ or kg ha^{-1} $period^{-1}$)
V	Stand volume (m^3 ha^{-1})

Symbols

α	Slope of linear relation
β	Constant of linear relation
d	Displacement height (m)
h	Canopy height (m or cm)
p	Significance of statistical test
σ_u	Standard deviation of wind speed or turbulence
u*	Friction velocity (m s^{-1})
u_y	Wind speed at height y (m) above the surface (m s^{-1})
x	Horizontal distance (m or cm)
z	Height above ground surface (m or cm)
z_0	Roughness length (m)

Chemical compounds

Al	Aluminium
Ca^{2+}	Calcium
Cl^-	Chloride
Fe	Iron
H^+	Proton
K^+	Potassium
Mg^{2+}	Magnesium
Na^+	Sodium
NH_3	Ammonia
NH_4^+	Ammonium
NH_x	$NH_3 + NH_4^+$
N_2	Nitrogen gas
NO	Nitric oxide
NO_2	Nitrogen dioxide
NO_2^-	Nitrite
NO_3^-	Nitrate
NO_y	$NO_2 + NO_3^- + NO_2^-$
N_2O	Nitrous oxide
PO_4^{3-}	Phosphate
SO_2	Sulphur dioxide
SO_4^{2-}	Sulphate
SO_x	$SO_2 + SO_4^{2-}$

Definitions of terms as applied in this thesis

(Forest) edge s.l.	The border between a forest and an area with a differing land cover, in this thesis mainly an open area, such as a meadow, field, or heathland (i.e., an external forest edge) unless stated otherwise
(Forest) edge s.s.	Location of the front of the stem(s) of the first tree (row)
Edge front	First 2 m of the forest edge
Outer edge	First 10-20 m of the forest edge
Edge zone	Zone from forest edge onwards where enhanced deposition is appreciable

1 Introduction

In many landscapes around the world, human activities such as timber harvesting, agricultural expansion, and urbanization cause the extent of habitat fragmentation to increase (Harper et al. 2005; Broadbent et al. 2008). Forest fragmentation, the process of ‘splitting up’ continuous or contiguous forested area into smaller isolated patches of varying size, is actually regarded as the most important treat to forests in the tropics (Gascon et al. 2000; Broadbent et al. 2008). Also in temperate regions of Europe, the USA, South-America, and Asia, forests suffer from increasing land use pressure and fragmentation (Wade et al. 2003; Fig. 1.1). Particularly forests in Europe have already experienced high rates of fragmentation, mainly induced by centuries of human activities (Fig. 1.1). With forest fragmentation, the proportion of the total forested area experiencing influence from edges increases, and hence, forest edges become a dominant feature in the landscape matrix (Riiters et al. 2002; Erisman and Draaijers 2003; De Schrijver et al. 2007a; Echeverria et al. 2008).

1.1 Forest edges as atmospheric deposition hotspots

Of all matrix and ecotonal boundaries, particularly forest edges have been studied intensively, due to their vast influence on both the (created) non-forested open area and the (remaining) forested area. Within these so-called edge effects, we can distinguish primary effects, e.g., on microclimate and fluxes of nutrients (Saunders et al. 1991; Chen et al. 1992; Weathers et al. 2001), and secondary effects or ecosystem responses, e.g., effects on forest structure and biodiversity (Yahner 1988; Thimonier et al. 1992; Lloyd et al. 2000; Harper et al. 2005; Broadbent et al. 2008). Although most research on edge effects focuses on secondary effects, and particularly on biodiversity [for reviews, see Murcia (1995), Harper et al. (2005), and Broadbent et al. (2008)], a considerable number of studies looked into edge effects on fluxes of nutrients through the forest edge. Forest edges are subject to significant edge effects on atmospheric deposition, or as Weathers et al. (2001) stated it: forest edges are hotspots of atmospheric deposition.

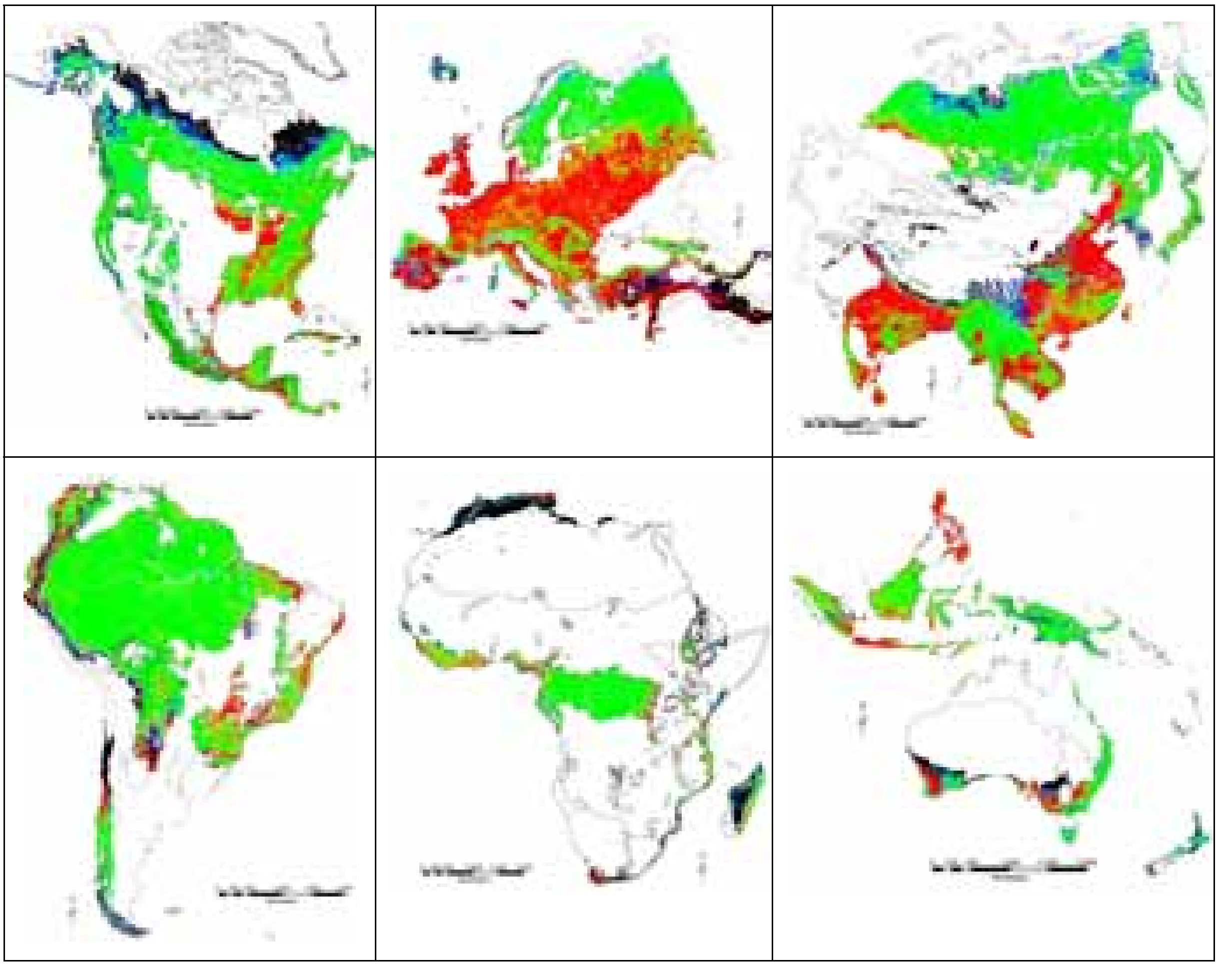

Fig. 1.1: Global forest fragmentation per continent (green: well-connected forest; red: highly fragmented forests, human-induced; blue: highly fragmented forests by natural causes) according to Wade et al. (2003)

In general, forest edges are steep transitions of vegetation height, which drastically disrupt air flow: when approaching an edge, wind flow partially penetrates it and partly lifts over it. As a result, mean wind speed and air turbulence are enhanced at the edge and decline towards the forest interior (Meroney 1970; Chen et al. 1995; Irvine et al. 1997; Morse et al. 2002; Fig. 1.2). As canopy wind speed and turbulence control the dry deposition process of atmospheric particles and gasses (Beckett et al. 2000; Smith et al. 2000), alterations in these factors at the forest edge induce forest edge effects on dry deposition. Next to these aerodynamic processes, edge effects in forest edges can also be attributed to altered dry deposition and/or canopy exchange induced by edge gradients in soil, precipitation, and microclimatic characteristics, as described by Matlack (1994), Chen et al. (1995), Klaassen et al. (1996), Marchand and Houle (2006), Herbst et al. (2007), and Heithecker and Halpern (2007).

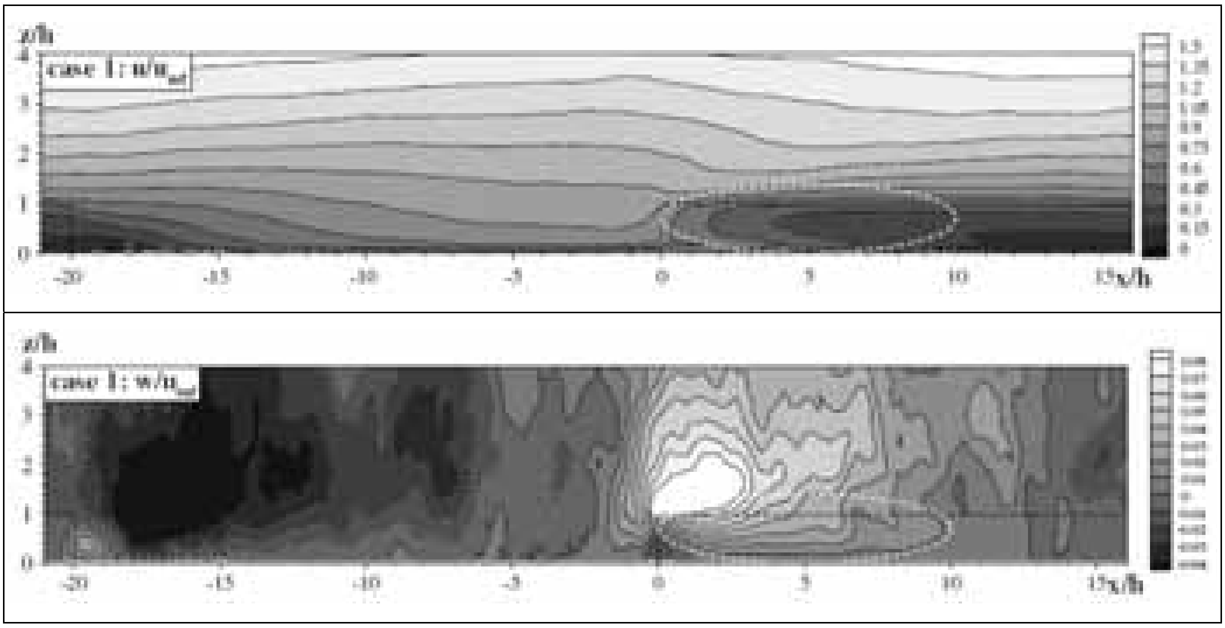

Fig. 1.2: Simulated normalized mean streamwise (u/u_{ref}, top) and vertical (w/u_{ref}, bottom) speed of wind flow directed to the right, over a steep forest edge in function of (i) the distance from the forest edge (x) and (ii) the height above ground level (z), both expressed in multiples of tree height h. For streamwise wind speed, lighter shade corresponds to higher wind speeds, while for vertical wind speed, lighter zones (positive values) refer to upward wind flows and darker zones (negative values) denote downward wind flows. The dashed black line indicates the contour of the forest and the white dashed ellipse shows the adjustment region (Dupont and Brunet 2008a).

The manifestation of edge effects on atmospheric deposition in forests has been abundantly demonstrated, mainly by throughfall deposition measurements [Hasselrot and Grennfelt 1987; Beier and Gundersen 1989; Ivens 1990; Draaijers et al. 1994; Weathers et al. 2001; Spangenberg and Kölling 2004; Devlaeminck et al. 2005; for a review, see De Schrijver et al. (2007a)], but also by a number of air concentration, modelling, and wind tunnel studies (Wiman and Ågren 1985; Wiman and Lannefors 1985; Pahl 2000; Ould-Dada et al. 2002; Dupont and Brunet 2008a). The deposition of N and the potentially acidifying ions NO_3^-, NH_4^+, and SO_4^{2-} is increased at the front of the edge by up to a fourfold compared to the forest interior, and these edge effects decrease exponentially with increasing distance from the edge until the deposition reaches a more or less constant 'interior forest level' at 8 to 108 m from the edge. Fig. 1.3 demonstrates the theoretical throughfall deposition curve from the edge to the forest interior. The size of edge effects can be expressed by the penetration depth of the edge effect, i.e., the depth of edge influence [DEI; Chen et al. (1995); Fig. 1.3] or forest edge distance (FED), and the level of throughfall deposition enhancement at the outer edge, i.e., the magnitude of edge influence [MEI; Harper et al. (2005); Fig. 1.3] or forest edge enhancement

(FEE). The depth and magnitude of edge influence on throughfall deposition depend on (i) meteorological conditions such as wind speed and direction (Draaijers et al. 1988) and herewith related the orientation of the edge (De Schrijver et al. 1998; Draaijers et al. 1994) and (ii) the ion considered (Beier and Gundersen 1989; Draaijers et al. 1994; Spangenberg and Kölling 2004). In addition to these external factors, edge effects in forests are influenced by internal factors such as stand and edge structure, as indicated by wind flow simulations by Pahl (2000) and Dupont and Brunet (2008a). Studies by Wiman and Ågren (1985), Weathers et al. (2001), and Spangenberg and Kölling (2004) give only fragmentary indications of the impact of leaf area index (LAI), edge closing, and forest type (coniferous versus deciduous tree species) on edge patterns of deposition. Draaijers (1993) attempted to assess the influence of several stand and edge structure characteristics (such as LAI, stand density, tree height, crown depth, edge porosity) based on eight coniferous forest stands, but had to deal with confounding factors. For example, Draaijers (1993) found a positive relation between the deposition enhancement in the forest edge zone relative to the interior deposition on the one hand and LAI and stem density on the other. However, different LAI values coincided with different tree species and, thus, different needle morphology, so the effect of LAI was obscured by an effect of needle morphology.

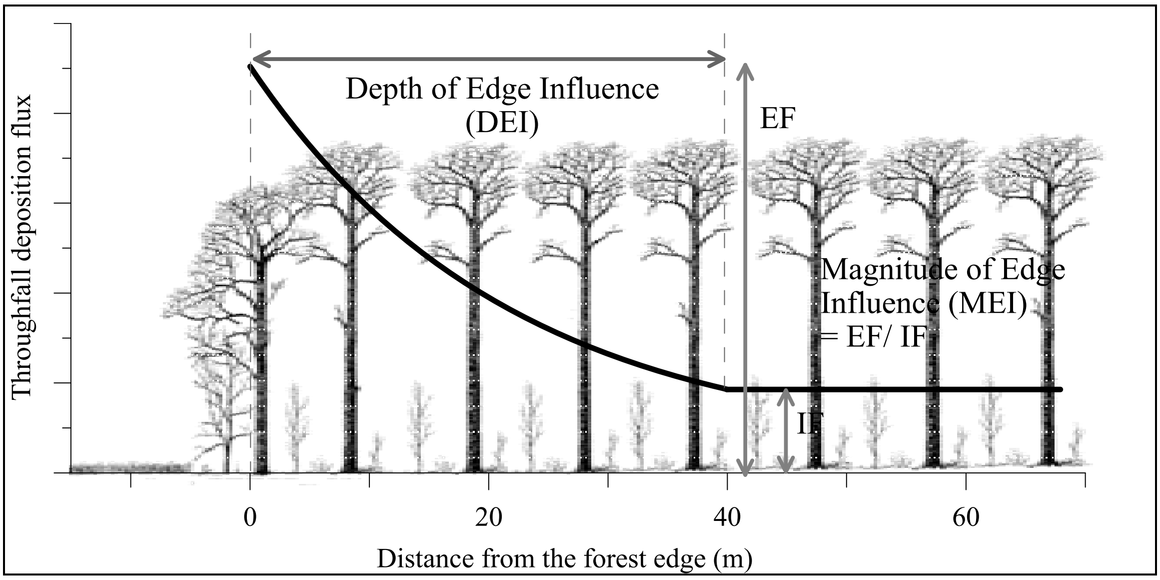

Fig. 1.3: Theoretical throughfall deposition curve along a transect across a forest edge [modified from Draaijers (1993) and Stortelder et al. (2001)] described by the depth of edge influence (DEI) and the magnitude of edge influence (MEI). The latter can be calculated as the ratio of deposition flux at the front of the edge (edge flux, EF) to the deposition flux in the forest interior (interior flux, IF).

1.2 Effects of nitrogen and potentially acidifying deposition in interiors and edges

In the interior of forests, high levels of nitrogen (N) deposition are associated with shifts from N limitation to N saturation, increased levels of nitrate (NO_3^-) seepage to the groundwater, and increased N gas emissions (Aber et al. 1998; Macdonald et al. 2002; de Vries et al. 2003; Kristensen et al. 2004; Vestgarden et al. 2004; Pilegaard et al. 2006; De Schrijver et al. 2008). In pristine temperate forest ecosystems, an almost closed internal cycle between plants, microbes, and the soil organic matter is assumed, with litter production, decomposition, mineralization, immobilization, and plant uptake as the main processes involved (Gundersen et al. 2006; Fig. 1.4).

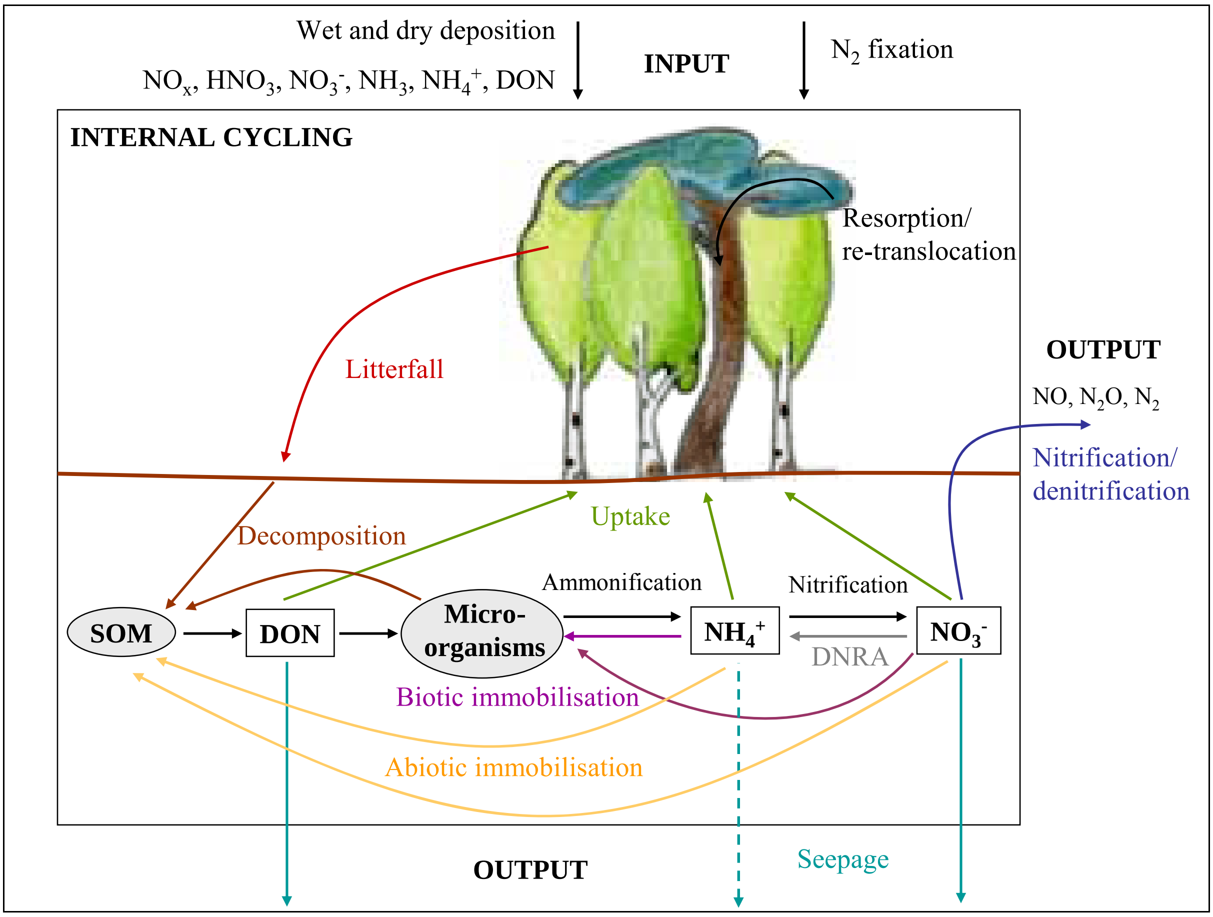

Fig. 1.4: Simplified schematic overview of the cycling of N in a temperate forest ecosystem [modified from Bengtsson and Bergwall (2000), Schulze (2000), Qualls et al. (2002), Davidson et al. (2003), Gundersen et al. (2006), Hagen-Thorn et al. (2006), and De Schrijver (2007)]. SOM stands for soil organic matter, DON for dissolved organic nitrogen, and DNRA for dissimilatory NO_3^- reduction to ammonium (NH_4^+).

This tight N cycling can be disturbed when the sources (or availability) of N from external inputs, like air pollution, and mineralization exceed the N sinks of uptake and immobilization by plants, microbes, and soils (Gundersen et al. 2006). This condition is commonly referred to as N saturation (Aber et al. 1989) and is mostly determined as elevated NO_3^- seepage below the main rooting zone (Gundersen et al. 2006). Gundersen et al. (2006) provided a schematic overview of the potential responses of temperate forest ecosystems to increased N deposition during the development of forests from N limitation via N saturation to N excess (Fig. 1.5). Responses of flora include the loss of characteristic, N-efficient species and the dominance of N-requiring species, which leads to floristic shifts and declines in forest biodiversity (Thimonier et al. 1994; Bobbink et al. 1998; Pitcairn et al. 2002; Gilliam 2006).

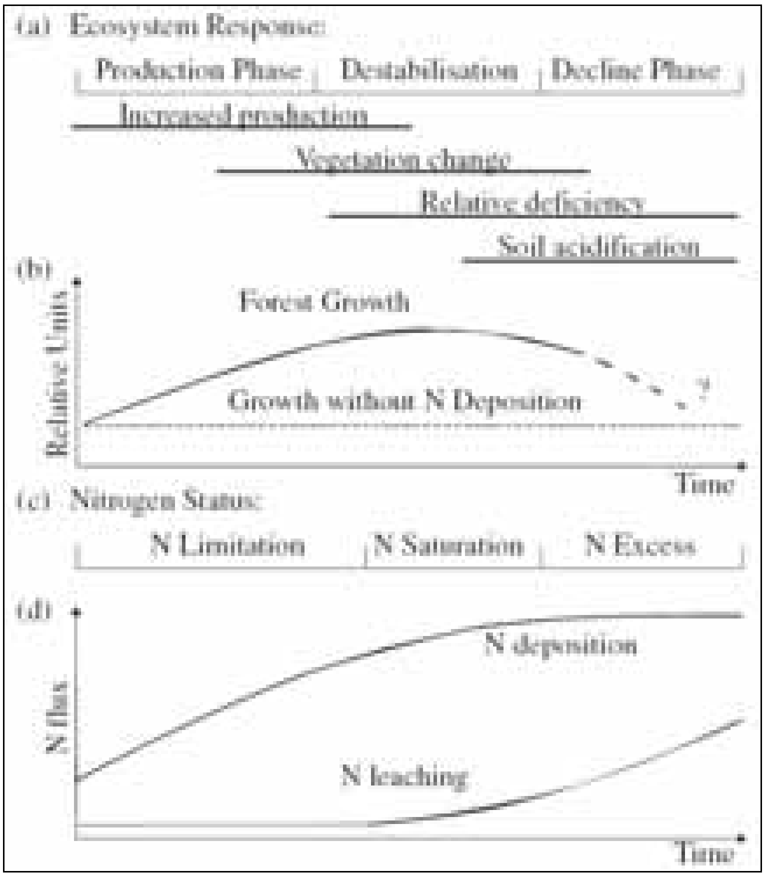

Fig. 1.5: Hypothesized (a) ecosystem responses, (b) relative changes in growth, (c) shifts in N status, and (d) changes in input-output relations in managed temperate forest ecosystems as a result of increased N deposition (Gundersen et al. 2006)

Increased deposition of N and sulphur (N+S) has been identified as one of the major causes of accelerated soil acidification (van Breemen et al. 1982; Bredemeier 1989; De Schrijver et al.

1998; Kreutzer et al. 1998; Bowman et al. 2008). Soil acidification can be defined as the build-up of protons (H^+) in the soil and the associated decrease in acid neutralising capacity, when the proportion of sodium (Na^+), potassium (K^+), calcium (Ca^{2+}), and magnesium (Mg^{2+}) on the clay-humus-exchange complex decreases in favour of H^+, aluminium (Al), and iron (Fe) (van Breemen et al. 1984). Soil acidification is the result of H^+ producing processes (sources) superseding H^+ consuming processes (sinks), among which (van Breemen et al. 1984):

(i) input (via atmospheric deposition) vs export (drainage) of potentially acid substances,
(ii) net assimilation of cations (Na^+, K^+, NH_4^+, Ca^{2+}, Mg^{2+}) vs anions [NO_3^-, $H_2PO_4^-$, sulphate (SO_4^{2-})] by the vegetation,
(iii) net-release of anions vs cations from organic matter,
(iv) oxidation vs reduction reactions,
(v) deprotonation of weak acids vs protonation of anions,
(vi) precipitation vs mineral weathering of cations (Na^+, K^+, Ca^{2+}, Mg^{2+}, Al^{3+}), and
(vii) mineral weathering vs precipitation of anions ($H_2PO_4^-$, SO_4^{2-}).

The acidifying effect of the potentially acidifying ions NO_3^- and SO_4^{2-} depends on the chemical composition of the deposited ions. When deposited as an acid, acidification takes place as a result of the release of H^+ and the seepage of NO_3^- and SO_4^{2-} from the system together with a base cation. Deposition of NH_4^+ can be taken up by plants or microbes (release of 1 H^+), assimilated into organic matter (release of 1 H^+), or oxidized to nitrate (release of 2 H^+). Calcium carbonates, silicates, base cations, Al, and Fe may buffer the acidifying effect of NH_4^+, NO_3^-, and SO_4^{2-} deposition (Binkley and Richter 1987; Ulrich 1991). Within the cation exchange buffer range ($4.2 < pH(H_2O) < 5$), the base cations of the cation exchange complex of minerals and organic matter buffer the incoming H^+, resulting in a reduction of base saturation (the percentage of exchange sites on the complex occupied by the base cations). At the same time, the cation exchange capacity (CEC) is reduced due to the blocking of permanent charges on the exchange complex. At a $pH(H_2O) < 4.2$, Al and, at a $pH(H_2O) < 3.8$, Fe are included in the buffer reactions (Ulrich 1991). Soil acidification causes nutrient imbalances and vitality decreases in vegetation via nutrient leaching from the soil and high Al concentrations that impede plant ion uptake (Schulze 1989). As a result, rare plant species often typical of intermediate pHs decline while acid-tolerant species become dominant (Falkengren-Grerup 1986; Thimonier et al. 1994; Bobbink et al. 1998; Baeten et al. in press).

Due to the higher input of N and S in forest edges (primary effect), even larger effects on abiotic processes and on fauna and flora (secondary effects or ecosystem responses) can be expected at edges than in forest interiors. Research results on the consequences of elevated N and S deposition on NO_3^- seepage and soil acidification in forest edges are, however, ambiguous. Both lower and higher NO_3^- seepage fluxes and concentrations in soil water have been reported at edges in comparison with forest interiors, with peak values occurring at the outer edge to several meters behind the edge (Balsberg-Påhlsson and Bergkvist 1995; Kinniburgh and Trafford 1996; Spangenberg and Kölling 2004; Mellert et al. 2008). In addition, both higher and lower pH values have been detected at the edges of forests in comparison with the interiors (Balsberg-Påhlsson and Bergkvist 1995; De Schrijver et al. 1998; Szibalski and Felix-Henningsen 1999; Honnay et al. 2002). At edges, additional factors such as microclimatic gradients, higher deposition of the so-called base cations K^+, Ca^{2+}, and Mg^{2+} (Spangenberg and Kölling 2004; Devlaeminck et al. 2005), and higher biomass production (McDonald and Urban 2004; Sherich et al. 2007) may affect nutrient cycling via altered soil processes and physiological reactions. Hence, the relationships found in forest interiors between N and S deposition on the one hand and NO_3^- seepage and soil acidification on the other may be altered at forest edges. Microclimatic gradients at forest edges generally encompass higher air and soil temperatures, higher radiation, lower relative air humidity and soil moisture, and higher throughfall volumes, although occasionally opposite patterns are observed (Fig. 1.6). In the northern hemisphere, the largest differences in temperature, humidity, and radiation between edge and interior can be found at southwest facing edges (Chen et al. 1995). In Western Europe, these differences can be even more pronounced due to the exposure of southwest facing edges towards the prevailing wind direction. The penetration depth of the microclimatic edge effects in temperate forests varies greatly (Fig. 1.6), between 3 m (Honnay et al. 2002) and 240 m (Chen et al. 1995) from the edge.

When considering edge effects on biota, and on plant vegetation in particular, difficulties in isolating the effect of increased deposition of N and S at edges arise because of several confounding factors, such as increased light transmittance, higher temperatures, lower soil moisture, higher predation and herbivory, and a higher pressure of external seed fluxes from the open land into the forest edge (Chen et al. 1995; Devlaeminck et al. 2005; López-Barrera et al. 2006). Thimonier et al. (1992), however, found a 20-year change in vegetation towards a more eutrophic level to be the highest at forest edges, and particularly at edges exposed to the prevailing winds, and they attributed this pattern to enhanced atmospheric deposition at these edges. Pitcairn et al. (1998) observed larger changes in vegetation composition at edges next to livestock farms than at edges further away from the farms, i.e., they found a reduction in diversity and an increase in presence and abundance of (woodland) generalist. More recently, by comparing plant species composition of arable land-woodland and non-arable land-woodland edges, Willi et al. (2005) also found ancient woodland ground flora to be substantially altered at arable land-woodland edges due to eutrophication. The changes occurred in terms of greater diversity but also in terms of a higher percentage cover of nutrient demanding species such as the competitive *Urtica dioica* L., hereby out-competing the more stress-tolerant woodland species such as *Carex sylvatica* Huds. and *Primula vulgaris* Huds.

Forest edges are ecotones that amalgamate species from both forest and open land, but may also harbour characteristic 'ecotonal' species (Yahner 1988; Menzel et al. 1999; Lloyd et al. 2000; Magura 2002; Ohwaki et al. 2007). Hence, as forest edges can encompass high species richness, they are particularly vulnerable to negative effects of high N and S deposition. Surprisingly, when the exceedance of critical loads (i.e., the maximum exposure to one or more pollutants below which significant harmful effects on specified sensitive elements of the environment do not occur according to present knowledge; Nilsson and Grennfelt 1988) is calculated for forests, increased deposition at forest edges is often neglected, hereby underestimating the negative effects of deposition. Lövblad et al. (1995) roughly modelled that the enhancement of deposition in forest edges is of minor importance for the exceedance of critical loads in forests of southern Sweden due to the low proportion of forest edges in the entire forested area and the simultaneous enhancement of base cation deposition. However, for more fragmented landscapes like in Flanders, De Schrijver et al. (2007a) provided a rough insight into the error on current calculations of the exceedance of critical loads of N for the protection of biodiversity, and found the exceedance of critical loads to be underestimated by up to 30 % when the calculations did not account for edge effects.

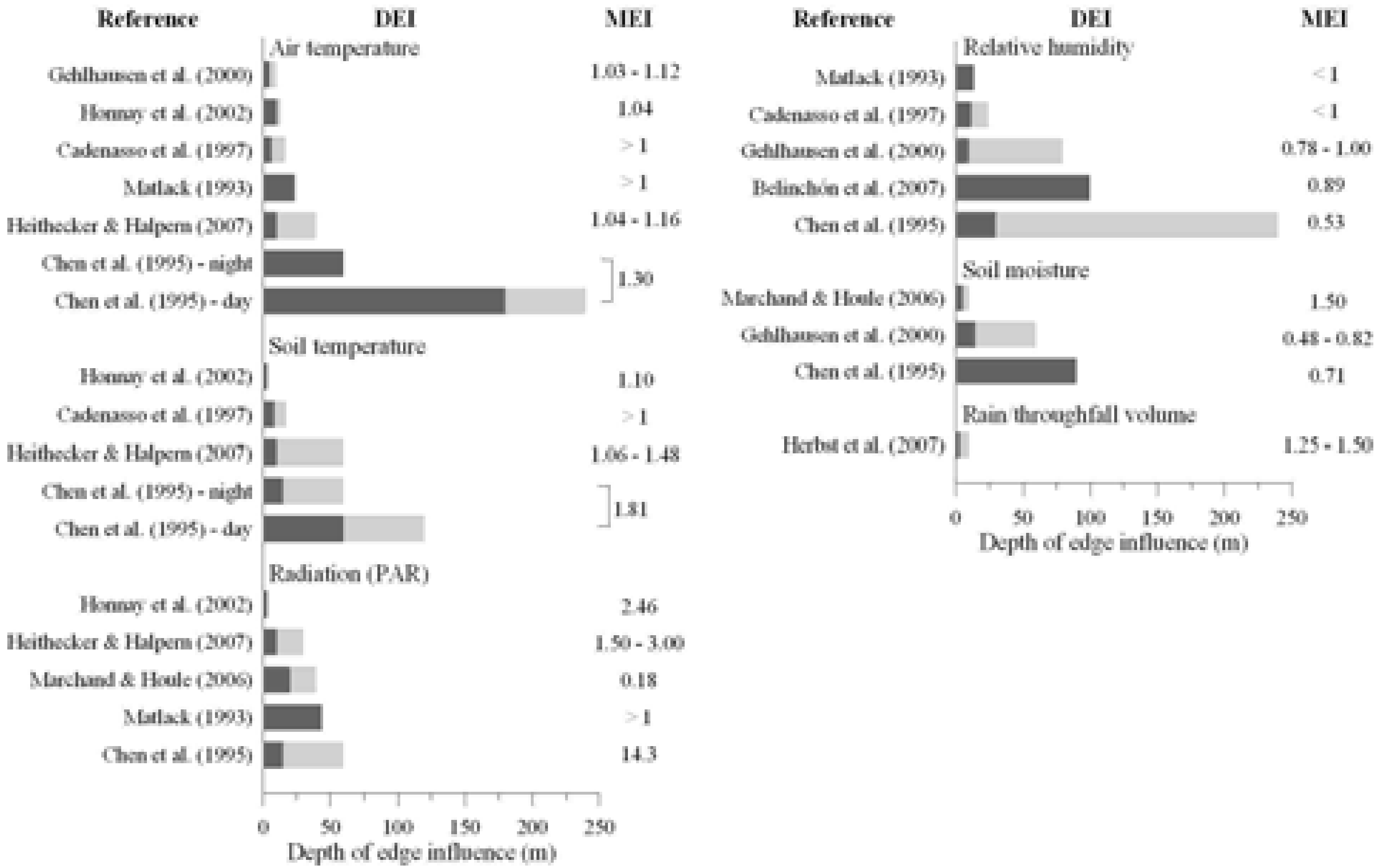

Fig. 1.6: Edge effects on microclimatic characteristics in temperate deciduous and coniferous forests in Europe and the USA: indicated are the penetration depth (depth of edge influence, DEI, m) and the level of increase (MEI > 1) or decrease (MEI < 1) at the edge front (magnitude of edge influence, MEI). In some studies, the DEI and the MEI were provided as value ranges (indicated by the lighter grey bars). (PAR: photosynthetically active radiation)

1.3 Forest fragmentation and atmospheric deposition in Flanders

To which extent are forests in Flanders affected by forest fragmentation and atmospheric deposition? Flanders is subject to a high level of forest fragmentation (Hermy et al. 2004). As a result, forest patches in Flanders have a high ratio of forest edge to forest interior (Fig. 1.7), and, when considering an edge depth of 50 m, almost 60 % of the total forested area in Flanders consists of external forest edges bordering a non-forested area (De Schrijver et al. 2007a). This implies that 60 % of the forested area in Flanders is exposed to enhanced deposition of N and potentially acidifying deposition. The deposition of the pollutants SO_x (SO_2 and SO_4^{2-}), NO_y (NO_2, NO_2^-, and NO_3^-), and NH_x (NH_3 and NH_4^+) which the forest edges receive additionally due to edge effects depends on the atmospheric deposition load of N and S.

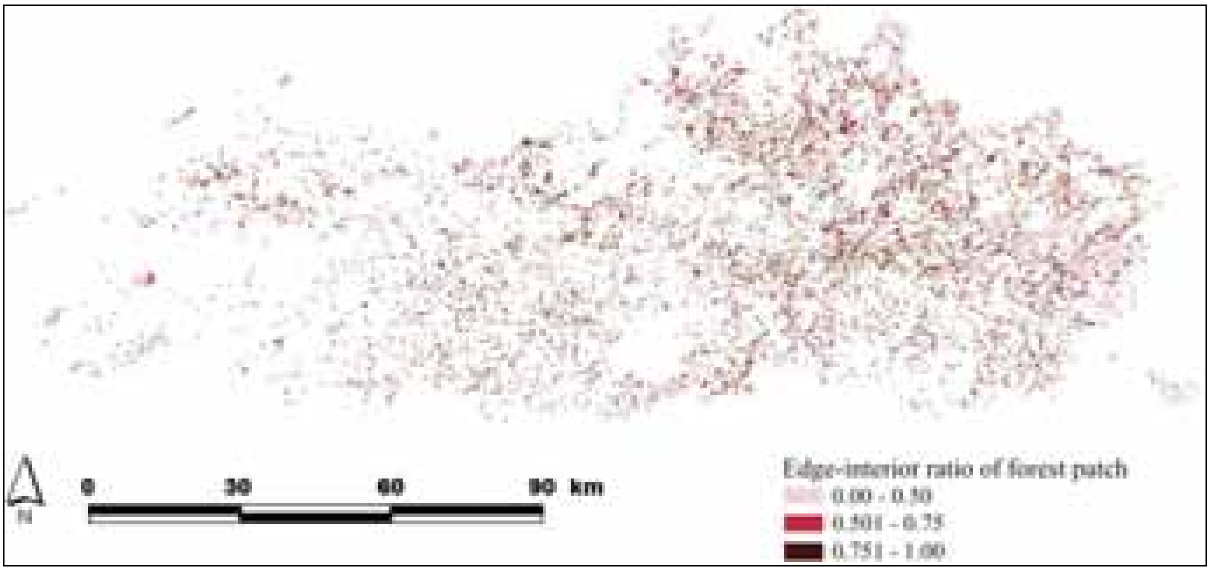

Fig. 1.7: Map of forested areas in Flanders classified according to their ratio of edge area to interior area: the darker the forest, the higher the ratio of edge area to interior area (De Schrijver et al. 2007a).

Flanders suffers levels of N and S deposition that are among the highest in Europe (ICP Forests 2007). By means of the atmospheric dispersion and deposition model OPS (van Jaarsveld 2004), the mean N deposition in 2006 was calculated at 37 kg N ha^{-1} y^{-1} and the mean potentially acidifying deposition in 2004 was 3925 equivalents ha^{-1} y^{-1} (of which 40 % was NH_x, 33 % NO_y, and 27 % SO_x; Van Avermaet et al. 2006; Overloop et al. 2007). Deposition fluxes vary greatly within Flanders: N deposition ranged from 20 to 78 kg N ha^{-1} y^{-1} (in 2006), while the potentially acidifying deposition varied from 2011 to 11900 equiv ha^{-1}

y^{-1} (in 2004) (Van Avermaet et al. 2006; Overloop et al. 2007). The highest deposition levels occur in the vicinity of large cities and industrial areas (SO_x), along the main motorways (NO_y), and in regions with intensive livestock breeding (NH_x) such as the south-eastern part of West Flanders and the northern part of the Campine region in Antwerp and Limburg (Van Avermaet et al. 2006). These high deposition loads result in exceedance of the critical loads (CL) in large parts of forested Flanders: in 2004, 66 % of the forested area was exposed to deposition higher than the $CL_{acidification}$ for protection of roots, and in 2006, the $CL_{eutrophication}$ for protection of biodiversity was exceeded with, on average, 23 kg N ha^{-1} y^{-1} in all forest ecosystems (Van Avermaet et al. 2006; Overloop et al. 2007). The highest CL exceedances are found in the poor sandy soil region of Flanders, where the buffering capacity for acidifying deposition is low (Van Ranst et al. 2002; De Schrijver et al. 2006) and coniferous forests with higher deposition levels and lower CL values for N prevail (Langouche et al. 2002; De Schrijver et al. 2004, 2008). It must be kept in mind, however, that these deposition fluxes and levels of CL exceedance are calculated without taking into account forest edge effects, so these values underestimate the actual deposition fluxes and exceedances of CL.

1.4 The role of forest edge management in the mitigation of edge effects

The abovementioned figures on the prevalence of edge area in Flemish forests, together with the high N and S deposition load in the region of Flanders, point to the importance and need for mitigating measures, i.e., measures that reduce the effects of a high level of atmospheric pollution in the forest edge. Harper et al. (2005) suggest that the magnitude and depth of edge influence are determined directly by the contrast in structure and composition between adjacent communities, and that this contrast is affected by climate and by edge and stand characteristics. So, in the same way as forest type conversion acts as a mitigating measure for N and S deposition in forest interiors (Augusto et al. 2002; De Schrijver et al. 2007b, 2008), a well-considered layout and management of forest edges may be able to reduce edge effects on potentially acidifying and N deposition and their associated consequences. Forest edge management practices for temperate and tropical forests are primarily put forward because of biodiversity reasons, i.e., to create high quality edges that provide a favourable habitat for edge species (e.g., Temple and Flaspohler 1998; Magura 2002; Wermelinger et al. 2007) and to protect interior habitat from deeply penetrating edge effects (Murcia 1995; Didham and Lawton 1999; Gascon et al. 2000; Cadenasso and Pickett 2001). Only Weathers et al. (2001) provided suggestions for edge management for the purpose of reducing edge effects on atmospheric deposition. As they had found that edge effects penetrate deeper into the forest at open edges (without shrubs, saplings, or other understory vegetation) than at closed edges (characterized by 'a wall' of dense vegetation), they concluded that edges should be kept as dense as possible. However, in the study of Weathers et al. (2001), a higher level of inorganic N and S throughfall deposition was observed at the closed edge than at the open edge as a result of the extra deposition on the shrubs and ground vegetation in the closed, unthinned edges.

1.5 Aims and schematic overview of the thesis

The general aim of this thesis was to gain insight into the impact of forest type and edge structure on the magnitude and depth of edge effects on (throughfall) deposition, and into the patterns of ecosystem responses such as nitrate seepage and soil acidification; this in order to formulate recommendations for the design and management of forest edges for mitigating edge effects. We focused on forest ecosystems on sandy soils, which are vulnerable to nitrate seepage and soil acidification (Van Ranst et al. 2002; Rothwell et al. 2008). Moreover, most parts of the sandy soil region in Flanders are characterized by intensive livestock breeding, which results in high NH_3 emissions.

More specifically, the aims of this thesis were:

(i) to assess the influence of forest type on edge patterns of atmospheric deposition, and throughfall deposition of N and acidifying deposition in particular;

(ii) to assess the influence of edge structure on edge patterns of atmospheric deposition, and throughfall deposition of N and acidifying deposition in particular; and

(iii) to assess and quantify the general patterns of important ecosystem responses as soil nutrient seepage and soil acidification from the edge to the interior, in forests on sandy soils.

These aims are addressed in the next six chapters, assembled in three parts (Fig. 1.8). In the first part of the thesis, which encompasses the second and third chapter, **the effect of forest type on edge patterns of throughfall deposition** is looked into. The second chapter focusses on the seasonal variation of the effect of forest type, the differences in edge effects between N and S deposition on the one hand and the deposition of the base cations on the other hand, and the role of dry deposition and canopy exchange in the edge patterns of throughfall deposition.

Moreover, we present the calculation of an alternative measure for edge effects that integrates both the magnitude and depth of edge influence. In the third chapter, it is investigated whether the effect of forest type, as observed in the second chapter, is a coincidental observation or can be generalised to other forests on sandy soils in different regions of Flanders.

The second part of the thesis deals with **the impact of edge structure on patterns of throughfall deposition** by means of a wind tunnel study (chapter four) and two field studies (chapters five and six). Under the controlled circumstances of the wind tunnel, a model forest was configured to study combinations of stem density (or LAI), crown depth, and edge type (i.e., gradual versus steep edge transitions). Based on measurements of wind speed, turbulence, and dry deposition of a chloride aerosol, the effect of edge structure on aerodynamically generated edge effects on dry deposition was explored. In chapters five and six, our findings from the wind tunnel study are tested in the field: chapter five focuses on the effect of edge type as adjacent steep and gradual forest edges are compared for their edge patterns of chloride, N, and S throughfall deposition, while in chapter six, the influence of other edge structural characteristics, and LAI in particular, is considered.

In the last part of the thesis (chapter 7), **the general spatial patterns of soil nutrient seepage of NO_3^- and of soil acidification in forest edges** are assessed, and the effect of forest type on these patterns is discussed. A schematic overview of the thesis' outline is given in Fig. 1.8.

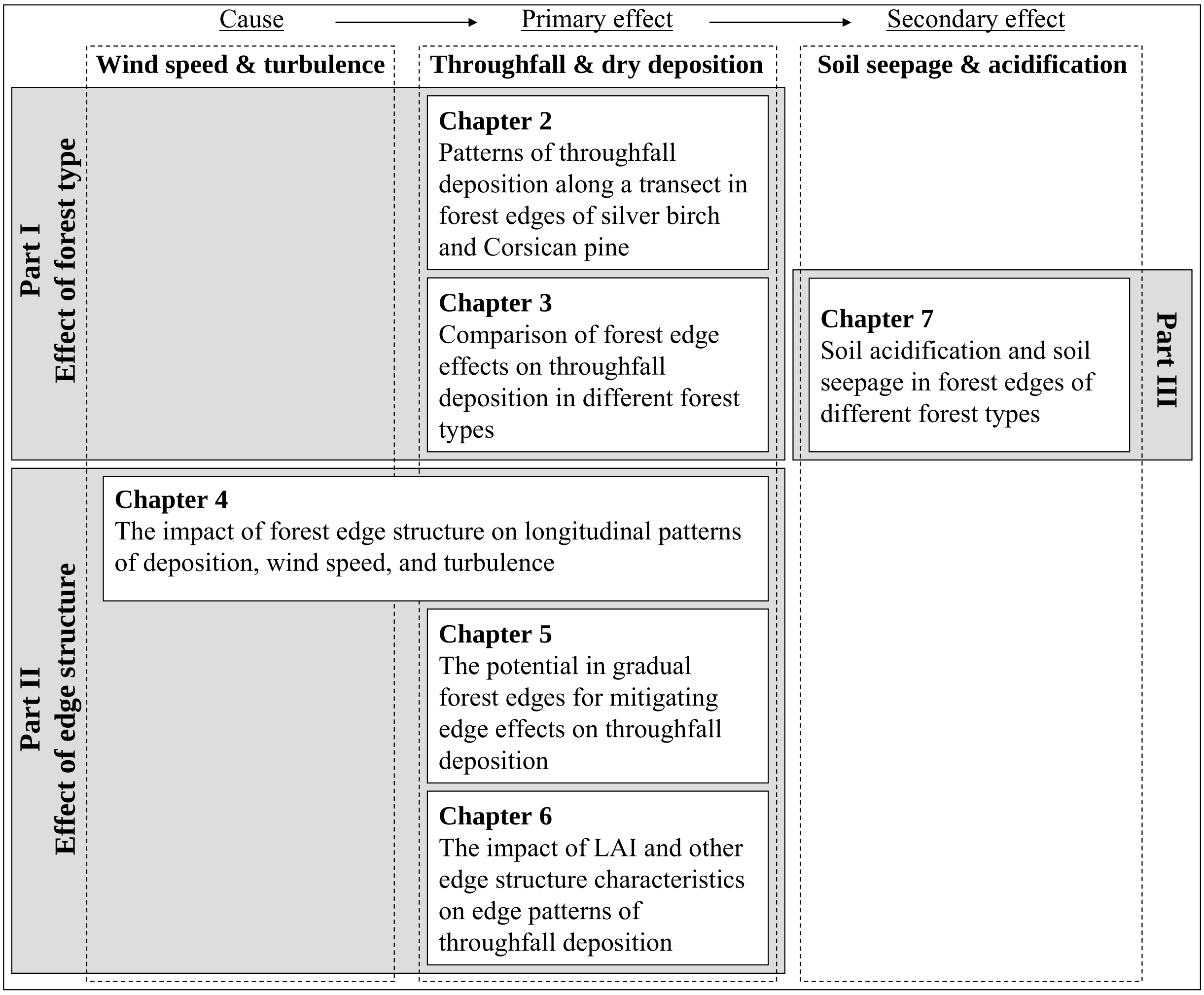

Fig. 1.8: Outline of the thesis

2 Patterns of throughfall deposition along a transect in forest edges of silver birch and Corsican pine

After: Wuyts, K., De Schrijver, A., Staelens, J., Gielis, M., Geudens, G., Verheyen, K., 2008. Patterns of throughfall deposition along a transect in forest edges of silver birch and Corsican pine. Canadian Journal of Forest Research 38, 449-461.

2.1 Abstract

In two adjacent forest stands in Flanders, one dominated by *Pinus nigra* ssp. *laricio* and another dominated by *Betula pendula*, throughfall deposition was monitored along a transect perpendicular to the forest edge exposed to the prevailing wind direction. Throughfall deposition of Na^+, K^+, Ca^{2+}, Mg^{2+}, NH_4^+, NO_3^-, Cl^- and SO_4^{2-} was examined on forest edge patterns expressed in the depth of influence of the edge effect (forest edge distance) and the level of enhancement at the edge (forest edge enhancement). In addition, an integrated forest edge enhancement factor was computed which incorporates these two parameters. Our results show that the edge effects on throughfall deposition of Na^+, Cl^-, the sum of base cations, the sum of potentially acidifying ions and the sum of inorganic nitrogen (NH_4^+ + NO_3^-) are more pronounced in the pine stand. The edge zone of the pine stand receives as a result of the edge effect 9.4 times more extra potentially acidifying ions and 12.7 times more extra inorganic nitrogen than the birch stand. We conclude that an appropriate design or conversion of the edge structure, from high-density Corsican pine plantations into lower density deciduous forests, can reduce the input of N and potentially acidifying pollutants in the forest edge.

2.2 Introduction

Forest edges function as hotspots for atmospheric deposition (Weathers et al. 2001). They are effective at filtering and concentrating airborne pollutants, causing an up to eightfold increase of throughfall deposition compared with the forest interior (e.g., Beier and Gundersen 1989). This edge effect decreases exponentially with increasing distance to the forest edge, until deposition reaches a more or less constant 'interior forest' level at 15 to 108 m into the forest (Hasselrot and Grennfelt 1987; Beier and Gundersen 1989; Draaijers 1993; Spangenberg and Kölling 2004; Devlaeminck et al. 2005; De Schrijver et al. 2007a). In highly fragmented landscapes, most forest patches are characterized by a high edge to interior ratio.

The width of the edge zone with elevated deposition and the level of deposition enhancement both depend on tree species, edge structure, edge aspect, and the element considered (Draaijers 1993; Weathers et al. 2001; Spangenberg and Kölling 2004; De Schrijver et al. 2007a). Most of the research concerning forest edge effects on atmospheric deposition is focused primarily on coniferous forests, while enhanced deposition in forest edge zones has been quantified in only few deciduous forests, such as two mixed deciduous stands in the eastern United States (Weathers et al. 2001) and *Fagus sylvatica* stands in Western and Northern Europe (Balsberg-Påhlsson and Bergkvist 1995; Spangenberg and Kölling 2004; Devlaeminck et al. 2005). Comparison of forest edge effect in coniferous and deciduous forest types is, except for the studies by Spangenberg and Kölling (2004) and Balsberg-Påhlsson and Bergkvist (1995), lacking, although this topic is of significant importance. Management and lay-out of forest edges in order to protect forest ecosystems from further acidification and eutrophication is of crucial importance, particularly in countries with high deposition load. Furthermore, forest edge effects on throughfall deposition are often studied at a coarse spatial resolution, which restricts the correct demarcation of the forest edge effect.

The study's hypothesis is that forest edge effects on atmospheric deposition are less pronounced and exhibit a higher seasonal variation in lower density forests of deciduous trees than in higher density forests of coniferous trees. The aims of this study were therefore (i) to compare the edge effects on throughfall deposition between two highly differing forest edge types, one dominated by Corsican pine (*Pinus nigra* ssp. *laricio* Maire) and one by silver birch (*Betula pendula* Roth), and (ii) to assess the seasonal variation of these edge effects. For this purpose, an integrated forest edge enhancement factor was proposed as a measure for the forest edge effect on throughfall deposition flux that covers the depth of influence as well as the enhancement at the front of the edge.

2.3 Materials and methods

2.3.1 Site description

The experimental site is located at the edge of the state forest 'Domeinbos Ravels' (51°23' N 5°03' E), in the northeast of Flanders (Belgium) (Fig. 2.1). The forest of 810 ha is dominated by homogeneous plantations of Corsican pine, homogeneous plantations of Scots pine (*P. sylvestris* L.) and stands in conversion to silver birch and common oak (*Quercus robur* L.). At the periphery edge of the forest complex, a forest stand dominated by 40-year old Corsican pines and a neighboring stand dominated by 25-year old silver birches were selected, both exposed to the prevailing westerly to southwesterly winds (Fig. 2.1). Table 2.1 lists the stand characteristics of the two forest stands in 2003. The crowns of front trees of the edge were more developed than the crowns of forest interior trees. However, the forest edges were open as branches did not reach the ground. Both edges border undisturbed pasture, in an abrupt interface as a result of absence of any forest edge vegetation.

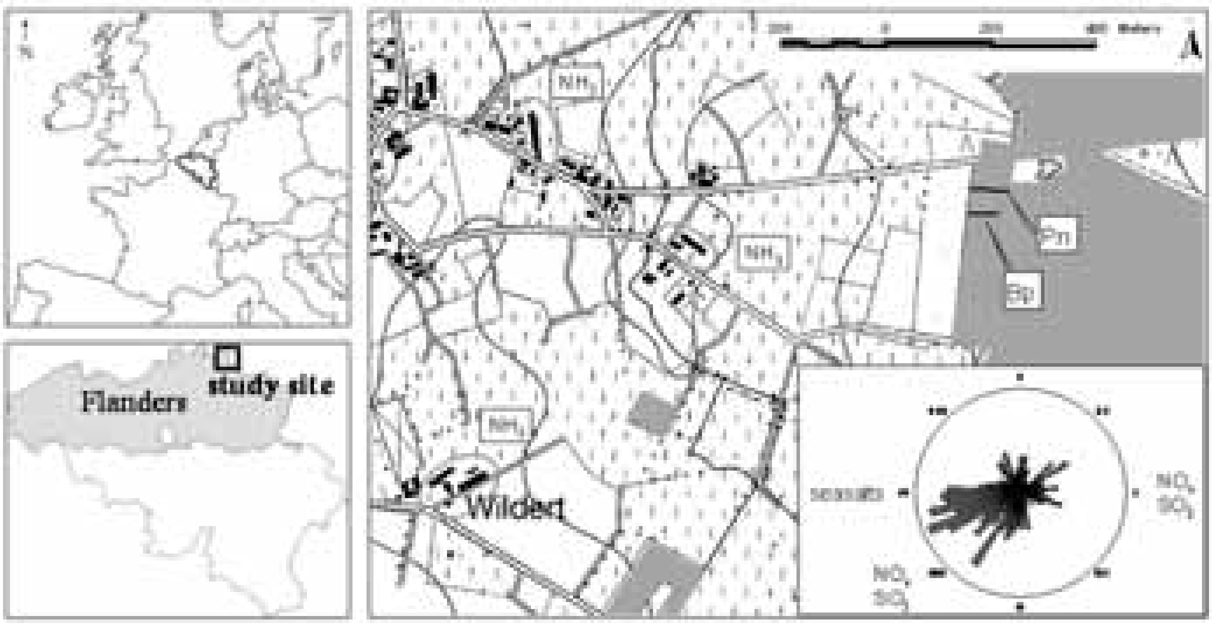

Fig. 2.1: Map of Europe and Belgium, showing the location of the study site and location of the transects with indication of the nearest low point sources of NH_3 and wind rose for the monitoring period 15 July 2003 - 30 July 2004 (KMI) with direction of important NO_x and SO_2 sources. Pn and Bp show the transect in the Corsican pine stand and in the silver birch stand, respectively

Table 2.1: Tree species composition, stand density SN (ha^{-1}), basal area BA ($m^2\ ha^{-1}$), stem volume V ($m^3\ ha^{-1}$), and mean tree height H (m) of the silver birch and Corsican pine stand in 2003.

	Silver birch				Corsican pine			
Tree species	SN	BA	V	H	SN	BA	V	H
Betula pendula	2049	17.12	83.95	11.6	174	0.59	0.65	9.5
Pinus nigra ssp.	49	2.42	19.94	13.1	739	47.96	414.23	17.5
Alnus glutinosa	230	0.85	1.71	7.1	333	3.93	15.62	11.0
Prunus serotina	49	0.26	1.06	8.0	101	0.67	3.46	10.5
Larix sp.	131	1.70	12.35	5.5				
Pinus sylvestris					58	2.40	18.65	16.8
Picea abies					29	1.04	8.14	15.7
Fagus sylvatica					14	0.42	3.24	17.0
Total	2508	22.3	119.0		1448	37.0	464.0	

Values for pH (KCl) and C:N ratio are given for the ectorganic layer and the soil depth ranges 0 – 0.05 m and 0.05 – 0.10 m in Table 2.2. The stands are situated on poor sandy soils, classified as Haplic Podzols (FAO-ISRIC-ISSS 1998). These soils are particularly sensitive to acidification because of low base saturation (5 - 10 % in the upper 100 cm of the mineral soil, Neirynck et al. 2002) and small amounts of weatherable silicate minerals such as feldspar (Van Ranst et al. 2002; De Schrijver et al. 2006). The region is characterized by intensive livestock breeding, which results in high atmospheric concentrations of reduced nitrogen (NH_x). Important sources of SO_2 and oxidized nitrogen (NO_y) are situated 50 km southwest (petrochemical industry at Antwerp harbor) and 140 km east (the German Ruhr area) of the study site; the North Sea is 115 km to the west. Management history was reconstructed using historical maps. The site was converted from heather to a Corsican and Scots pine plantation around 1920. After 1960, pines were harvested and the site was replanted with Corsican pine. A 25 m wide strip of the pine stand was ravaged by wind throw in approximately 1976 and replanted with silver birch.

Table 2.2: Soil characteristics of the silver birch and Corsican pine stand in 2003

Soil depth	pH(KCl)		C/N ratio	
	Birch	Pine	Birch	Pine
ectorganic horizon	3.1	3.0	20	23
0 - 0.05 m	3.4	3.2		
0.05 - 0.10 m	3.8	3.8		

During the study period, mean, minimum, and maximum temperatures were 10.7, -9.6, and 34.6°C respectively, mean wind speed was 2.7 m s^{-1} (2.2 and 3.3 m s^{-1} in the growing and dormant season, respectively) and mean relative humidity was 84 % according to daily data obtained from the Royal Meteorological Institute of Belgium (KMI) at the nearest station (Retie, at 15 km).

2.3.2 Experimental setup

In both stands, one transect was established perpendicular to the forest edge and parallel with the prevailing wind directions (Fig. 2.1). Along these transects, two parallel rows of throughfall collectors were placed from the forest edge up to 54 m from the edge in the silver birch stand and 64 m in the Corsican pine stand. Nearby the forest edge, collectors were placed every 2 to 3 m while further than 8 m from the edge, collectors were spaced over 5 to 6 m. In total, 16 and 18 distance plots of two throughfall collectors each, were set up in the birch and the pine stand, respectively. Throughfall collectors partially or fully located underneath the canopy of other species were excluded from calculation and regression if throughfall deposition was clearly influenced by the presence of the other species (i.e., at 4, 13 and 28 m from the edge in the birch transect and at 16, 41 and 51 m in the pine transect). Bulk deposition was collected in a recent clear-cutting, adjacent to the forest, at 80 m from the end of the transects, using three bulk collectors positioned above the seedlings and at least 50 m from the nearest interior forest edge. The throughfall collectors consisted of white polyethylene funnels (diameter 143 mm) mounted upon 2-liter white polyethylene bottles. As for the open-field bulk deposition collector, an opaque tube connected the polyethylene funnel with the bottle. The bottles were partially buried in the ground to keep samples cool and protected from direct sunlight. A nylon mesh was placed in the neck of the funnels to avoid contamination by large particles. Stemflow was not considered in this study, since its contribution to the total throughfall + stemflow flux to the forest floor is negligible for Corsican pine (Neirynck et al. 2004). For deciduous species, this contribution is generally less than 10 % (Chang and Matzner 2000). Furthermore, the birches' canopies did not have large, heavy branches.

Throughfall and bulk deposition were monitored from 15 July 2003 until 30 July 2004. Samples were collected and measured in fortnightly to monthly intervals. As it had hardly rained in July and August 2003, the first sampling event took place after five weeks. On every sampling event, the volume of the water samples was measured in the field and the bottles, funnels, and mesh were replaced by those rinsed with distilled water. The throughfall samples of two collectors placed at the same position from the edge were pooled volume-weighted to one sample for chemical analysis, as well as the samples of the three open-field bulk deposition collectors. A 350 ml subsample was taken for chemical analysis. Samples in collectors with bird droppings on the funnel were not collected for analysis, but their volume was measured.

Leaf area index (LAI) was measured in August 2004 using the LAI-2000 Plant Canopy Analyzer (LI-COR) every four meters, as a mean value from four measuring points located on the corners of a 3 by 2 m rectangle. All measurements were performed with a lens cap covering the half side of the lens oriented to the forest edge. The LAI-2000 device underestimates LAI as a result of the non-random distribution of the foliar elements, i.e., between-shoots and within-shoot clumping (Chen et al. 1997; Jonckheere et al. 2005). No correction was made for the input of nonleafy materials (branches, dead leaves, tree trunks, lichen and moss) in the optical measurement, as these elements contribute to the throughfall deposition also. The LAI predictions were corrected with the within-shoot and between-shoots clumping factor. For deciduous forests, the within-shoot clumping factor equals 1 (Chen et al. 1997). The between-shoots clumping factor was determined for deciduous forests by Gower et al. (1999) for *Acer saccharum* (0.95) and *Quercus* (0.88) and by Kucharik et al. (1999) for *Populus tremuloides* (0.65). For the birch stand, the LAI estimations were corrected via division by the between-shoots clumping factor which was estimated as the mean of these factors, i.e., 0.83. For the Corsican pine stand, correction factors determined for Scots pine by Jonckheere et al. (2005) were applied: the LAI predictions were multiplied with 2.00 and divided by 0.84. As we are only intereseted in the patterns of LAI across the edges, LAI values were expressed relative to the lowest LAI value in the birch stand.

2.3.3 Sample handling and analysis

Samples were kept cool during transport and stored in darkness at maximum 5°C. A part of each sample was analyzed for pH (pH ORION electrode), the other part was filtered through a 0.45 µm nylon filter (GELMAN) and analyzed for Cl^-, NO_3^-, SO_4^{2-}, and PO_4^{3-} by ion chromatography (ICS-90, DIONEX; column IonPac AS14A-5µm and guard column IonPac AG14A-5µm), for Na^+, K^+, Ca^{2+}, and Mg^{2+} by flame atomic absorption spectrophotometry

(SpectrAA-220, VARIAN), and colorimetric for NH_4^+ (Cary 50 Probe, VARIAN). H^+ concentrations were calculated from pH values.

2.3.4 Sample quality control

The quality of the chemical analyses was checked by including method blanks and repeated measurements of internal and certified reference samples (CRM 409, Quevauviller et al. 1993). Determination limits of the analytical methods were 0.099 ppm for NH_4^+, 0.09 for NO_3^-, 0.05 for SO_4^{2-}, 0.06 for Cl^-, 0.057 for Na^+, 0.04 for K^+, 0.09 for Ca^{2+}, and 0.008 for Mg^{2+}. Two samples were below the detection limit for NO_3^- and one sample was below detection limit for NH_4^+. The coefficient of variation of repeated measurements of CRM 409 samples, conducted throughout the study period, was smaller than 5 % for all ions except for H^+ (5.3 %), Mg^{2+} (5.3 %), and K^+ (8.8 %) and the recovery rate was higher than 95 % for all ions except H^+ (90 %) and K^+ (79 %).

Each sample was checked on its Na^+:Cl^- ratio, ion balance, and PO_4^{3-} concentration. Samples with an elevated PO_4^{3-} concentration could have been contaminated by bird droppings (Erisman et al. 2003) and their K^+, NH_4^+, and H^+ throughfall flux were omitted and estimated by averaging the throughfall flux originating from the two nearest distance plots. Concentrations in solitary samples showing a deviating Na^+:Cl^- ratio were also recalculated. An entire set of samples with deviating Na^+:Cl^- ratio, i.e., for the first sampling event of 21 August 2003, were excluded from calculations of deposition because no phenological event (e.g., leaf senescence, bud burst) could have been the basis for these deviant concentrations. No error in sample analyzing seemed to have occurred, so we hypothesize that the deviation of the Na^+:Cl^- ratio was the result of dust deposited onto the funnels, because of the prolonged period of drought, and/or the result of the long residence time of the collectors and its samples in the field.

2.3.5 Element input

Total free field bulk and throughfall volume (mm) were derived from the volume captured by the collector and the collector's surface area. Throughfall and bulk deposition (equiv ha^{-1}) were calculated by multiplying total throughfall or free field bulk volume by element concentration in that volume. Subsequently, the deposition (in equiv ha^{-1}) of the base cations

(K^+, Ca^{2+}, and Mg^{2+}), the potentially acidifying ions (NH_4^+, NO_3^-, and SO_4^{2-}), and the inorganic nitrogen delivering ions (NH_4^+ and NO_3^-) were calculated and will be abbreviated as the Bc, N+S, and N deposition, respectively. The fortnightly to monthly collected throughfall and bulk deposition data were summed for the growing season (21 August 2003 until 3 November 2003 and 9 April 2004 until 30 July 2004), for the dormant season (3 November 2003 until 9 April 2004), and for the whole study period.

2.3.6 Data analysis

In order to determine the Depth of Edge Influence (DEI), i.e., the width of the edge zone with enhanced throughfall volume and throughfall deposition flux, the method designed by Beier and Gundersen (1989) was used. Throughfall deposition was transformed to natural logarithms and plotted against distance to the forest edge. In this plot, two distance groups were delineated if feasible, separated by the DEI (Fig. 2.2). The first distance group clusters the throughfall deposition data in the forest edge zone (until DEI) and displays a linear relation with slope $\alpha < 0$, indicating an exponential decrease of throughfall deposition as a function of distance to the forest edge. For distances greater than the DEI, a linear relationship with slope α close to 0 is shown.

If a forest edge zone could be defined, the Magnitude of Edge Influence (MEI), i.e., the level by which the fluxes at the beginning of the edge are elevated, was calculated by dividing the flux at the beginning of the edge by the flux in the forest interior. The interior flux in the pine transect was computed as the mean flux of the measuring points from 31 to 64 m, as the zone with enhanced flux ended and the zone with interior flux started at 31 m from the edge or less. In the birch transect, the edge effect ended at 23 m or less (except for Na^+ and Cl^- in the dormant season and for K^+): the interior flux was calculated as the mean flux from 23 to 54 m. The edge flux was chosen to be calculated as the mean flux from the first two measuring points at the forest edge, in order to account for fine-scale spatial variability in throughfall deposition (Staelens et al. 2006) and because of the occurrence of the throughfall deposition peak at 0 m or at 2 m from the forest edge.

To obtain an overall quantification of the forest edge effect upon deposition flux that accounts for the MEI as well as for the DEI, the Integrated Forest Edge Enhancement factor (IFEE) was calculated (Fig. 2.2). Area A represents the extra throughfall deposition flux that occurs in the edge zone, 1 m wide and as deep as the DEI, as a result of the forest edge effect. Area B gives the throughfall deposition flux a forest edge of 50 m deep would receive in absence of a forest edge effect. Dividing the sum of area A and area B by area B, yields the IFEE factor, which enables comparison of the forest edge effect between the two forest types. The area A between the graph of the best linear fit to the ln-transformed deposition data of the first distance group and the axis y = flux at the end of the edge zone (i.e., the beginning of the interior zone), was determined (Fig. 2.2). By means of an integral of the exponential curve $y = \exp(\alpha x + \beta)$ (with y representing the throughfall deposition flux and x distance to the forest edge), evaluated over the interval 0 m until edge distance, the area between the curve and the x-axis was calculated. Subsequently, the area between the x-axis and the axis y = flux at the end of the edge zone was calculated over the interval [0 m - DEI] and was subtracted from the definite integral (see Fig. 2.2) to obtain area A. This IFEE factor was modified from the WEINTE-factor by Draaijers (1993).

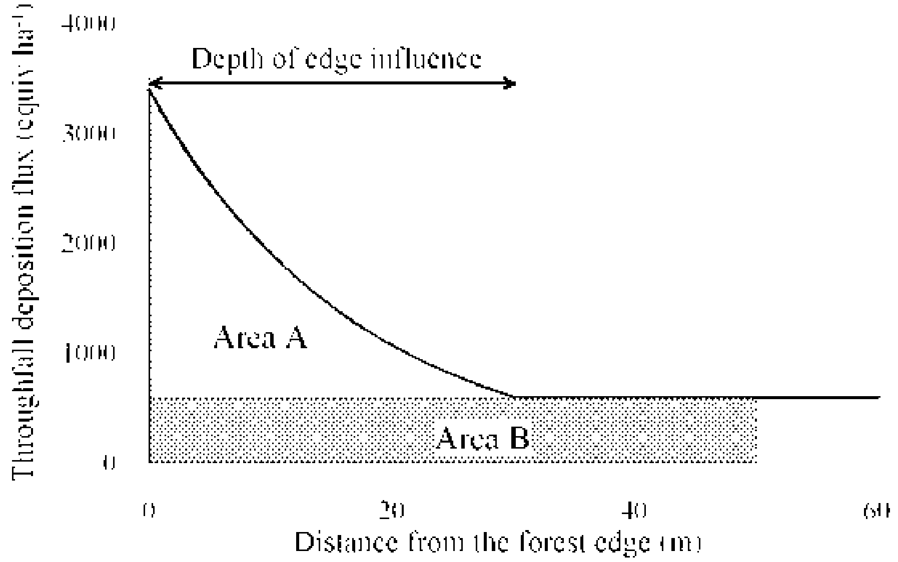

Fig. 2.2: Example curve of the forest edge effect on throughfall deposition flux, displaying the parameters depth of edge influence (DEI), area A, and area B, and herewith the calculation of the integrated forest edge enhancement IFEE.

The edge effect of the two stands was compared using paired t-tests on the IFEE values of Na^+ throughfall deposition, Bc, N+S, and N throughfall deposition obtained for every sampling occasion (the differences were significantly normally distributed). The tests were also performed on IFEE values of sampling occasions in the growing season only and in the dormant season only.

The relation between the IFEE values of Na^+ throughfall deposition and wind speed and wind direction was examined with the Pearson correlation coefficient. The direction weighted summed wind speed was calculated for every sampling event as:

$$\sum_{i=m}^{n}(v_i \times d_i)$$

where i is the number of the day, m the number of the first day of the sampling event, n the number of the last day of the sampling event, v_i the daily mean wind speed (in m s^{-1}), and d_i the daily mean wind direction, ranging from 0 (for winds perpendicular to the forest edge, coming from the east) to 1 (for winds perpendicular to the forest edge, coming from the west).

2.4 Results

Table 2.3 presents total free field bulk water volume (mm) and deposition (equiv ha^{-1}) and mean total throughfall water volume (mm) and deposition (equiv ha^{-1}) in both stands during the growing and the dormant season.

2.4.1 Edge effects

In both stands, throughfall water volume was not subject to a clear edge effect (Fig. 2.3 and Table 2.4). Only during the dormant season a small but significant edge effect was found (Table 2.5) for the first two meters of the Corsican pine and the silver birch edges.

The LAI increased in both stands in the first 9 m from the forest edge (Fig. 2.3). The LAI value at the front trees in the Corsican pine stand was, on average, 10 % higher than deeper in the stand. After 40 m, the LAI in the pine stand rose again because of the occurrence of silver birch and common alder (*Alnus glutinosa*), which also caused slight increases in throughfall deposition at the end of the transect. In the birch stand, the LAI value at the front of the edge was, on average, 20 % higher than inside the stand. A local increase of LAI can be seen in the birch stand at a distance of 25 m until 37 m from the edge and at the end of the transect as a result of the presence of two larch (*Larix* sp.) standards. It should be noted that, for correct interpretation of the correlation between LAI and throughfall deposition, measurements with the LAI 2000 Plant Canopy Analyzer are probably not optimal. These measurements yield mean values of the surrounding area of each collector, while the canopy cover just above the collectors should be known: the LAI 2000 does not measure what is really needed. The exposed canopy coverage over the collectors may be probably estimated in a more correct way using vertically taken photographs of the canopy with a restricted angle of view.

Table 2.3: Open-field bulk volume (mm) and ion deposition (equiv ha^{-1}) adjacent to the forest stand and mean throughfall water (mm) and ion deposition (equiv ha^{-1}) at the forest edge (0-2 m; edge) and the forest interior (23-54 m in the birch stand and 31-64 m in the pine stand; int.) of the silver birch and Corsican pine stand

	Open-field		Silver birch				Corsican pine			
	Grow. season	Dorm. season	Growing season		Dormant season		Growing season		Dormant season	
			Edge	Int.	Edge	Int.	Edge	Int.	Edge	Int.
Volume	380	363	210	224	259	243	180	208	207	215
Na^+	193	323	324	193	1259	437	1264	274	2477	517
K^+	25	14	254	304	143	91	374	251	203	202
Ca^{2+}	80	79	253	238	208	151	354	171	341	134
Mg^{2+}	59	74	192	156	319	147	407	152	609	152
H^+	17	18	4	2	4	4	1	2	1	2
NH_4^+	348	299	417	424	638	407	1749	841	1708	888
Cl^-	244	376	494	327	1575	613	1489	447	3052	671
NO_3^-	192	121	211	196	165	115	764	347	363	195
SO_4^{2-}	287	241	432	360	690	414	1187	543	1576	774
Bc	164	167	699	698	670	389	1135	574	1153	488
N+S	827	661	1060	980	1493	936	3700	1731	3647	1857
N	540	420	628	620	803	522	2513	1188	2071	1083

Throughfall deposition of Mg^{2+} and particularly Na^+ and Cl^- were subject to the largest and deepest edge effect in both forest stands (Fig. 2.3), which resulted in the largest values for the DEI, the MEI, and the IFEE factors (Table 2.4). On the contrary, NO_3^-, NH_4^+, and SO_4^{2-} displayed the smallest and shortest edge effects. K^+ throughfall deposition showed no edge effect in the birch stand and a small edge effect in the pine stand (IFEE = 1.11) (Fig. 2.3 and Table 2.4). The H^+ flux did not show consistent patterns with distance, with exception of a small and short edge effect during the growing season in the silver birch stand. In the birch edge, the linear fit to the logarithm of throughfall flux was only significant ($p < 0.05$) for Na^+, Cl^-, and Mg^{2+}, while in the pine edge the fits were significant for all ions.

Table 2.4: Depth of edge influence (DEI, m) and magnitude of edge influence (MEI) of the edge effect on throughfall volume and throughfall deposition in the silver birch and Corsican pine stand, slope α of the best linear fit to the logarithm of throughfall deposition in the first distance group and resulting area A (see Fig. 2.2), and integrated forest edge enhancement (IFEE). Bc stands for K^++Ca^{2+}+Mg^{2+}, N+S for NH_4^++NO_3^-+SO_4^{2-} and N for NH_4^++NO_3^-. (* $p < 0.05$; -: no edge effect detected and therefore no IFEE determined)

	Silver birch					Corsican pine				
	DEI	MEI	α	A	IFEE	DEI	MEI	α	A	IFEE
Volume	-	1.00				-	0.92			
Na^+	23	2.51	-0.041*	9375	1.31	31	4.73	-0.058*	31503	2.13
K^+	-	1.00				20	1.13	-0.024*	2200	1.11
Ca^{2+}	6	1.19	-0.065	187	1.03	20	1.84	-0.053*	3726	1.31
Mg^{2+}	18	1.68	-0.033*	1780	1.13	26	3.34	-0.064*	7471	1.81
H^+	2	1.32	-0.450	5.9	1.02	-	0.51			
NH_4^+	8	1.27	-0.032	935	1.02	12	2.00	-0.066*	11100	1.13
Cl^-	23	2.20	-0.035*	11311	1.25	31	4.06	-0.055*	38899	2.01
NO_3^-	8	1.21	-0.021	231	1.01	12	2.10	-0.075*	3901	1.15
SO_4^{2-}	8	1.45	-0.046	1352	1.03	12	2.03	-0.068*	9039	1.13
Bc	8	1.26	-0.036	1339	1.03	20	1.92	-0.051*	12484	1.30
N+S	8	1.33	-0.037	2559	1.03	12	2.05	-0.068*	23993	1.13
N	8	1.25	-0.029	1181	1.02	12	2.02	-0.068*	14996	1.13

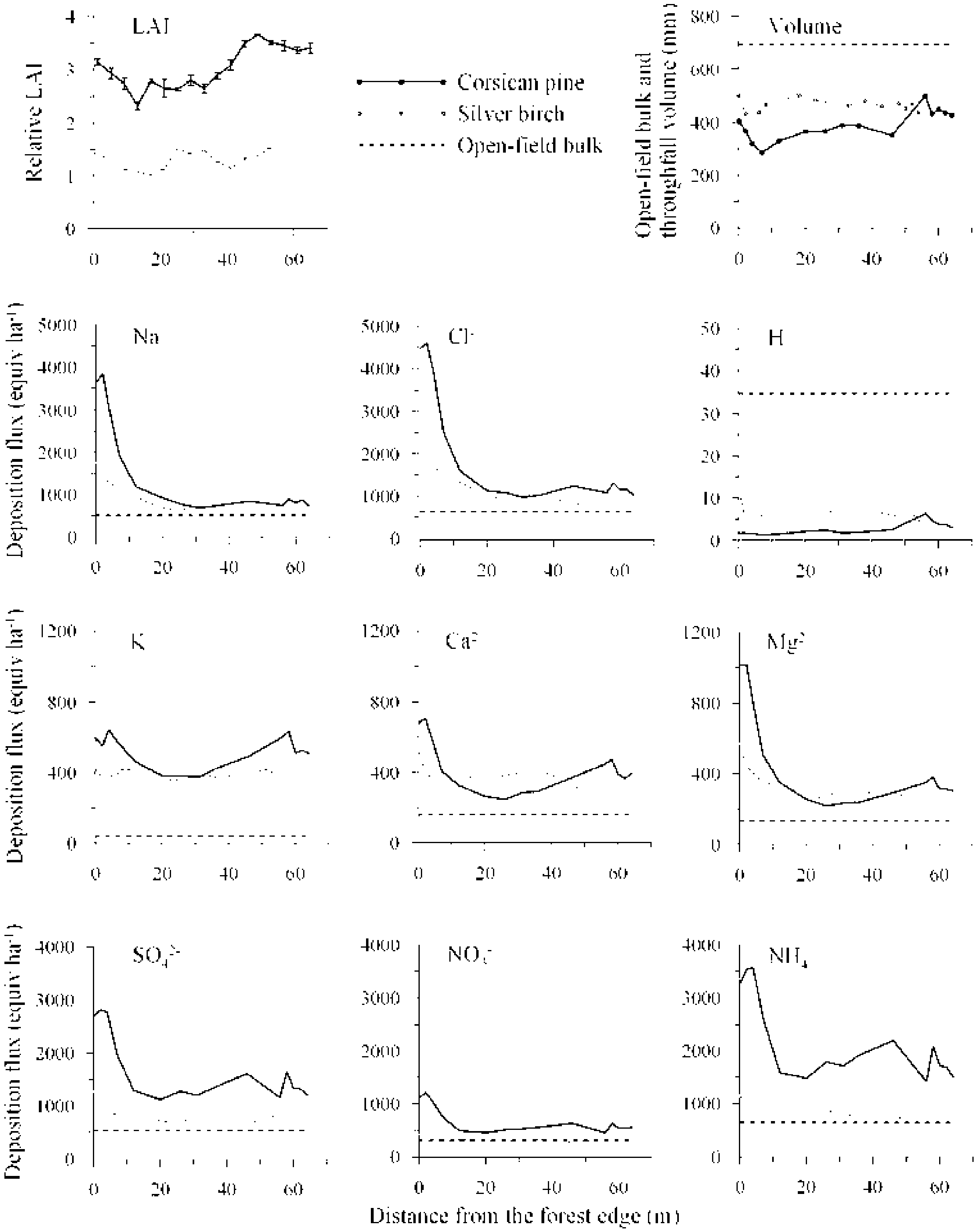

Fig. 2.3: Patterns of LAI, precipitation and throughfall volume (mm), and open- field bulk and throughfall deposition (equiv ha^{-1}) during the entire monitoring period in the silver birch and the Corsican pine stand

2.4.2 Forest structure effect

The transition of pasture to forest resulted in more pronounced edge effects on the throughfall deposition in the Corsican pine stand than in the silver birch stand. The surplus of throughfall deposition (area A) that the pine stand received in the forest edge zone and the IFEE factor in the pine edge were higher for all ions except H^+ (Table 2.4). The IFEE factor in the pine stand was 1.6 (Na^+), 1.3 (Bc), 1.1 (N+S), and 1.1 (N) times as high as in the birch stand and area A was 9.3 (Bc) and 9.4 (N+S) times higher in the pine edge zone than in the birch edge zone. The ratio of area A for N+S to Bc equaled 1.9 in the edge zone of both forest types. For the throughfall flux of N, area A was 12.7 times higher in the pine stand than in the birch stand. The summed throughfall deposition of N+S and Bc in the edge zone of the pine stand was 3.7 and 3.0 times higher, respectively, than in the birch edge zone; for the N throughfall deposition this ratio amounted to 4.0. In a forest edge of 50 m, N+S throughfall flux was twice as high for pine and N throughfall flux was 2.2 times higher in the pine stand, while the Bc flux was the same for both stands.

At the level of individual sampling occasions, the IFEE factors for the throughfall deposition of Na^+ and Bc were higher in the pine than in the birch stand, in exception of two consecutive sampling events during spring for Bc (Fig. 2.4a). The IFEE factors for Na^+ and Bc in the pine stand were 1.43 ± 0.29 (mean ± standard deviation) and 1.19 ± 0.16 times higher, respectively, than in the birch stand. Similarly, the IFEE factors of individual samplings of N+S and N were generally higher in the pine stand (1.13 ± 0.12 and 1.07 ± 0.15, respectively), but to a lesser extent than for Na^+ and Bc. For Na^+, Bc and N+S, the IFEE factors differed significantly ($p < 0.001$) between the two forest types, while this was not the case for N ($p = 0.189$).

The higher values for the IFEE factor and area A in the pine stand were a result of both a larger DEI and larger MEI (Table 2.4). Overall, the MEI values were, on average, twice as high as in the birch stand. The throughfall fluxes of Na^+ and Cl^- were subject to an edge effect that reached about 10 m deeper in the pine stand than in the birch stand. In addition, the enhancement of the throughfall deposition at the pine edge was almost twice as high as in the birch stand. For the throughfall deposition of NH_4^+, NO_3^-, SO_4^{2-}, and consequently the sum of these potentially acidifying ions, the forest edge distances differed only approximately 4 m, so that the edge effect for N+S differed between both stands primarily in forest edge enhancement. In the birch stand, no edge effect was detected for throughfall deposition of K^+, while for Ca^{2+} the edge enhancement was small and only apparent over a short distance. In the pine stand however, the edge effect on K^+ and Ca^{2+} reached 20 m and the enhancement for Ca^{2+} was higher, but the edge enhancement of K^+ was still small. The edge effect by which throughfall deposition of Mg^{2+} was influenced, penetrated deeper in the pine stand than in the birch stand and the throughfall enhancement was twice as high as in the birch edge. The edge effect on the total Bc flux was largest in the pine stand and the difference between the two stands was even more pronounced than for the N+S flux. The edge effect on the throughfall deposition of H^+ was absent in the pine stand and negligibly short in the birch stand.

2.4.3 Effect of season

The IFEE factor fluctuated throughout the year, with the highest variation for Na^+ and Bc (Fig. 2.4a), and only small variations for N+S and N (Fig. 2.4b). In the birch transect, the highest IFEE factors occurred in winter and spring while in autumn, IFEE values beneath 1.0 were measured, accompanied with high IFEE values for N. In the pine transect, the variation in IFEE seemed less related to season. During the growing season, IFEE values differed significantly ($p < 0.01$) between the two stands for Na^+, Bc, and N+S, while in the dormant season only the IFEE values of Na^+ were significantly different ($p = 0.029$) between the stands. During both seasons, the IFEE values of N were not significantly ($p > 0.1$) different between the stands. For both transects, the IFEE factor for Na^+ throughfall deposition of individual sampling periods was significantly ($p < 0.01$) correlated ($r = 0.667$ for pine , $r = 0.680$ for birch) with the direction weighted summed wind speed in the time period of a given sampling event.

While in the Corsican pine stand forest edge effects were clearly present during the growing and the dormant season, throughfall deposition in the birch stand was scarcely influenced by the presence of the edge during the growing season (Table 2.5). In both stands, the DEI for Na^+ and Cl^- throughfall deposition was larger in the dormant than in the growing season, and particularly in the birch stand. The increase of the MEI and the IFEE factor during the dormant season was also more pronounced in the birch stand. H^+ in throughfall deposition did not show an edge effect in any time period in the Corsican pine stand. The difference in DEI of H^+ throughfall deposition between growing and dormant season was negligible in the birch stand.

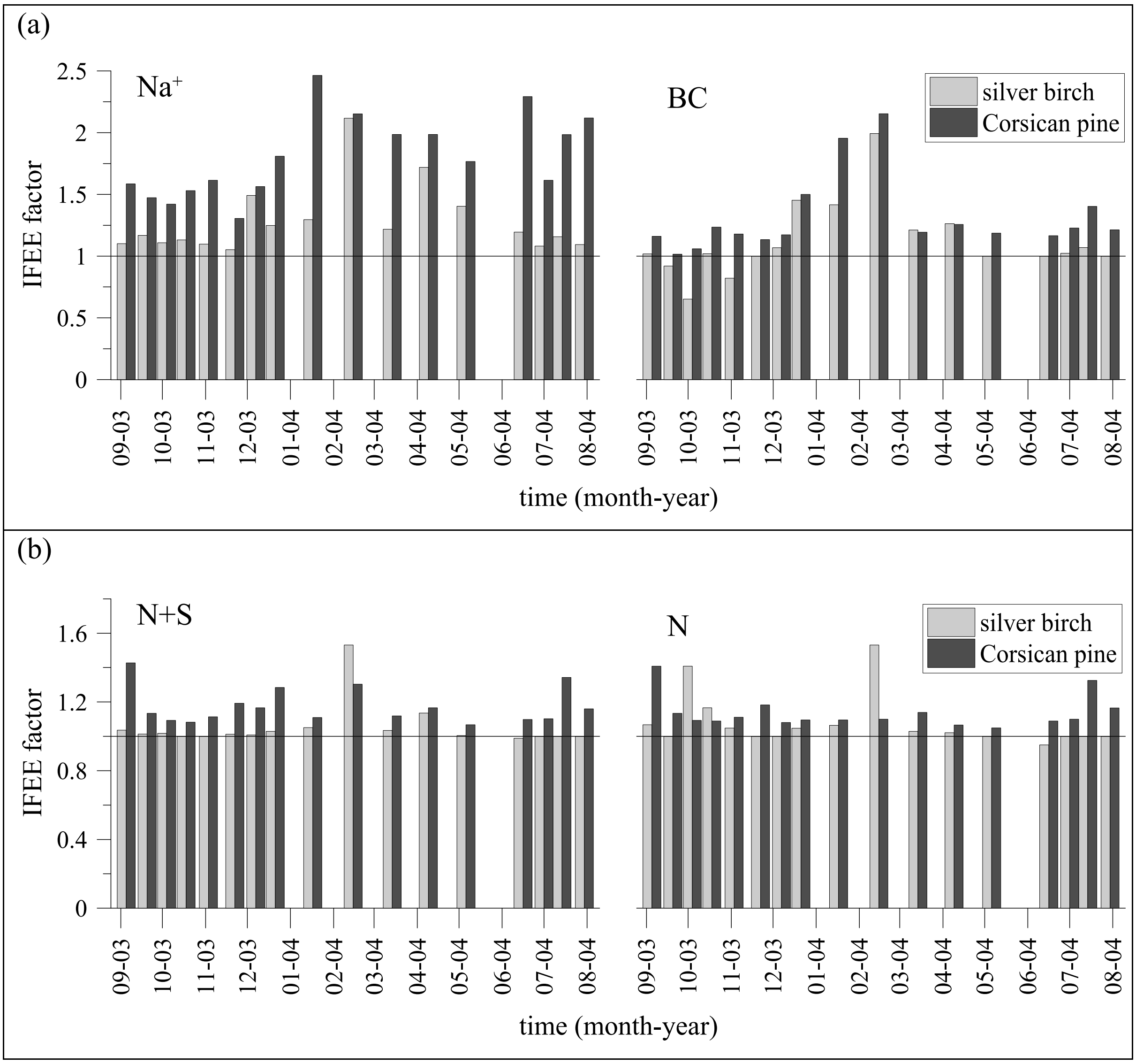

Fig. 2.4: IFEE factor for throughfall deposition of Na+ (a, left), the sum of throughfall deposition of the base cations (Bc) (a, right), the sum of throughfall deposition of potential acidifying ions (N+S) (b, left) and the sum of throughfall deposition of nitrogen delivering ions (N) (b, right) for every sampling event. Values higher than 1.0 signify higher throughfall deposition resulting from a forest edge effect; values lower than 1.0 imply lower throughfall deposition. IFEE values equal to 1.0 indicate no edge effect.

The edge effect on throughfall deposition of K^+ changed in the birch stand from negative during the growing season to positive in the dormant season. In the Corsican pine stand, an edge effect influenced throughfall deposition of K^+ in the growing season, but it was lacking in the dormant season. Throughfall deposition of Ca^{2+} reached the same DEI in both time periods, but generated a MEI that is 1.2 times higher in the dormant than in the growing season. This small enhancement in MEI was not reflected in the IFEE factor, which displayed similar values in the two seasons. For Mg^{2+}, both DEI and MEI increased in the dormant season, resulting in a higher IFEE value in the dormant season, and to a greater extent in the birch stand. In the birch stand, an edge effect on throughfall deposition of NH_4^+, NO_3^-, and SO_4^{2-} was absent or negligible in the growing season, but emerged in the dormant season. A different seasonal effect was observed in the pine stand for the N and S containing ions: the edge effect displayed smaller values for the MEI in the dormant season, while the DEI was similar in both seasons. Consequently, the IFEE factor in the pine edge was smaller in the dormant season for these ions, but the difference between the seasons was small.

Table 2.5: Depth of edge influence (DEI, m), magnitude of edge influence (MEI), and integrated forest edge enhancement (IFEE) of the edge effect on throughfall deposition in the growing and dormant season in the silver birch and Corsican pine stand. (-: no edge effect detected and therefore IFEE was not determined)

	Silver birch						Corsican pine					
	Growing season			Dormant season			Growing season			Dormant season		
	DEI	MEI	IFEE	DEI	MEI	IFEE	DEI	MEI	IFEE	DEI	MEI	IFEE
Volume	-	0.94		2	1.07	1.00	-	0.87		2	0.96	1.00
Na^+	23	1.68	1.19	50	2.88	1.93	20	4.60	1.77	31	4.80	2.12
K^+	> 50	0.83	0.88	> 50	1.57	1.45	20	1.21	1.14	-	1.01	
Ca^{2+}	6	1.06	1.03	6	1.38	1.03	20	2.07	1.31	20	2.54	1.32
Mg^{2+}	8	1.23	1.09	50	2.17	1.62	20	2.69	1.62	26	3.99	1.78
H^+	2	1.91	1.03	-	1.01		-	0.50		-	0.51	
NH_4^+	-	0.98		8	1.57	1.05	12	2.08	1.15	12	1.92	1.11
Cl^-	23	1.51	1.17	50	2.57	1.82	20	3.33	1.57	31	4.55	2.09
NO_3^-	-	1.08		8	1.43	1.03	12	2.20	1.17	12	1.86	1.11
SO_4^{2-}	2	1.20	1.01	8	1.67	1.05	12	2.19	1.16	12	2.04	1.11
Bc	-	1.00		8	1.72	1.03	20	1.66	1.29	26	2.27	1.45
N+S	2	1.08	1.01	8	1.59	1.05	12	2.14	1.16	12	1.96	1.11
N	-	1.01		8	1.54	1.05	12	2.11	1.15	12	1.91	1.10

2.5 Discussion

The application of only two throughfall collectors per distance plot was considered to be sufficient for the research questions considered in this study. Although both stands are not absolutely homogeneous, the contribution of other tree species in the total stand density or stem volume is limited. The smooth course of throughfall deposition in function of the distance to the forest edge suggested that the variation in throughfall deposition was covered sufficiently. Moreover, we expect that a higher number of throughfall collectors per distance plot would not significantly affect the results for the DEI, MEI, and IFEE factor, which were calculated by means of regression or as a ratio of means and thus by use of the throughfall deposition of more than one distance plot.

2.5.1 Ion-specific throughfall edge effects

The absence of a clear edge effect on throughfall volume is in agreement with previous studies in coniferous (Hasselrot and Grennfelt 1987; Draaijers et al. 1988; Beier and Gundersen 1989; Ivens 1990) and deciduous stands (Devlaeminck et al. 2005). The small and restricted edge effect in both edges during the dormant season may be appointed to the horizontal influx of rain droplets through the trunk area (Potts 1978; Draaijers 1993) due to higher mean wind speed during this season. As the enhancement of throughfall volume in the forest edge is restricted, also in depth, the observed higher throughfall deposition enhancement penetrating deeper into the forest edge must be attributed to other processes, such as (i) higher LAI (LAI values were elevated in the first 9 m of both forest edges) causing elevated dry deposition and/or canopy exchange (Beier and Gundersen 1989; Lovett et al. 1989; Draaijers 1993; Stachurski and Zimka 2000), (ii) an aerodynamic forest edge effect upon dry deposition (advection and inflow; Draaijers et al. 1994), and/or (iii) an edge effect on canopy exchange as a result of an edge gradient in abiotic factors such as soil, precipitation, and microclimate characteristics (Ranney et al. 1978; Lovett et al. 1989; Weathers et al. 1995; Davies-Colley et al. 2000; Honnay et al. 2002).

Throughfall deposition of Na^+, Cl^-, and Mg^{2+} displays in both stands the most pronounced edge effect, in penetration depth (DEI) as well as in edge enhancement (MEI). NO_3^-, NH_4^+, and SO_4^{2-} are subject to the smallest edge effects, showing both the smallest MEI and the shortest DEI. In the birch stand, the edge effect on these ions is even not statistically significant according to the linear fit to the logarithm of throughfall. Similarly, Beier and

Gundersen (1989), Ivens (1990), and Draaijers (1993) found larger enhancement factors for Na^+, Cl^-, and Mg^{2+} than for NO_3^-, NH_4^+, and SO_4^{2-}. Devlaeminck et al. (2005) found significant edge effects for Na^+, Cl^-, and NH_4^+ throughfall deposition, but with the lowest enhancement factor for the latter.

With respect to dry deposition, Na^+, Cl^-, Mg^{2+}, Ca^{2+}, and K^+ are assumed to be deposited as coarse particles, while NO_3^-, NH_4^+, and SO_4^{2-} are deposited as smaller particles as well as gasses. Particles larger than 1 µm display a forest edge effect that is more pronounced than in the case of small particles and gases (Ivens 1990; Draaijers 1993), which may be attributed to the difference in deposition mechanisms operating on gasses, submicron and supermicron particles. Draaijers (1993) putt forward that in-canopy wind speeds determine to a large extent the difference between forest edge and forest interior dry deposition of Na^+, Cl^-, and Mg^{2+}. For NO_3^-, NH_4^+, and SO_4^{2-}, differences between forest edge and forest interior dry deposition may be more explained by differences in turbulence intensities. When precipitation passes through the canopy, its chemical composition may not only be altered by washing off dry deposition, but even so by canopy exchange (Parker 1983; Erisman et al. 1994). Na^+ and Cl^- are considered to pass through the canopy without or with negligible interaction (Ivens 1990; Draaijers et al. 1997; Houle et al. 1999; Stachurski and Zimka 2002). Sulphur is assumed to act as a conservative element in respect to the canopy, as stomatal uptake of SO_2 is thought to be balanced by canopy leaching of SO_4^{2-} (Lindberg and Lovett 1992; Draaijers et al. 1997; Stachurski and Zimka 2002). On the other hand, NO_3^- can be absorbed by the canopy, but to a lesser extent than NH_4^+ (Lovett and Lindberg 1984, 1993; Neary and Gizyn 1994; Houle et al. 1999; Stachurski and Zimka 2000, 2002). K^+ is released by the canopy (Parker 1983; Ulrich 1983; Lovett 1992; Neary and Gizyn 1994; Stachurski and Zimka 2000) and $Mg^{2+}$and Ca^{2+} can either be taken up or released (Ulrich 1983; Alcock and Morton 1981; Ivens 1990). As a result, the more pronounced edge effects on the throughfall deposition of Na^+ and Cl^- and the less pronounced edge effect for SO_4^{2-} are in line with the dry deposition that would be expected. We found no or only small edge effects for K^+ throughfall deposition, in line with Beier and Gundersen (1989), Spangenberg and Kölling (2004), and Devlaeminck et al. (2005). This likely results from a smoothing influence of canopy exchange on the presumed pronounced edge effect on dry deposition of larger particles such as K^+, as it originates to a large extent from canopy leaching (De Schrijver et al. 2004; Staelens et al. 2006). Ca^{2+} and Mg^{2+} display more pronounced edge effects on throughfall deposition than K^+, which can be explained by the larger contribution of dry deposition to the throughfall flux of Mg^{2+} and Ca^{2+}

in comparison to K^+ (Draaijers et al. 1997; De Schrijver et al. 2004). The absence or negligibility of edge effects on the H^+ throughfall deposition may be the consequence of higher canopy buffering at the edge or reaction of with NH_3, forming NH_4^+ (Beier and Gundersen 1989).

The values found for the measure 'depth of edge influence' are small in comparison to studies by Hasselrot and Grennfelt (1987), Draaijers et al. (1988), Draaijers (1993), Spangenberg and Kölling (2004), and Devlaeminck et al. (2005) but comparable with results of Beier and Gundersen (1989) and Weathers et al. (2001). However, comparison of our values with literature data is restricted because of the dissimilarity in the definition and the methodology of the calculation of the measure 'magnitude of edge influence' (De Schrijver et al. 2007a) throughout these studies.

2.5.2 Forest structure effect

For all ions except H^+, the pine stand is subjected to more pronounced edge effects on throughfall deposition than the birch stand, a result of both a larger DEI and a larger MEI. When every sampling occasion is considered, the edge effect is significantly more pronounced for Na^+, the sum of potentially acidifying ions, and the sum of base cations, but the difference between both forest types is not significant for the sum of inorganic N throughfall. The ratio of potentially acidifying ions to base cation throughfall deposition generated by the edge effect is similar in both forest edges: for each equivalent deposition of base cations that the stand receives as a result of the edge effect, 1.9 equivalents of potential acidifying N+S ions is deposited. Nevertheless, the pine edge is subject to a throughfall deposition surplus of potentially acidifying ions that is more than 9 times the surplus in the birch edge, which results in a 3.7 times higher throughfall deposition of N+S in the pine edge zone than in the birch edge zone. A fixed 50 m forest edge in the pine stand receives twice as much potential acidifying ions but the same amount of base cations as a 50 m forest edge in the birch stand. Although no significant difference in the IFEE factor for N throughfall is found between the two stands, indicating a similar edge effect on N, the N throughfall deposition surplus is 12.7 times smaller in the birch stand. This results from the lower throughfall deposition input in the birch forest interior in comparison to the pine forest interior, as the IFEE factor is expressed relatively to this forest interior deposition. A fixed 50 m forest edge in the pine stand receives more than two times the amount of N throughfall deposition a 50 m forest edge in the birch stand receives. The result that edge effects were more pronounced in the deciduous forest than in the coniferous one is in line with the findings of Balsberg-Påhlsson and Bergkvist (1995) in a beech and Norway spruce stand.

Important factors determining the forest edge effect on ion deposition are meteorological conditions, edge aspect, and edge structure characteristics. Considering forest edge structure, Draaijers (1993) found net throughfall enhancement in edges to depend strongly on forest density. Overall, old, closed forest edges are more efficient in trapping and concentrating nutrients, and thereby limit the penetration of edge effects to a greater extent than new, open edges (Weathers et al. 2001). The stands considered in this study differed in tree species composition, stand density, and age and hence in mean tree height (Table 2.1) and LAI (Fig. 2.3). In addition, there is a difference between the 'summergreen' and the 'evergreen character' of the silver birch and Corsican pine stand, respectively. Finally, the leaf shape affects the amount of elements deposited, as leaves with long narrow shape (needles) are more efficient in salt accumulation than circular ones (Woodcock 1953 in Smith 1981). As meteorological conditions and edge aspect were identical for both forest edges, differences in edge effect on throughfall deposition between the two stands must be the result of differences in edge structure, i.e., structure characteristics of the edge influencing in-canopy wind speeds and turbulence intensities (Draaijers 1993). Conversion of the edge structure, in this case from high-density Corsican pine plantations into lower density deciduous forests, can reduce the input of potentially acidifying and nitrogen delivering pollutants in the forest edge. Well-considered forest edge structure may consequently diminish the level of output, i.e., nitrate seepage to groundwater and soil acidification, through a reduction of input. To protect the forest in its entirety, both decreases in forest edge distance and edge enhancement for throughfall deposition should be intended. Regarding these results, it should be kept in mind that the study period was limited to one year and that deposition load and other environmental factors, such as meteorological conditions and the appearance of (a)biotic stress, may vary between years. Nonetheless, the differences in edge effect between the birch and the pine stand were vast. Furthermore, the MEI and the IFEE factor express the edge effect relative to the forest interior throughfall deposition, which makes them more robust in regard to annual variations in environmental factors than when only throughfall deposition values in the forest edge would be considered.

2.5.3 Effect of season

The birch and the pine stand showed dissimilar temporal variation in forest edge patterns, with the birch stand displaying the largest divergence in edge effect between the dormant and the vegetation season. Clear temporal variation in the forest edge effect occurs in the birch stand, where throughfall deposition during the growing season is less or even not influenced by the edge in contrast to the dormant season. Beier and Gundersen (1989) also found very small edge effects during the growing season; for SO_4^{2-}, NO_3^-, Ca^{2+}, and Mg^{2+}, the edge effect even was almost restricted to the front trees. Also in shorter time periods (when considering every sampling event), the IFEE factors vary mostly in the birch edge, with Na^+ and Bc fluxes displaying the highest variation.

According to Draaijers et al. (1988), the edge zone with increased throughfall deposition is wider and the level of enhancement at the edge is higher in case of high wind speeds. The relative increase also depends on the wind direction during the dry period preceding a rainstorm: throughfall deposition flux gradients are steeper when the proportion of wind blowing towards the forest edge is larger. Therefore, edge aspect influences the forest edge effect: in forest edges oriented to other than the prevailing wind directions, higher deposition is restricted to the first few meters of the forest edge caused by some small turbulent currents at the leeside (De Schrijver et al. 1998). In both stands, the edge effect on throughfall deposition of Na^+ and Cl^-, which are indicators for dry particulate deposition, penetrated deeper during the dormant season. The increase of DEI and MEI for Na^+ and Cl^- throughfall deposition in the pine stand was most probably the result of different meteorological conditions (e.g., higher wind speeds and different prevailing wind direction during the dormant season). It is hypothesized that the higher DEI and MEI values in the birch stand during the dormant season compared to the growing season are the result of both different meteorological conditions and the absence of leaves. According to the significant correlation, the direction weighted summed wind speed is shown to be a good predicting factor for the seasonal fluctuation of the IFEE factor for Na^+ throughfall deposition in both stands. Leaf loss in the birch stand may have a strengthening effect on the increase of DEI and MEI in the dormant season: the edge effect is able to penetrate to a deeper extent into the stand which may be due to slower deceleration of wind speed in the crown area.

In both stands, K^+ throughfall deposition exhibits very different edge effects in the two time periods considered. As K^+ in throughfall deposition originates mostly from canopy exchange, differences in edge effect on throughfall deposition may primarily be the result of differences in canopy exchange processes and its contribution to the throughfall flux. Differences in DEI and MEI for Mg^{2+} throughfall deposition between both periods are comparable with those for Na^+ and Cl^-, which can be explained by the similar deposition processes as particles, added to the smaller contribution of canopy exchange to throughfall deposition than for K^+ and/or a possible similar edge effect on Mg^{2+} canopy exchange in both seasons. For Ca^{2+} throughfall deposition, the contribution of canopy exchange to throughfall deposition is rather low, nonetheless differences in edge effects on dry deposition between both periods can be inferred or surpassed by different canopy exchange patterns along the transect in the dormant and the growing season (e.g., lower canopy exchange at the forest edge in comparison to the forest interior during the dormant season).

During the growing season in the birch edge, no edge effect occurred for NH_4^+ and NO_3^- and, for SO_4^{2-}, only a short and small edge effect was found, while all three ions exhibited an edge effect in the dormant season. This finding could again be related to (i) the different meteorological conditions (e.g., different wind speed and prevailing wind direction) between the seasons, which may influence the deposition of particulate SO_4^{2-}, NH_4^+, and NO_3^-, and to (ii) the absence of leaves in the dormant season. For NH_4^+ and NO_3^-, this effect can be strengthened or weakened by possibly different edge effects on canopy uptake. However, the edge effect on SO_4^{2-} throughfall deposition in the pine stand is comparable in both seasons, implying that seasonal differences in meteorological conditions have a minor impact on the edge effect of SO_4^{2-} fluxes and probably also of dry deposition of other ions deposited as gasses or fine particles. Thus, the appearance of an edge effect in the birch stand during the dormant season is primarily a consequence of leaf loss for SO_4^{2-} and of leaf loss and/or differences in edge effects on canopy uptake for NH_4^+ and NO_3^-. However, the edge effects for NH_4^+ and NO_3^- in the birch stand in the dormant season are of the same magnitude as for SO_4^{2-} (analogous DEI and IFEE values), so the importance of possibly different edge effects on the canopy uptake of NH_4^+ and NO_3^- during the two seasons likely is minimal.

For the sum of potentially acidifying ions and the sum of base cations on a yearly basis, differences in edge effects between both stands primarily originate from differences in edge effects during the growing season. In the birch transect, no or a negligible throughfall deposition enhancement is displayed during the growing season, in contrast to the dormant season. However, edge effects in the pine transect are similar in both seasons. For Na^+ throughfall deposition, difference in edge effect between the two stands occurs during the dormant season and the vegetation season. For N, difference in edge effect between the stands occurs in neither season.

2.6 Conclusions

The use of the parameter Integrated Forest Edge Enhancement (IFEE) allowed us to compare edge effects as a whole, instead of making a comparison by use of Magnitude and Depth of Edge Influence separately. Furthermore, the IFEE factor would facilitate (i) the comparison of edge effects between different studies, as forest edge enhancement is inconsistently defined through the variety of surveys, and (ii) the implementation into calculations of exceedance of critical loads, taking the edge effect into account.

The forest edge effect, expressed as an IFEE value, on throughfall deposition of potentially acidifying ions (N+S) was significantly more pronounced in the stand dominated by Corsican pine than in the silver birch stand, a result of both a larger depth of influence and a higher deposition enhancement at the edge. The edge effect in the pine stand generated a surplus of potentially acidifying throughfall deposition in the edge zone that was 9.4 times as high as in the edge zone with birch. This difference between the stands originated primarily from differences in edge effects between the stands during the growing season. Although the edges of the two stands generated no significantly different edge effects on inorganic N throughfall deposition when expressed as IFEE factor (i.e., relative to the forest interior flux), the edge effect caused in the pine stand an inorganic N throughfall deposition surplus of more than 12 times the surplus in the birch stand.

We conclude that an appropriate design or conversion of the edge structure, from high-density Corsican pine plantations into lower density deciduous forests, can reduce the input of acidifying and nitrogen delivering pollutants in the forest edge. Hereby, a decrease in both the depth of influence and the enhancement at the edge should be aimed at.

3 Forest edge effects on throughfall deposition in different forest types

After: Wuyts, K., De Schrijver, A., Staelens, J., Gielis, L., Vandenbruwane, J., Verheyen, K. 2008. Comparison of forest edge effects on throughfall deposition in different forest types. Environmental Pollution 156, 854-861.

3.1 Abstract

This study examined the influence of distance to the forest edge, forest type, and time on Cl^-, SO_4^{2-}, NO_3^-, and NH_4^+ throughfall deposition in forest edges. The forests were dominated by pedunculate oak, silver birch, or Corsican/Austrian pine, and were situated in two regions of Flanders (Belgium). Along transects, throughfall deposition was monitored at distances of 0-128 m from the forest edge. A repeated measures analysis demonstrated that time, forest type, and distance to the forest edge significantly influenced throughfall deposition of the ions studied. The effect of distance to the forest edge depended significantly on forest type in the deposition of Cl^-, SO_4^{2-}, and NO_3^-: the edge effect was significantly greater in pine stands than in deciduous birch and oak stands. This finding supports the possibility of converting of pine plantations into oak or birch forests in order to mitigate the input of nitrogen and potentially acidifying deposition.

3.2 Introduction

In general, forest edges are steep transitions of vegetation height, which disturb the vertical wind profile. When approaching a forest edge, wind flow partially penetrates it and partially lifts over it. As a result, wind speed and air turbulence are enhanced at the forest edge compared to the forest interior, increasing dry deposition velocities via inflow and advection processes (Draaijers et al. 1994). Through inflow, pollutants are blown onto the collecting surface in the forest edge and advection enhances exchange of pollutants between the atmosphere and the forest (Draaijers et al. 1994). The resulting dry deposition enhancement in the forest edge has been amply demonstrated by deposition models (e.g., Wiman and Ågren 1985), wind tunnel studies (e.g., Ould-Dada et al. 2002), air concentration measurements (e.g., Wiman and Lannefors 1985), and throughfall deposition measurements (e.g., Hasselrot and Grennfelt 1987; Weathers et al. 2001; Wuyts et al. 2008a, chapter 2; for an overview see De Schrijver et al. 2007a]. In forest edges, throughfall deposition is increased up to four times compared to the forest interior. This so-called forest edge effect decreases exponentially with increasing distance from the edge, until it reaches a more or less interior forest level starting at 8-108 m from the edge (Draaijers 1993; Devlaeminck et al. 2005; De Schrijver et al. 2007a; Wuyts et al. 2008a, chapter 2). The level of throughfall deposition enhancement and the penetration depth of the edge effect depend on several factors, including the ion under consideration (Beier and Gundersen 1987; Draaijers et al. 1994; Spangenberg and Kölling 2004; Wuyts et al. 2008a, chapter 2), meteorological conditions such as wind speed and direction (Draaijers et al. 1988; Wuyts et al. 2008a, chapter 2), edge orientation (Draaijers et al. 1994; De Schrijver et al. 1998), and edge structural features such as leaf area index (LAI) and stand density (Draaijers 1993; Weathers et al. 2001; Wuyts et al. 2008a, chapter 2). Research on how forest type influences edge effects is lacking, except for a comparison of *Picea abies* (L.) Karst. and *Fagus sylvatica* L. by Spangenberg and Kölling (2004) and by Balsberg-Påhlsson and Bergkvist (1995) and our study in chapter 2 (Wuyts et al. 2008a) on *Pinus nigra* ssp. laricio Maire and *Betula pendula* Roth.

Recent studies have shown that forest type significantly influences on throughfall deposition in forest ecosystems (Van Ek and Draaijers 1994; Lovett et al. 1996; Robertson et al. 2000; Augusto et al. 2002; Rothe et al. 2002; De Schrijver et al. 2004, 2007b, 2008; Oulehle and Hruska 2005; Herrmann et al. 2006). A review by De Schrijver et al. (2007b) revealed that, when the site and climate are similar, coniferous forests annually receive significantly higher throughfall deposition quantities of nitrogen (N) and sulphur (S) than do deciduous forests.

The review revealed that in regions with high open field N deposition (> 10 kg N ha^{-1} y^{-1}) throughfall deposition was higher in coniferous stands than in deciduous stands, and that this difference increased with increasing NH_4^+ deposition in open field. In regions with relatively low open-field N deposition (< 10 kg N ha^{-1} y^{-1}), NH_4^+ throughfall deposition fluxes were lower under coniferous canopies than under deciduous ones. Furthermore, Van Ek and Draaijers (1994) found similar throughfall deposition fluxes of SO_4^{2-} for stands of pedunculate oak (*Quercus robur* L.) and Scots pine (*Pinus sylvestris* L.), which they attributed to enhanced canopy leaching, induced by leaf sprouting, in the oak stands. Consequently, both time and region are expected to influence the forest type effect on throughfall deposition of potentially acidifying ions (N+S), as well as of inorganic N (NH_4^+ + NO_3^-).

Conversion of secondary spruce and pine forests into deciduous forests, on sites where deciduous tree species would thrive under natural conditions, has been suggested for several reasons. These include a decrease in the deposition of N and potentially acidifying ions and a consequent decrease in soil acidity (Brandtberg et al. 2000; Brandtberg and Simonsson 2003), a decrease in seepage of NO_3^- and SO_4^{2-} and accompanying cations into groundwater (von Wilpert et al. 2000; De Schrijver et al. 2004; Herrmann et al. 2005), and less biodiversity loss (Gärtner and Reif 2004). In Flanders, most of the Scots and Corsican pine plantations are located on sandy soils (Bos & Groen 2001), which are particularly sensitive to acidification (Van Ranst et al. 2002; De Schrijver et al. 2006). In the first successional stages, pedunculate oak and silver birch are the intended tree species in the closer-to-natural forest ecosystems. In order to gain more insight into how edge effects are influenced by forest type, we compared throughfall deposition of N+S and N in six forest stands that differed in tree species composition along transects starting at the forest edge and ending in the forest interior. In addition, we examined the influence of region and time, and their interaction with the forest type effecting influencing edge effects. To accomplish this, we chose three forest types (*Q. robur*, *B. pendula*, and *P. nigra*) and selected two regions, one in the west and one in the east of Flanders.

3.3 Materials and methods

3.3.1 Site description

We selected two regions in the northern part of Belgium (Flanders), located approximately 135 km apart (Fig. 3.1). These regions were characterized by high deposition of NH_3 as a result of intensive livestock breeding. Region 1 is near the North Sea coast and may be subject to higher wind speeds and a higher load of sea spray (e.g., NaCl and $MgCl_2$) than region 2, which lies further inland. Mean pollutant concentration in the two areas was calculated from air concentration data obtained from three measuring stations of the Vlaamse Milieumaatschappij measuring program: AC1 at Wingene and AC2 at Zwevegem for the western region, and AC3 at Retie for the eastern region. During the study period, mean SO_2 concentrations were 3.27, 3.46, and 3.33 µg m^{-3}, and mean NO_2 concentrations were 14.9, 16.9, and 17.0 µg m^{-3}, at AC1, AC2, and AC3, respectively.

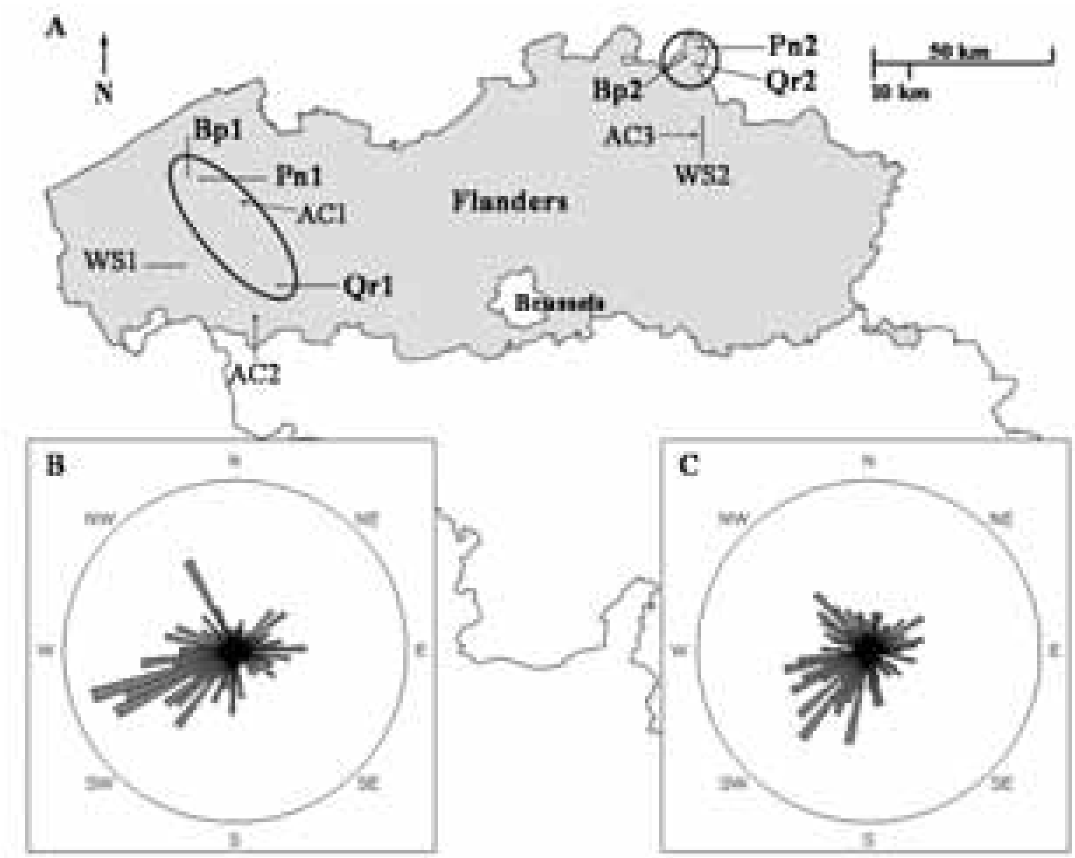

Fig. 3.1: A. Map of Flanders, the northern part of Belgium, indicating the location of the two regions containing the six forest stands in this study. Qr, pedunculate oak; Bp, silver birch; Pn, Corsican/Austrian pine; WS, weather station; 1, western region; 2, eastern region. B. Wind rose at Beitem weather station (WS1). C. Wind rose at Retie weather station (WS2).

Climatological data were obtained from two weather stations of the Royal Meteorological Institute of Belgium, WS1 at Beitem (Fig. 3.1a) was selected for the western region and WS2 at Retie for the eastern region. During the study period, mean temperature was 10.4°C (WS1) and 10.6°C (WS2). Mean temperature of the coldest month (January) was 1.4°C and 0.9°C, and mean temperature of the warmest month (July) was 21.7°C and 22.0°C. Relative humidity was, on average, 80 % (WS1) and 85 % (WS2). Mean wind speed during the study period was 3.7 (± st. dev. 1.6) m s^{-1} and 2.6 (± 1.2) m s^{-1} at WS1 and WS2, respectively. The wind rose diagrams of WS1 (Fig. 3.1b) and WS2 (Fig. 3.1c) illustrate that the daily wind direction was south to west during 37 % (WS1) and 43 % (WS2) of the study period. From our open-field data, we calculated that rainfall during the study period amounted to a mean of 786 mm in the western region and 783 mm in the eastern region. Of this rainfall, a mean of 53 % and 60 % fell during the spring-summer period.

In each region, three forest stands were selected on poor sandy soils (Haplic podzols; FAO-ISRIC-ISSS 1998) with low buffering capacity for acidifying deposition (De Schrijver et al. 2006; Van Ranst et al. 2002). All stands in the study were dominated by one tree species: pedunculate oak (*Quercus robur* L.), silver birch (*Betula pendula* Roth), Corsican pine (*Pinus nigra* ssp. *laricio* Maire), or Austrian pine (*P. nigra* ssp. *nigra* var. *nigra* Arnold). The stands had an abrupt forest edge lacking any gradual edge vegetation, either shrub or herbal. The edges were oriented towards the prevailing southwesterly winds. The open terrain in front of the forest edges was not interrupted by forest fragments, hedgerows, or single trees for 250 m or more. According to Gash (1986), this is a sufficient fetch length for the wind profile to adjust to the open-field surface following a forest-heath interface. Site code, location, stand characteristics, and mean pH (KCl) of the forest types in the study are presented in Table 3.1. The stands Qr1, Bp1, and Pn1 were situated in the western part of Flanders and Qr2, Bp2, and Pn2 in the eastern part. Canopy openness was measured before bud burst (between 15 March and 15 April 2006) and in late summer (between 31 August and 15 September 2006) by means of digital hemispherical photographs (Nikon D70S with fish-eye lens Sigma EXDG Fisheye 8 mm 1:4 D). Photos were taken above every throughfall collector of the distance plots at 64 and 128 m from the edge, and they were processed with Gap Light Analyzer GLA 2.0 (www.ecostudies.org/gla).

Table 3.1: Description of the six sites analysed: site code, latitude, and longitude of the sites and age, stand density (SN), mean tree height (H), dominant tree height (Hdom), basal area (BA), stem volume (V) of the dominant tree species, mean canopy openness (CO) as measured in 2006, and mean pH (KCl) values for the upper 0-0.05 m and 0.05-0.10 m of the mineral soil. -: not measured.

Site code	Location	Dominant tree species	Age	SN	H	H_{dom}	BA	V	CO		pH (KCl)	
									summer	winter	0-0.05	0.05-0.10
			(y)	(ha^{-1})	(m)	(m)	($m^2.ha^{-1}$)	($m^3.ha^{-1}$)	(%)	(%)	(-)	(-)
Qr 1	50°52'08"N 03°27'59"E	*Quercus robur*	90	187	24.2	26.4	31	343	20.5	52.3	3.36	3.25
Qr 2	51°24'44"N 05°02'45"E	*Quercus robur*	68	135	21.1	22.4	23	221	23.0	68.1	2.89	3.15
Bp 1	51°09'22"N 03°04'48"E	*Betula pendula*	30-40	3628	11.2	18.2	26	194	32.5	41.3	2.93	3.28
Bp 2	51°25'56"N 05°00'31"E	*Betula pendula*	20-30	2715	8.1	13.8	13	74	36.5	70.2	3.04	3.37
Pn 1	51°08'26"N 03°06'36"E	*Pinus nigra* ssp. *nigra*	65	388	19.2	22.0	36	336	32.0	-	2.91	3.16
Pn 2	51°26'37"N 05°05'14"E	*Pinus nigra* ssp. *laricio*	43	1162	17.3	19.1	55	488	18.2	-	2.90	3.11

Values of pH (KCl) were very low at all sites (Table 3.1), which is a result of former land use. From historic maps from around 1775 and later, we can infer that the forest sites we studied were until up to 80 years ago managed as heath. Grazing by sheep and turf cutting resulted in a depletion of nutrients in the heathland (Webb 1998). So, at the time forest was established on these sites, these sandy soils, which are already naturally poor in exchangeable base cations, were extremely impoverished and acidic.

3.3.2 Experimental setup and sample analysis

At each experimental site, a transect was established perpendicular to the forest edge, along which throughfall water was collected at distances of 0, 2, 4, 8, 16, 32, 64, and 128 m from the forest edge. At each distance, three throughfall collectors were placed parallel to the forest edge and were spaced 5 m apart. For each experimental site, open-field collectors were placed in the nearest sufficiently large grassland or recent clear cut, at least 50 m from the nearest forest edge. Because of intensive agricultural land use, no open-field collectors could be installed near the pine stand in the western region (Pn1). As the silver birch stand (Bp1) was situated close to this pine stand, one location was selected for collecting open-field deposition for both stands. The throughfall and open-field collectors are described in chapter 2 (§2.3.2). For practical reasons, stemflow was not considered in this study. A large number of trees are necessary for a representative sampling of stemflow fluxes (Van Ek and Draaijers 1994) and the number of forest stands and of experimental distance plots per stand did not permit adequate sampling. From the literature, it could be deduced that the contribution of stemflow to the total flux into the forest floor is small or negligible for the tree species in this study (e.g., Neirynck et al. (2004) for *P. nigra*, Alcock and Morton (1985) for *B. pendula*, and Nachtergale et al. (2002) for *Q. robur*).

Sample collection took place every fortnight throughout the year, from 1 September 2005 to 30 August 2006. Sampling procedure was similar as in chapter 2 (§2.3.2). Samples potentially contaminated by bird droppings (e.g., spots on the funnel) were excluded from pooling and analysed separately. Aliquots of two consecutive sample collections were pooled into volume-weighted monthly samples, which were filtered (0.45 µm, Rothe) and analysed for NH_4^+ by photometric determination of a reaction product of NH_4^+ at λ = 660 nm according to the Dutch standard method NEN 6576 using a Cary 50 spectrophotometer (Varian). The samples were also analysed for Cl^-, NO_3^-, SO_4^{2-}, and PO_4^{3-} using ion chromatography with an ICS-90 Dionex system (with an IonPac AS14A-5µm column and an IonPac AG14A-5µm guard column). The quality of the chemical analyses was checked through the use of method blanks and repeated measurements of internal quality controls. Measurements of standard reference material CRM 409 (Quevauviller et al. 1993) were made throughout the study period and all ions showed coefficients of variation less than 5 % and recovery rates higher than 90 %.

Possibly contaminated samples were analysed separately. The NH_4^+ flux was omitted from calculations if the PO_4^{3-} and NH_4^+ concentrations were higher than the concentrations in the samples from the remaining throughfall collectors of the particular distance plot under consideration. If no anomalous PO_4^{3-} and NH_4^+ concentrations were detected, the samples were re-included in the calculation of the monthly fluxes. Because of vandalism at the Bp2 stand, data for one of the fortnightly data collection runs were missing, so eight of 96 monthly measurements of throughfall ion deposition and water volume had to be recalculated. The missing monthly fluxes were recalculated based on the monthly fluxes at the two neighbouring distance plots along the transect, and on the ratios of the flux of the plot under consideration to the fluxes at the two neighbouring plots at the time of the ‘undisturbed’ fortnight in that month.

2.3. Element input and data analysis

The monthly throughfall deposition fluxes were calculated from the ion concentration in the subsample of a given plot, multiplied by the monthly throughfall volume in the three collectors and divided by the surface area of the three collector funnels. A repeated-measures ANOVA analysis (SPSS 12.0) was applied separately to the monthly fluxes of Cl^-, SO_4^{2-}, NO_3^-, NH_4^+, and water volume, with ‘time’ as a within-subject factor. The following parameters were applied as between-subjects factors: ‘region’ [west (region 1) and east (region 2) of Flanders], ‘forest type’ (*Q. robur*, *B. pendula*, and *P. nigra*), and ‘distance to the forest edge’ (0-128 m from the edge). No three-way interactions were included in the model, since the degrees of freedom were too low. Von Ende (2001) stated that when time is the within-subject factor, data collected on adjacent sampling dates are usually more highly correlated than are data from separated sampling dates, and so the circularity (or sphericity) condition for repeated-measures analysis is not met in this study. To avoid this sphericity assumption, the MANOVA approach (based on Pillai’s test statistic) was adopted for analyzing the within-subject effects (O’Brien and Kaiser 1985). As differences in throughfall water volume between the forest types may be obscured by differences in open-field deposition between the stands, the repeated-measures ANOVA was also performed on the proportion of open-field water volume that reaches the forest floor as throughfall water volume. Prior to the analyses, the data were tested for normality and homoscedasticity.

Based on the year-round throughfall flux (obtained by summing all twelve monthly fluxes), we calculated the penetration depth (DEI)and the Integrated Forest Edge Enhancement factor (IFEE). Calculations were performed for both the sample volume and the ion deposition fluxes of all forest edges in the study, as described in chapter 2 (§2.3.6). The level of deposition enhancement (MEI) was calculated as the ratio of the mean throughfall deposition (or volume) at 0 and 2 m from the edge to the mean throughfall deposition (or volume) at 64 and 128 m. Because throughfall deposition enhancement in some of our stands was appreciable to a distance of 64 m from the forest edge, the calculation of the IFEE factor was performed for an edge zone of 64 m instead of 50 m. The IFEE factor was not calculated if no DEI could be delineated, i.e. if the slope α of the linear regression of the ln-transformed throughfall data did not differ significantly from zero. Finally, the IFEE factors of the potentially acidifying ions SO_4^{2-}, NO_3^-, and NH_4^+ of the six forest stands were used to perform a Mann-Whitney U test and a hierarchical cluster analysis using complete linkage clustering.

3.4 Results

Comparing the different forest types within the same region, throughfall deposition of all ions was higher in the pine stands than in either the pedunculate oak or the silver birch stands (Table 3.2 and Fig. 3.2). Considering the plot at 128 m from the forest edge (forest interior plot) in the three forest types within the same region, throughfall deposition of Cl^-, inorganic N (NO_3^- + NH_4^+), and the sum of inorganic N and SO_4^{2-} (N+S) was, on average, 1.6, 1.9, and 1.6 times higher in the pine stands than in the oak stands. Throughfall deposition fluxes were 1.3, 2.2, and 2.0 times higher for pine than for silver birch for Cl^-, N, and N+S. The oak stands received, on average, 0.9 times less Cl^- and 1.2 and 1.3 times more N and N+S by throughfall deposition than did the birch stands.

Table 3.2: Open-field water (mm y^{-1}) and ion deposition (equiv ha^{-1} y^{-1}) fluxes adjacent to six forest stands and mean throughfall water and ion fluxes at the forest edge (0 - 2 m) and the forest interior (64 - 128 m). The interior throughfall water volume as a fraction of open-field water volume is given in parentheses. (a: one location for collecting open-field deposition was selected for both the Bp1 and Pn1 sites)

Site code	Flux	Water	Cl^-	SO_4^{2-}	NO_3^-	NH_4^+
Qr1	Open-field	762	433	341	249	491
	Edge	516	1174	1180	416	906
	Interior	584 (76)	793	939	353	896
Qr2	Open-field	831	439	474	298	632
	Edge	704	1153	1532	559	1602
	Interior	621 (75)	692	1055	355	1314
Bp1	Open-field	810	782	407	269	709
	Edge	568	1584	825	314	947
	Interior	580 (72)	1096	542	264	720
Bp2	Open-field	730	415	497	298	687
	Edge	569	1279	1517	504	1541
	Interior	648 (88)	699	971	393	1298
Pn1	Open-field[a]	810	782	407	269	709
	Edge	613	4556	2212	1154	3185
	Interior	568 (70)	1518	1105	498	1719
Pn2	Open-field	790	415	455	291	546
	Edge	565	1873	2066	890	2791
	Interior	559 (71)	768	1105	575	2251

Comparing the same forest type between the two regions, throughfall fluxes of NH_4^+ in the forest interior (128 m plot) were consistently higher in the eastern part of Flanders than in the western part. Throughfall deposition fluxes of SO_4^{2-} were also higher in the eastern region, particularly in the silver birch stands. The difference between the regions was, however, more pronounced for $NH4^+$ than for SO_4^{2-}. For Cl^-, in contrast, fluxes were higher in the western part of Flanders for both the pine and silver birch stands. For NO_3^-, no difference in the forest interior throughfall deposition in a given forest type was observed between the two regions.

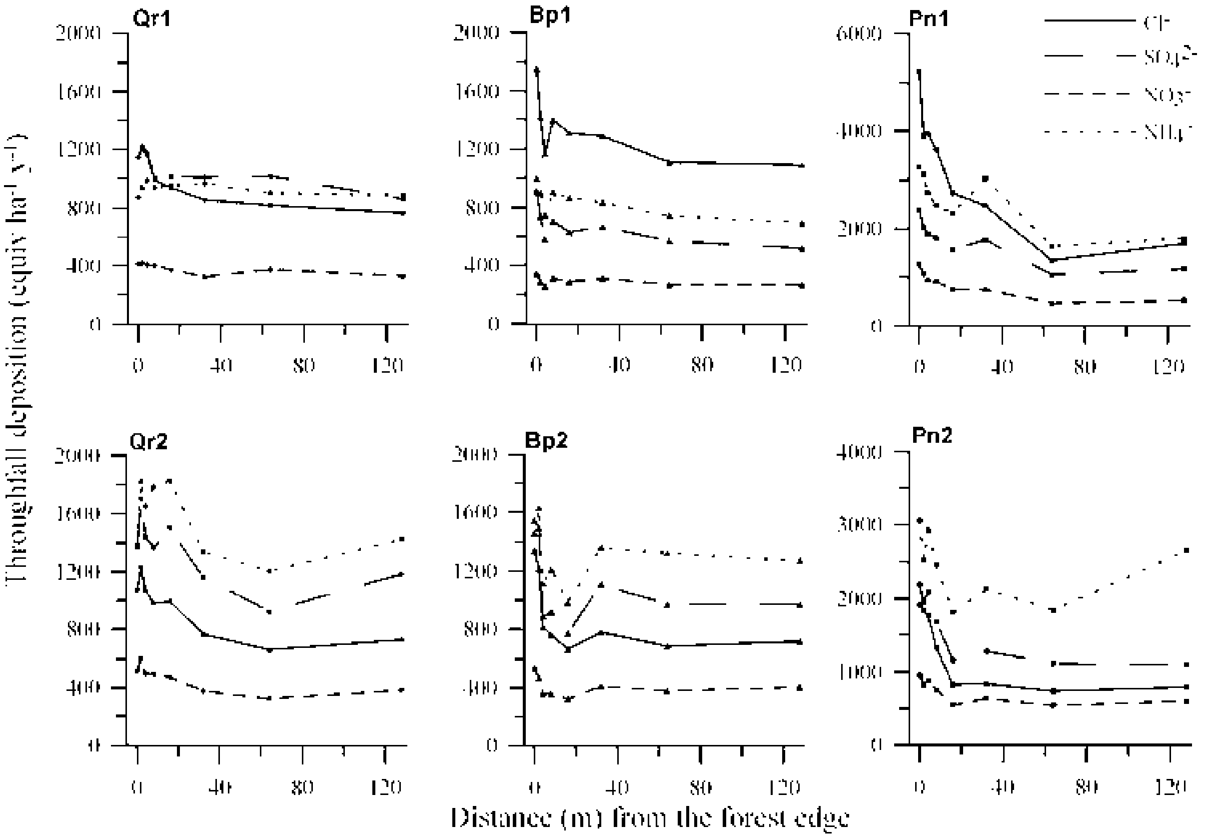

Fig. 3.2: Throughfall deposition of Cl^-, SO_4^{2-}, NO_3^-, and NH_4^+ (equiv ha^{-1} y^{-1}) at the forest edge of six stands. Qr, pedunculate oak; Bp, silver birch; Pn, Corsican/Austrian pine. The upper row (1) shows the three sites in the western part of Flanders, and the lower row (2) shows the eastern part of Flanders. Note the different range on the y-axis of the pine plots.

Throughfall ion deposition at the forest edges of all stands was visibly enhanced in comparison with the forest interior, with the exception of NO_3^- in the Bp1 stand (Table 3.2 and Fig. 3.2). The most pronounced edge effect for all six transects was detected for Cl^- throughfall deposition, with the highest enhancements at the very edge of the stand. Overall, the strongest edge effects were found in the pedunculate oak stand in the east (Qr2) and the pine stand in the west (Pn1), while the pedunculate oak stand and the silver birch stand in the west (Qr1 and Bp1) showed the smallest edge effects. Compared to the forest interior, throughfall water volume seemed enhanced at the edge of the oak stands (Qr1 and Qr2) and the pine stand in the east (Pn2) (data not shown). According to the repeated-measures ANOVA (Table 3.3), throughfall water volume was significantly affected by time ($p < 0.05$), but not by region, forest type, or distance to the forest edge. This time effect on water fluxes significantly interacted with the effects of region, forest type and distance to the forest edge. Throughfall water volume expressed as a fraction of open-field water volume was significantly affected by region ($p = 0.001$) and forest type ($p = 0.018$), with a significant interaction between region and forest type ($p = 0.015$). The throughfall ion fluxes of Cl^-, SO_4^{2-}, NO_3^-, and NH_4^+ were significantly influenced by time, forest type, and distance to the forest edge. Significant interactions with time were observed with the factors of region and forest type for all the ions, as well as with the factor of distance to the forest edge in the case of Cl^-. Furthermore, the effect of forest type significantly interacted with the factor of region for all ions. The factor of distance to the forest edge interacted significantly with the factor of forest type for Cl^-, SO_4^{2-}, and NO_3^-, as well as with time in the case of Cl^-. No significant interaction was observed between distance to the forest edge and region. Region significantly affected throughfall deposition of Cl^-, SO_4^{2-}, and NH_4^+.

Table 3.3: Effect of region, forest type, distance to the forest edge, and time on the throughfall volume and the throughfall deposition of Cl^-, SO_4^{2-}, NO_3^-, and NH_4^+. Bold: $p < 0.05$.

Source of variation	Water	Cl^-	SO_4^{2-}	NO_3^-	NH_4^+
Within-subjects					
time	< **0.001**	**0.001**	< **0.001**	< **0.001**	**0.001**
time x region	< **0.001**	**0.011**	< **0.001**	**0.001**	**0.004**
time x forest type	< **0.001**	< **0.001**	< **0.001**	< **0.001**	< **0.001**
time x distance to forest edge	< **0.001**	**0.020**	0.094	0.557	0.361
Between-subjects					
region	0.052	< **0.001**	**0.001**	0.194	**0.001**
forest type	0.209	< **0.001**	< **0.001**	< **0.001**	< **0.001**
distance to forest edge	0.690	< **0.001**	< **0.001**	< **0.001**	**0.033**
region x forest type	**0.015**	< **0.001**	< **0.001**	**0.002**	**0.004**
region x distance to forest edge	0.933	0.506	0.594	0.920	0.614
forest type x distance to forest edge	0.928	**0.032**	**0.011**	**0.012**	0.122

Fig. 3.3 presents the model-estimated mean monthly throughfall deposition for each forest type and ion considered at each sampling distance from the forest edge. For all ions, the edge effect in the pine stands was more pronounced than in either type of deciduous forest. In the oak stands, the DEI was similar to that in the pine stands, but the forest edge enhancement (MEI), defined as the enhancement of deposition at the very edge in comparison with deposition in the forest interior, was small compared to the MEI in the pine stands. The differences in edge effect between the birch and the pine stands originated from both a smaller DEI and a smaller MEI.

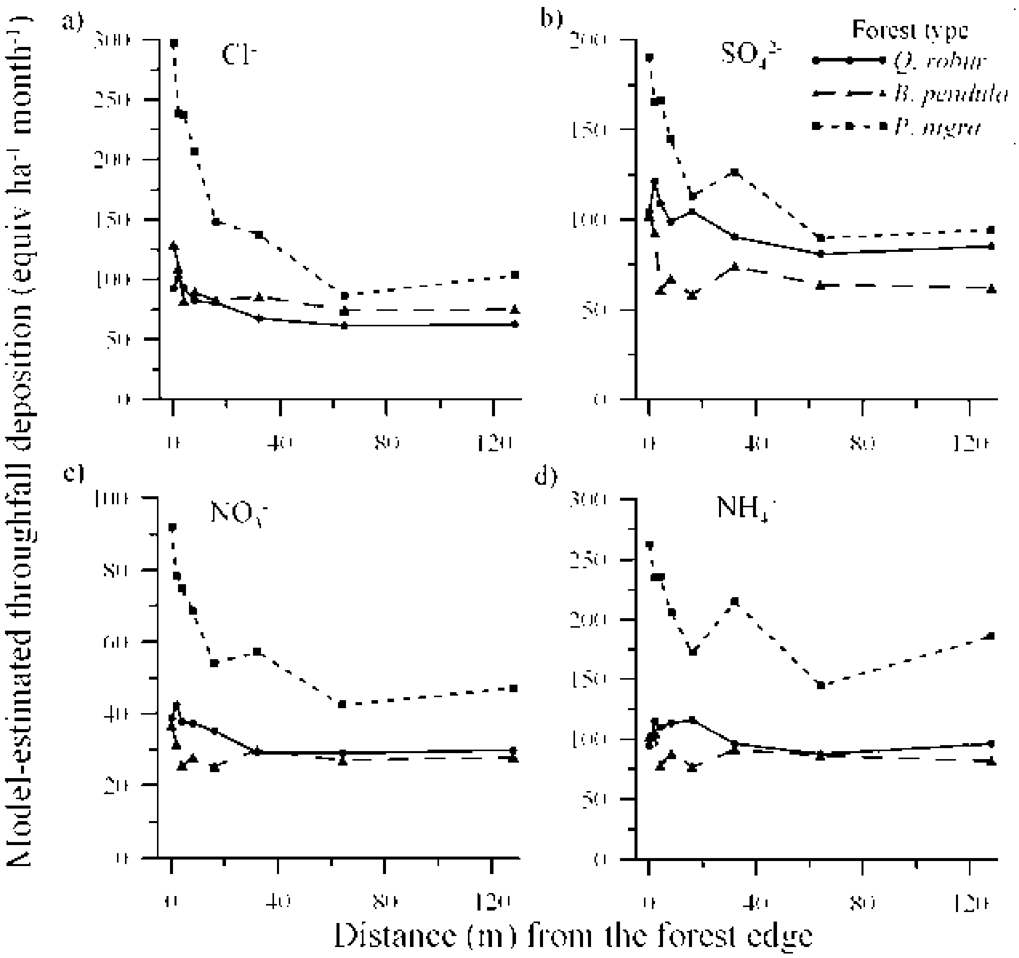

Fig. 3.3: The model-estimated means of monthly throughfall deposition (equiv ha^{-1} $month^{-1}$) of (a) Cl^-, (b) SO_4^{2-}, (c) NO_3^-, and (d) NH_4^+ for the three dominant tree species (forest type) as a function of distance (m) to the forest edge

For throughfall water volume, little or no edge effect was detected, and so the IFEE factor, when calculated, was close to one (Table 3.4). In all stands, the highest IFEE factors occurred for throughfall deposition fluxes of Cl^-. Within each stand, NO_3^-, SO_4^{2-}, and NH_4^+ throughfall deposition showed similar IFEE factors. Furthermore, similar IFEE values were found in the birch and the oak stands for all ions considered. The pine stands, however, displayed considerably higher IFEE factors for all ions than did the birch and oak stands. According to the Mann-Whitney U test, the IFEE factors in the pine stands differed significantly from those in birch ($p = 0.002$) and oak stands ($p = 0.002$), while the IFEE factors of the birch and the oak stands were not significantly different ($p = 0.180$). The IFEE factors of N and N+S throughfall deposition were, on average, 1.3 times higher in the pine stands than in the oak and birch stands. These findings were confirmed by cluster analysis of the IFEE factors for NH_4^+, NO_3^-, and SO_4^{2-}. A clear distinction emerged between the pedunculate oak and silver birch stands and the pine stands (Fig. 3.4). The difference between the pine stands on the one hand, and the birch and oak stands on the other, was much greater than the differences between stands of the same forest type in different regions. The edge effects in the oak and birch stands gave rise to extra N throughfall deposition that was, on average, 90 % lower than the extra throughfall deposition the pine stands received. The extra N+S throughfall deposition was, on average, 85 % lower in the oak and birch stands than in the pine stands.

Table 3.4: Integrated forest edge enhancement (IFEE) factor calculated for throughfall water and ion deposition, and for potentially acidifying (N+S) and inorganic nitrogen (N) throughfall deposition. -: no edge effect was detected with the method of Beier and Gundersen (1989) and therefore no IFEE factor calculated.

Site code	Water	Cl^-	SO_4^{2-}	NO_3^-	NH_4^+	N+S	N
Qr1	0.99	1.22	1.02	1.07	1.03	1.03	1.02
Qr2	1.01	1.12	1.08	1.12	1.08	1.08	1.08
Bp1	-	1.18	1.18	1.10	1.14	1.14	1.07
Bp2	-	1.11	1.11	1.07	1.06	1.08	1.06
Pn1	1.00	1.95	1.45	1.56	1.40	1.45	1.45
Pn2	-	1.62	1.40	1.31	1.26	1.31	1.26

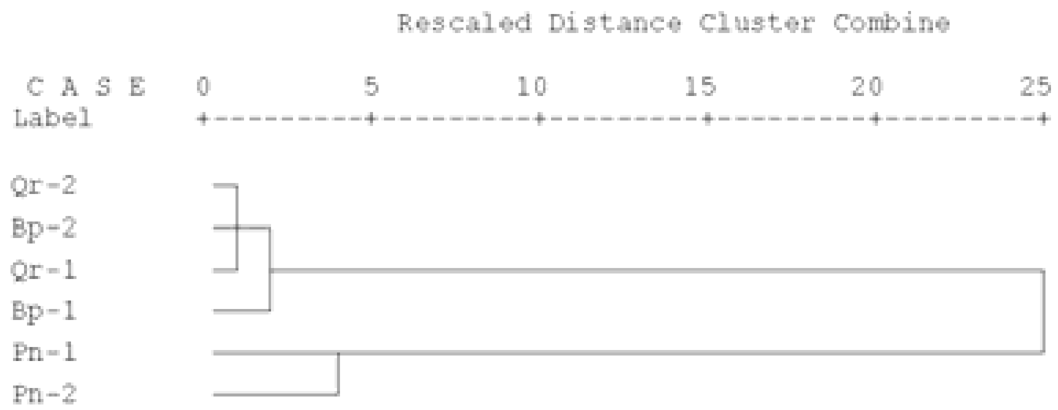

Fig. 3.4: Dendrogram of the cluster analysis based on the IFEE factors for the annual throughfall deposition of SO_4^{2-}, NO_3^-, and NH_4^+ of the six sites. For an explanation of site abbreviations, see Fig. 3.1 and Table 3.1.

3.5 Discussion

3.5.1 Throughfall water volume

Edge effects on throughfall water volume were either not observed or were negligible in all forest types in the study, based on the IFEE factors as well as on the repeated-measures ANOVA. The absence of significant edge effects on throughfall water volume agrees with the findings of Hasselrot and Grennfelt (1987) in a Scots pine stand (*Pinus sylvestris* L.), Draaijers et al. (1988) in douglas fir stands (*Pseudotsuga menziessii* Franco), and Devlaeminck et al. (2005) in a beech stand (*Fagus sylvatica* L.). Wuyts et al. (2008a, chapter 2) found only very small edge effects on throughfall water volume in silver birch and Corsican pine edges: the IFEE factors were as low as in this study. In addition, Herbst et al. (2007) found a slightly increased throughfall water volume at the 1-4 m outer edge zone of a windward edge, a result of more rain penetrating the edge from the side. Klaassen et al. (1996) observed that distance to the forest edge had little effect on throughfall water volume, although evaporation was predicted to be fetch dependent based on simulations, due to enhanced wind speed close to the edge. The difference in results between the simulations and the measurements was attributed to a lower water storage capacity at the edge due to higher wind speed, which counterbalanced the higher evaporation rate.

The effect of region on throughfall water volume was just below significance and, surprisingly, the throughfall water volume was not affected by forest type. Only the factor of time significantly influenced the throughfall water volume, and it interacted with region, forest type, and distance to the forest edge. The throughfall fraction of open-field precipitation, in contrast, was significantly affected by region and forest type. According to Neary and Gizyn (1994), Van Ek and Draaijers (1994), Robertson et al. (2000), De Schrijver et al. (2004), Oulehle and Hruska (2005), and Herrmann et al. (2006), interception loss is higher, and the proportion of incident precipitation that reaches the forest floor as throughfall is therefore lower for coniferous canopies than for deciduous ones. The throughfall fraction in the Bp2 stand was high (88 %), probably as a result of decreased LAI (or increased canopy openness) at the end of the transect, in the forest interior.

3.5.2 Throughfall deposition flux

Throughfall deposition fluxes of all studied ions (Cl^-, SO_4^{2-}, NO_3^-, and NH_4^+) were found to be significantly affected by time, forest type, and distance to the forest edge. In addition, throughfall deposition fluxes of Cl^-, NH_4^+, and SO_4^{2-} were significantly affected by the factor of region.

3.5.2.1 Effect of forest type on throughfall deposition flux

Throughfall deposition fluxes of inorganic N, SO_4^{2-}, and Cl^- were significantly affected by the forest type: throughfall deposition was higher under Corsican/Austrian pine than under silver birch and pedunculate oak. This finding confirms previous reports that coniferous forest types receive higher throughfall quantities of N and S than deciduous forest types at the same site and under the same climatological conditions (Van Ek and Draaijers 1994; Robertson et al. 2000; Augusto et al. 2002; Rothe et al. 2002; De Schrijver et al. 2004, 2007b; Oulehle and Hruska 2005; Herrmann et al. 2006). Important factors causing this difference are the following: (i) differences in vegetation structure, such as the generally lower height and LAI in deciduous stands (Augusto et al. 2002) and a higher stand density, crown density, and volume in coniferous stands (Cole and Rapp 1981); (ii) the absence of leaves in deciduous stands during the dormant season (Draaijers 1993; Houle et al. 1999); and (iii) the higher efficiency with which small needle-like structures collect particles and cloud droplets compared to larger leaf-like structures (Erisman and Draaijers 2003).

The significant interaction between forest type and region suggests that the effect of forest type on throughfall deposition depends on the local characteristics of the region under consideration. In both regions, throughfall deposition in the forest interior of the pine stand was higher than in the oak stand, which in turn was higher than in the birch stand. However, differences in ion throughfall deposition of SO_4^{2-}, NO_3^-, and NH_4^+ between the pine and the birch stands were the greatest in the western region. A meta-analysis by De Schrijver et al. (2007b) demonstrated that in regions with higher atmospheric NH_4^+ concentrations (as evidenced in a higher open-field deposition), greater differences between coniferous and deciduous throughfall depositions can be expected. In the present study however, the difference in mean NH_4^+ open-field deposition between the regions was negligible. It is likely that the greater differences between the forest types in the western region were the result of larger differences in stand characteristics influencing collecting efficiency, such as LAI, mean tree height, and stand density.

3.5.2.2 Effect of forest type on throughfall deposition flux in forest edges

The forest edge effect on throughfall deposition fluxes of Cl^-, NO_3^-, and SO_4^{2-} was significantly affected by the forest type; the pine stands displayed more pronounced edge effects than the pedunculate oak and silver birch stands. This was seen in the stronger enhancement (MEI) and/or greater penetration depth (DEI) in the model-estimated monthly means of throughfall deposition, as well as in the significantly higher IFEE factors in the pine stand. Wuyts et al. (2008a, chapter 2) found greater edge effects on the throughfall deposition of Cl^-, NO_3^-, SO_4^{2-}, and also of NH_4^+ in a dense Corsican pine stand than in a naturally regenerated silver birch stand. These observed differences in edge effect were also the result of both a greater depth of influence and a higher deposition enhancement at the edge. Differences in edge effect may be attributed to differences in meteorological conditions, edge aspect, and structural characteristics at the edge, such as forest type (Draaijers et al. 1988; Draaijers 1993; Weathers et al. 2001; De Schrijver et al. 2007a). As the meteorological conditions can be presumed to be alike within one region, and all edges were oriented towards the southwest, differing edge effects must be the result of differing forest types. Draaijers (1993) found that the enhancement of throughfall deposition at the edge depended strongly and positively on LAI for Na^+, Cl^-, and Mg^{2+}, and on stem density and silhouette area density (i.e., the frontal crown area per 100 m²) for NO_3^-, SO_4^{2-}, and NH_4^+. In a numerical modelling study by Pahl (2000), stand height had a significant influence on the magnitude of the edge effect: for a fixed distance to the forest edge, higher deposition velocity enhancement factors were calculated for higher stand heights. In the present study, stem density was higher, and winter canopy openness was lower in pine stands than in oak stands. Thus, the greater edge effects in the pine stands are in line with the findings of Draaijers (1993). Since no significant difference in edge effect, as measured by the IFEE factor, was found between the oak and the birch stands, we hypothesize that the effect of higher stem densities in the birch stands may have counterbalanced both the effect of higher canopy openness values during summer and the lower mean tree height in the birch stands compared to the oak stands. When comparing the birch and the pine stands, it appears that the evergreen character of the coniferous forest types, and the great increase of canopy openness in deciduous forest types in the winter, may also be very important factors. However, it is clear that a complex of factors determines the magnitude of the edge effect and that our data set is too small to specify the most important factors. Thus, we emphasize that further research is needed to identify the processes involved in the effects of forest type on edge patterns.

The greater edge effects on potentially acidifying and eutrophying throughfall deposition in pine stands compared to oak and birch stands, in both the present study and that of Wuyts et al. (2008a, chapter 2), implies that differences in throughfall deposition input between pine and pedunculate oak/silver birch stands are underestimated when only the throughfall deposition in the forest interior is considered (as in e.g., De Schrijver et al. 2004; Herrmann et al. 2006). Conversion of pine plantations to oak or birch forests would decrease the extra N and N+S throughfall deposition caused by edge effects by 90 % and 85 %, respectively. Consequently, the more pronounced edge effects in the pine stands, together with the higher level of throughfall deposition in the forest interior, strengthens the idea that conversion of pine plantations into oak or birch forests can reduce potentially acidifying and eutrophying deposition and its associated negative effects. At our study sites, we did not find a significant difference in pH (KCl) between the forest types (Table 3.1): (i) pH values are low at all sites as a legacy from former land use as described above and (ii) these soils can be situated in the aluminium buffer range, which implies that no strong change in pH will occur with increasing or decreasing acid load (Ulrich 1991).

The effect of distance to the forest edge on throughfall deposition did not depend on region for any of the ions considered. Thus, irrespective of region, edge effects on throughfall deposition are similar for the same forest type. The magnitude of edge effects, however, does depend on meteorological conditions such as wind speed and wind direction (Draaijers 1993). Mean wind speed in the western region was significantly higher than in the eastern region, but the stands in the east were more frequently subject to winds blowing into the forest edge (i.e., south to west wind directions), which may have counteracted the effect of higher wind speeds in the western region.

Edge effects on throughfall deposition of Cl^- were greater - as measured by a higher IFEE factor - than edge effects involving the potentially acidifying and eutrophying ions NO_3^-, NH_4^+, and SO_4^{2-}. These latter three had relatively similar edge effects, as seen in there comparable IFEE factors. Beier and Gundersen (1989) also found the most pronounced edge effect for substances deposited as coarse particles (Cl^- and Na^+) in a Norway spruce [*Picea abies* (L.) Karst.]. Draaijers (1993) found a greater deposition enhancement at the edge for larger aerosols (Na^+, Cl^-, and Mg^{2+}) than for smaller aerosols and gases (SO_4^{2-}, NO_3^-, NH_4^+, NH_3, NO_2, and SO_2), while in chapter 2 (Wuyts et al. 2008a), we found a higher enhancement and a higher DEI for Cl^- than for SO_4^{2-}, NO_3^-, and NH_4^+ in a birch and a pine stand.

The edge effects on throughfall deposition varied significantly throughout the year for Cl^-, but not for NO_3^-, SO_4^{2-}, and NH_4^+. This is in agreement with the results of chapter 2 (Wuyts et al. 2008a), who found greater differences between the growing and dormant seasons for Cl^-, Na^+, K^+, Ca^{2+}, and Mg^{2+} throughfall deposition than for NO_3^-, NH_4^+, or SO_4^{2-} in edges of Corsican pine and silver birch. It is likely that deposition of coarse particles containing Na^+ and Cl^- depends more on varying meteorological conditions than does deposition of finer particles and gases with NO_3^-, NH_4^+, SO_4^{2-}, NH_3, NO_2, and SO_2.

3.6 Conclusions

Pine stands displayed greater edge effects on throughfall deposition of NO_3^- and SO_4^{2-} than did deciduous oak and birch stands, because of higher throughfall enhancement at the edge and/or a greater edge distance. The less substantial edge effects on potentially acidifying and eutrophying throughfall deposition in the oak and birch stands of our study suggest that edge effects give rise to an even greater difference in throughfall deposition between pine and pedunculate oak/silver birch stands than previously suggested by research on forest interior throughfall deposition. Our findings therefore strengthen the idea that conversion of pine plantations into oak, birch, or mixed deciduous forests will reduce the input of eutrophying and acidifying ions in both the forest interior and the forest edge. This will consequently reduce soil acidification, nitrogen saturation, eutrophication of ground- and surface water, and loss of biodiversity.

4 The impact of forest edge structure on longitudinal patterns of deposition, wind speed, and turbulence

After: Wuyts, K., Verheyen, K., De Schrijver, A., Cornelis, W.M., Gabriels, D., 2008. The impact of forest edge structure on longitudinal patterns of deposition, wind speed, and turbulence. Atmospheric Environment 42, 8651-8660.

4.1 Abstract

The impact of forest edge structure on edge patterns of wind speed, turbulence, and atmospheric deposition was studied by means of a model forest in a wind tunnel. Tests were conducted with eight structure configurations, encompassing combinations of stem densities, crown depths, and edge transitions (steep or gradual edge). Mean wind speed and its standard deviation (as a measure for turbulence) were determined within and at the top of the canopy; deposition was simulated using Cl^- aerosols. Edge patterns of wind speed, turbulence, and deposition were closely related and were significantly affected by stem density and, particularly, by edge transition. In the dense forests, the edge effect on deposition extended less deeply into the forest than in the sparse forests, so the deposition in the forest edge zone was lowered with 40 %. Gradual edges were able to limit the level by which deposition is enhanced at the edge in comparison with the forest interior deposition, and consequently, they reduced the deposition in the forest edge zone with 66 %. Even when the deposition on the trees of the gradually ascending vegetation in front of the forest edge was taken into account, gradual edges were still advantageous in comparison with steep edges. A lower crown depth decreased the enhancement of deposition at the edge relative to the interior, but only at steep edges. We conclude that an adjusted layout of forest edges should be able to mitigate the edge effects on atmospheric deposition, through reducing the deposition enhancement at the edge or the penetration depth of the edge effect.

4.2 Introduction

Due to human-induced activities such as timber harvesting, agricultural expansion, and urbanisation, the extent of forest fragmentation has been increasing in many landscapes around the world (Harper et al. 2005). Consequently, forest edges are becoming a dominant feature of the landscape matrix (Draaijers et al. 1994; Devlaeminck et al. 2005; Harper et al. 2005). Edges affect numerous abiotic and biotic factors in the remnant forest patches, including the inflow of chemical compounds from the atmosphere or via drift [for an overview, see Murcia (1995) and Harper et al. (2005)]. Increase of atmospheric deposition in forest edges has been abundantly pointed out, mostly by throughfall deposition measurements [for a review, see De Schrijver et al. (2007a)]. Potentially acidifying and eutrophying depositions of nitrate, ammonium, and sulfate are increased at the front of the edge up to a fourfold, and this so-called edge effect extends to a distance of up to five times the stand height (e.g., Beier and Gundersen 1989; Devlaeminck et al. 2005; Draaijers et al. 1994; Wuyts et al. 2008a, 2008b: chapter 2 and 3). Edge effects on atmospheric deposition have also been demonstrated by a limited number of air concentration, modeling, and wind tunnel studies (e.g., Wiman and Ågren 1985; Wiman and Lannefors 1985; Ould-Dada et al. 2002).

The edge effect on atmospheric deposition occurring in homogeneous forest edges can be attributed to (i) altered dry deposition and/or canopy exchange at the edge induced by an edge gradient in soil, precipitation, and microclimate characteristics (Matlack 1994; Chen et al. 1995; Marchand and Houle 2006; Herbst et al. 2007) and/or (ii) enhanced dry deposition as a result of aerodynamic processes such as local advection and enhanced turbulent exchange at the edge (Draaijers et al. 1994; Veen et al. 1996; Ould-Dada et al. 2002). Forest edges as abrupt surface changes drastically disrupt air flow: mean wind speed and turbulence at canopy level decline from the edge to the interior, to values lower than upwind of the forest (Meroney 1970; Chen et al. 1995; Irvine et al. 1997; Morse et al. 2002). Canopy wind speed and turbulence control the dry deposition process (Ruijgrok et al. 1996; Beckett et al. 2000; Smith et al. 2000) and it is therefore likely that variations in wind speed and turbulence along the forest edge influence patterns of dry deposition. Draaijers et al. (1994) suggested that in-canopy wind speed and turbulence determine the difference between forest edge and forest interior deposition.

As forest edges are increasingly important elements in our landscapes and a drastic decrease in deposition load is not expected shortly, the layout of forest edges is of great relevance to protect forest ecosystems from further acidification and eutrophication. However, there is a lack of studies on the edge structure characteristics that are likely to influence the edge effects on deposition, such as tree height, stem density, and leaf area index (LAI), in exception of field experiments by Draaijers et al. (1994) and numerical simulations by Wiman and Ågren (1985) and Pahl (2000). And although the influence of a gradual edge (a gradual increase of height across the edge) on wind damage is subject to investigation (Agster and Ruck 2003; Yang et al. 2006; Dupont and Brunet 2008b), the impact of such a gradual vegetation on edge patterns of deposition has, to our knowledge, not been explored yet.

Therefore, the aims of our study were (i) to assess the impact of edge structure characteristics that can easily be altered by edge management on patterns of dry deposition induced by an aerodynamic edge effect and (ii) to determine their influence on wind speed and turbulence to understand the underlying processes. The study's hypothesis was that alteration of stem density, crown depth, or edge transition can significantly reduce wind speed and turbulence in the forest edge and consequently mitigate the forest edge effect on dry deposition. Tests were conducted with model forests in a wind tunnel rather than with full-scale forests, which enabled us to:

(i) investigate edge effects resulting from aerodynamic processes only, irrespective of edge effects caused by an edge gradient in soil, precipitation, or microclimate characteristics,

(ii) avoid the high level of variation associated with throughfall deposition measurements (i.e., point measurements) in field studies, by measuring the deposition on the entire forest stand, and

(iii) study the influence of multiple factors individually under controlled circumstances and on a small time scale, which would be unfeasible, time-consuming, and expensive if performed on full-scale forests.

4.3 Materials and methods

4.3.1 Wind tunnel and model trees

The study was performed in a closed-circuit low-speed wind tunnel at the ICE, Ghent University, Belgium [full description: Dierickx et al. (2003) and Cornelis et al. (2004)]. The tunnel has a rectangular-shaped working area, 12 m x 1.20 m, and the ceiling was adjusted to a height of 2.50 m. Three wooden spires (0.75 m high) and three rows of twelve wooden cubes (side length 0.04 m) were placed on the tunnel floor, leeward of the spires, over a length of 4.80 m starting from the entrance of the test section to create large-scale turbulence and to increase the boundary-layer depth to 0.60-0.65 m (Cornelis et al. 2004). Consequently, the experiments were carried out in a rough surface turbulent boundary layer. All measurements were performed at a free-stream wind speed of 7 m s^{-1}.

The scale model trees, modeled to the shape of *Picea abies* (L.) Karst. trees and, on average, 0.30 m high, consisted of one iron spiral (representing the ‘stem’) with attached shorter spirals (‘branches’) and polyethylene ribbon, densely weaved into the spirals and cut into every 1 mm (‘the needles’). The trees’ diameter at the crown base measured 0.15-0.20 m.

4.3.2 Experimental setup

The model trees of the forest stand were installed over a length of 2.00 m, starting at 8.00 m from the entrance of the test section, and along the tunnel’s entire width (Fig. 4.1). The stems were inserted into holes drilled into a wooden board on the tunnel floor, along its entire length. Before installation, the model trees were rinsed three times with distilled water and oven-dried at 40°C for four hours. Tests were conducted with eight stand structure configurations, encompassing a combination of:

(i) stem density: one tree per 0.15 m by 0.15 m and 0.10 m by 0.10 m, i.e., 44 and 100 trees m^{-2}, respectively;

(ii) crown depth (CD; Fig. 4.1): 0.18 and 0.24 m, i.e., 0.6 and 0.8 times the tree height, respectively;

(iii) edge transition: tests were performed with a steep and a gradual transition, i.e., in absence and in presence of a gradually ascending edge vegetation placed before the edge of the forest (i.e., the first tree row of the model forest). This gradual edge vegetation consisted of three rows of smaller trees, resulting in an inclination angle of 37° with the tunnel floor (Fig. 4.1 and Fig. 4.2).

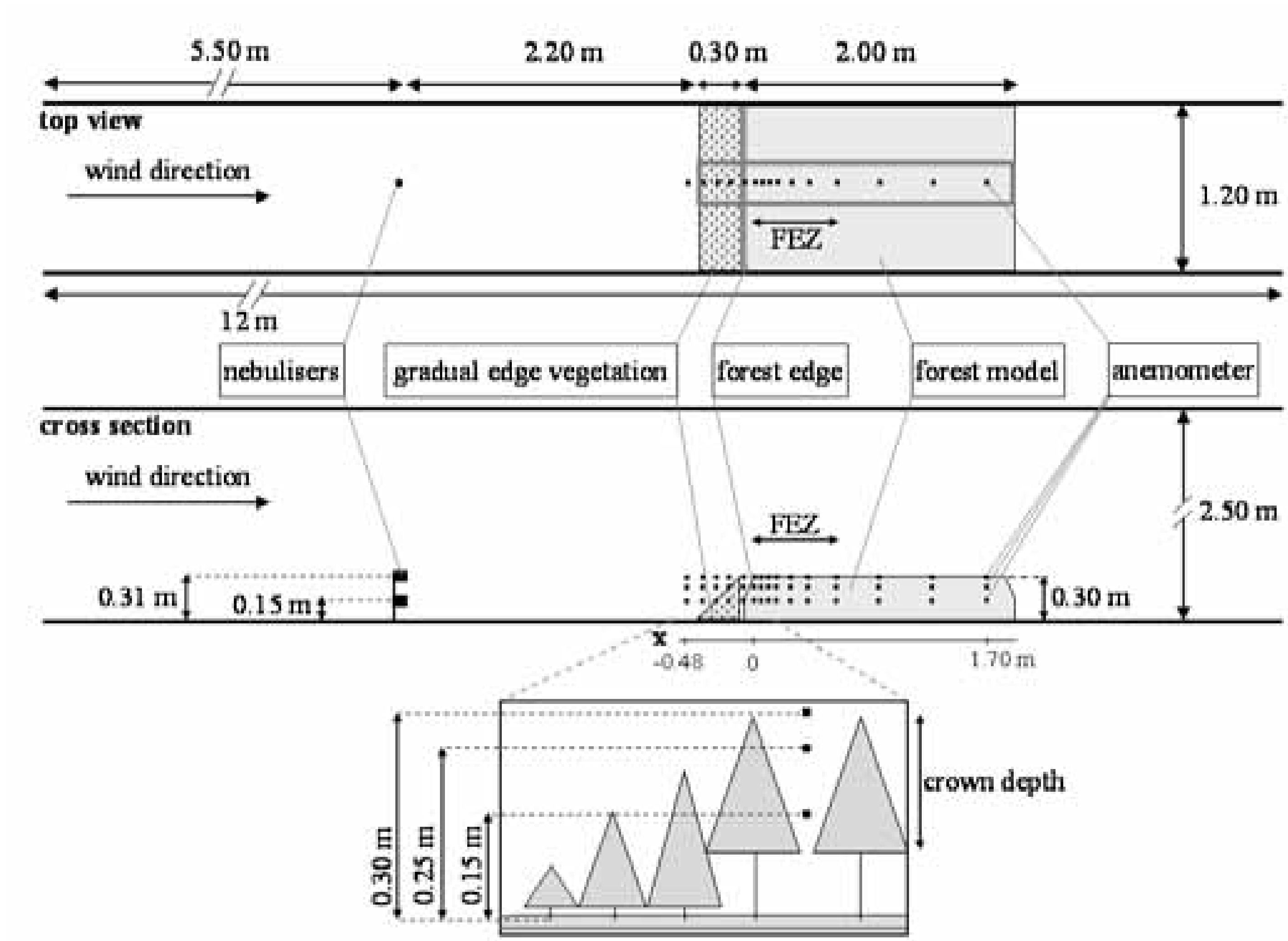

Fig. 4.1: Top view and cross section of the test section in the wind tunnel with location of the model forest, the gradual edge vegetation, the anemometers, and the nebulisers. The grey rectangle shows the two parallel tree rows sampled for Cl^- deposition measurement; the forest edge zone as delineated in this study is indicated by 'FEZ'.

Fig. 4.2: The gradual transition consisted of three rows of model trees with increasing height towards the edge of the model forest. In this photograph, the crown depth of the model trees of the stand itself was 0.24 m and the trees were set in a stem density of 100 ha^{-1}.

Wind speed (m s^{-1}) was measured with thermal anemometers connected to a digital reading unit (Testo GmbH, Lenzkirch; 1 Hz). The probes were placed at a height z of 0.15, 0.25, and 0.30 m above the tunnel floor and at 15 distances upwind and downwind of the first tree row of the model forest (Fig. 4.1): at x = -0.48, -0.38, -0.28, -0.18, -0.08, 0 (i.e., at the stems of the first tree row), 0.05, 0.10, 0.15, 0.25, 0.40, 0.60, 0.90, 1.30, and 1.70 m from the forest edge. For every anemometer position, 200 wind speed measurements were recorded per forest configuration. Furthermore, for each stand configuration, a wind speed profile was measured above the canopy at 0.010-0.050 m height intervals (until z = 0.35-0.60 m), at x = 1.70 m. Measurements within the canopy were made in the space between the trees to avoid direct contact with them. Prior to the experiment, the anemometers were intercompared in the wind tunnel in absence of a model forest. The wind speed measurements were started at least one minute after the turbine of the tunnel was initiated.

Subsequent to the wind speed measurements, the anemometers were removed and aerosol was produced during 120 minutes from a 25 % NaCl solution (i.e., 2.5 g l^{-1} or 146 mol l^{-1}) by two compressors (PARI TurboBoy, PARI GmbH). These compressors were located outside the test section of the wind tunnel and were connected to two aerosol generators (PARI LC PLUS nebuliser, PARI GmbH) within the test section. With this combination of compressor and nebuliser, droplets are obtained with a mass median aerodynamic diameter (MMAD) of about

3 µm (de Boer et al. 2003). During nebulisation, a shift in droplet size distribution occurs (de Boer et al. 2003; Steckel and Eskandar 2003), so to avoid a shift in deposition velocity (Lin et al. 1994), the same reservoir refilling protocol was applied throughout all configurations. The outlet of the two generators was positioned at z = 0.15 and 0.31 m and x = -2.50 m, oriented towards the two centre tree rows. At the end of the aerosol experiment, all trees of the two centre tree rows (the grey box in Fig. 4.1), were washed separately with 0.200 l distilled water in an inert polyethylene bag. All samples were analyzed for Cl^- concentration through ion chromatography (ICS-90, Dionex). Pre-tests were conducted to examine (i) the inertness of the trees with regard to Cl^- and (ii) the Cl^- contamination through the applied washing and drying procedures. Unexposed trees yielded a Cl^- concentration below or around detection limit (0.05 mg l^{-1}). Trees were manipulated with latex gloves to avoid Cl^- contamination.

4.3.3 Data analysis

Standard deviation of streamwise wind speed (σ_u, m s^{-1}) was used as an indicator of streamwise turbulence (Irvine et al. 1997; Morse et al. 2002). Based on the wind speed profile within the boundary layer, values for the roughness length z_0 (m) and the friction velocity u* (m s^{-1}) were derived from the intercept and the slope of the regression equation for wind speed against ln(z-d), where d is the displacement height (m). This d can be estimated as 0.65-0.70 x tree height = 0.195-0.210 m (Irvine et al. 1997; Kinnersley et al. 1994); based on the method described by Stull (1993), we chose d = 0.195 m. The z_0 was also calculated according to the formula of Jarvis et al. (1976): 0.075 x tree height. The magnitude of roughness transition can be quantified by z_0 of the upwind surface / z_0 of the downwind surface (Kaimal and Finnigan, 1994).

The Cl^- deposition per tree (mg) and per ground area (mg dm^{-2}) was determined. The depth of edge influence (DEI, m), i.e., the penetration depth of the edge effect on deposition, was calculated as described by Beier and Gundersen (1989). The magnitude of edge influence (MEI), i.e., the level of enhancement of the deposition at the front edge relative to the interior, was determined as the ratio of the deposition at x = 0 m to the mean deposition at the end of the transect. Additionally, the Cl^- deposition per tree was summed for one entire tree row, inclusive of the gradually ascending vegetation, yielding the deposition sum. Furthermore, the deposition (mg) was summed (i) in each forest between x = 0 and 0.60 m and (ii) on each gradual edge vegetation.

The effects of stand density, crown depth, and edge transition on u, σ_u, MEI, and total Cl^- deposition were tested by paired samples t-tests. Because we are interested in the edge effects, the tests were performed upon the data in the edge zone, where deposition is significantly enhanced. We delineated this edge zone between x = 0 m and 0.60 m, with seven measuring points, since the mean DEI of the Cl^- deposition equalled 0.54 m (see Table 4.2). Furthermore, a repeated-measures ANOVA was applied on the Cl^- deposition data (mg dm^{-2}) in the forest, with 'distance to the forest edge' as within-subjects factor and 'stand density', 'crown depth', and 'edge transition' as between-subjects factor. To avoid the assumption of sphericity, the multivariate approach was used to test for within-subject effects (Pillai's trace test statistic) (O'Brien and Kaiser 1985). Finally, linear regressions were performed between the ln-transformed Cl^- deposition (mg per tree) on the one hand and u and σ_u at the three heights on the other, to gain insight into the factors influencing Cl^- deposition.

4.4 Results

Above the canopy, wind speed decreased and turbulence intensity increased with decreasing height; within the canopy, turbulence intensity reached its maximum (Fig. 4.3). Within the trunk area, wind speed tended to increase again and turbulence intensity decreased. At the end of the transect, wind speed in the upper 0.15 m of the canopy was higher (i) in the sparse forests and (ii) in the case of a 0.18 m crown depth, indicating that, within this range of LAI, wind speed decreases with increasing LAI. At this distance from the edge, there was no effect of edge transition on the wind speed in the canopy. From the wind speed profiles, a mean friction velocity u* of 172 cm s^{-1} and a roughness length z_0 of 8.9 cm were derived. Following the formula of Jarvis et al. (1976), a roughness length of 2.25 cm was calculated. Based on this z_0, the magnitude of roughness transition equalled 0.074 (1.66 mm / 22.5 mm).

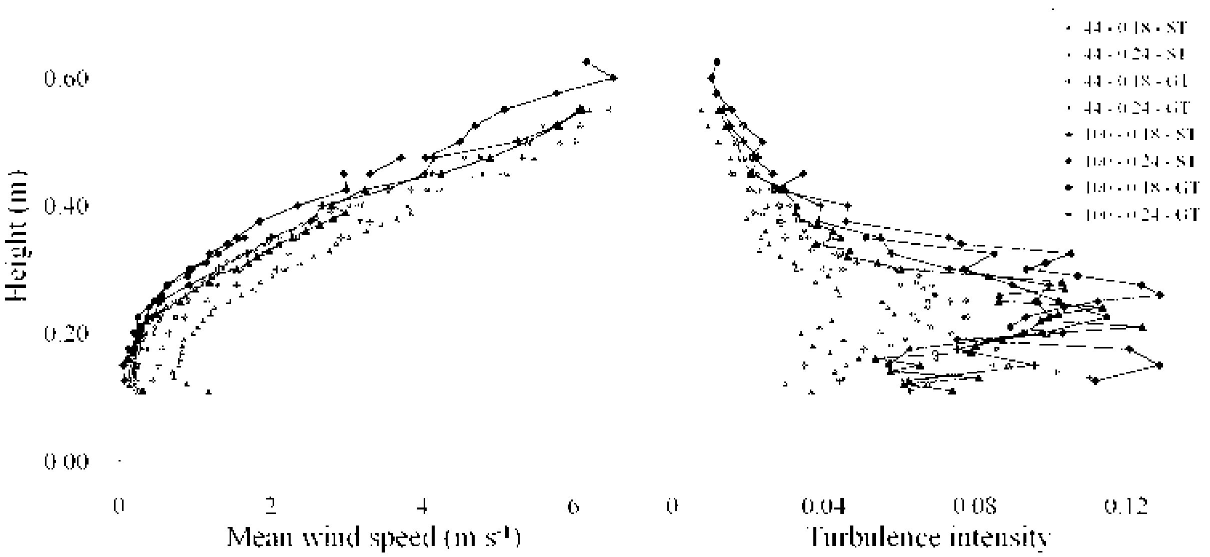

Fig. 4.3: Wind speed (m s^{-1}) and turbulence intensity (-) profile as measured at 1.70 m behind the edge of the forest. The different stand configurations are indicated as 'stem density (44 or 100 trees m^{-2}) - crown depth (0.18 or 0.24 m) - edge transition (steep transition ST or gradual transition GT)'. The mean tree height is indicated by the dotted line.

4.4.1 Streamwise wind speed and turbulence

The mean wind speed and turbulence along the transect in each of eight configurations are displayed in Fig. 4.4, by the black and grey lines, respectively. In the forest edge zone, u at all heights and σu at $z = 0.15$ and 0.25 m were significantly higher in the sparse forests than in the dense forest; Table 4.1; all $p < 0.01$). The effect of crown depth was only significant for the u at $z = 0.30$ m ($p < 0.001$). With a gradual transition, both the u and the σu were significantly lower at $z = 0.15$ m ($p < 0.001$ and $p = 0.001$, respectively), but the turbulence was significantly higher at $z = 0.25$ and 0.30 m than with a steep transition ($p = 0.006$ and 0.026, respectively).

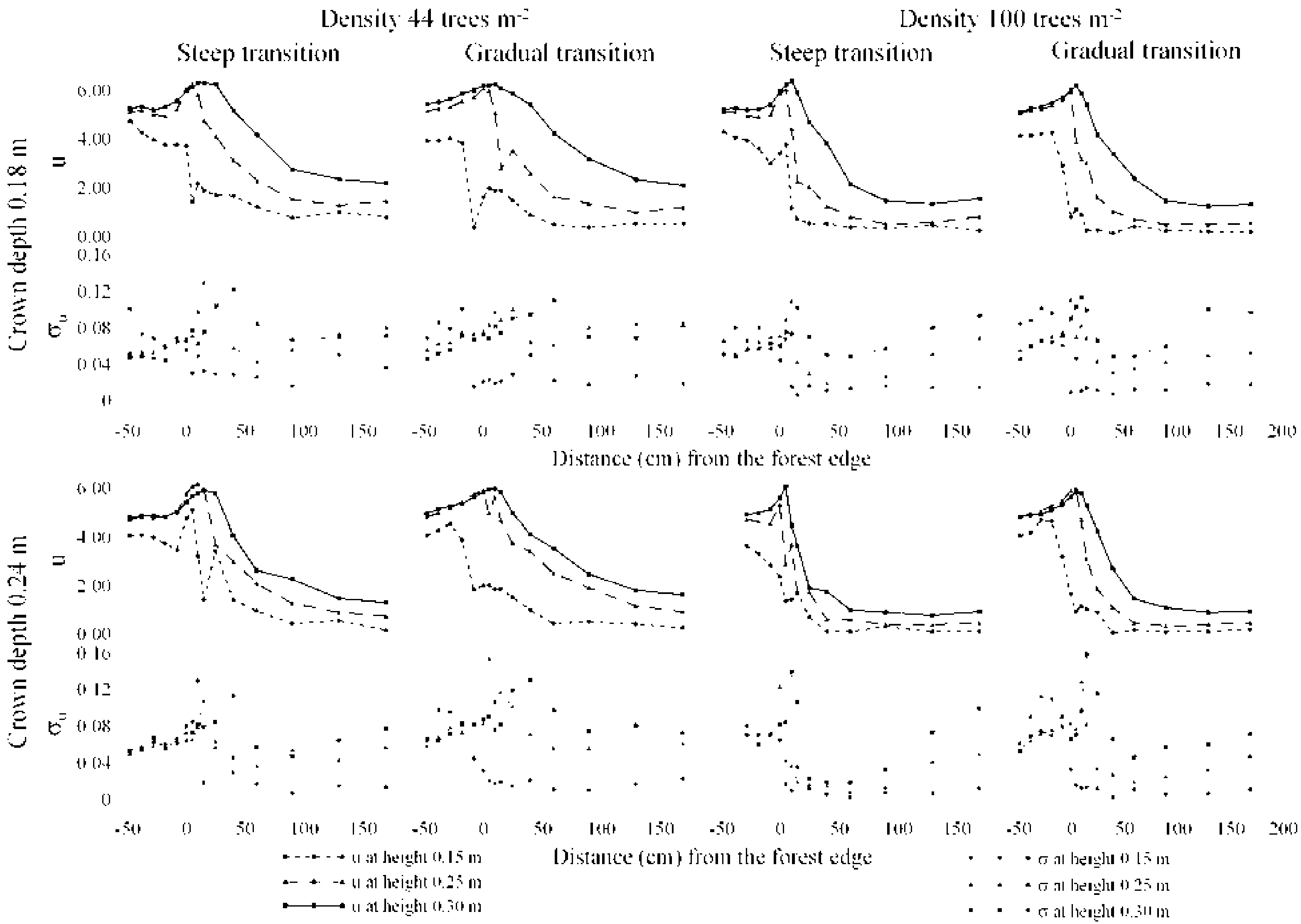

Fig. 4.4: Wind speed (m s^{-1}), indicated by the black lines, and turbulence (m s^{-1}), indicated by the grey lines, upwind and downwind of the edge of the forest, indicated by the vertical grey dotted line.

Table 4.1: Mean wind speed (u, m s^{-1}) and turbulence (σu, m s^{-1}) as the mean value of the first 0.60 m of the forest of the eight treatments and at the measuring heights 0.15, 0.25, and 0.30 m (CD: crown depth)

	Density 44 trees m^{-2}						Density 100 trees m^{-2}					
	Steep transition			Gradual transition			Steep transition			Gradual transition		
Height (m)	0.15	0.25	0.30	0.15	0.25	0.30	0.15	0.25	0.30	0.15	0.25	0.30
u (m s^{-1})												
CD 0.18 m	1.96	4.61	5.73	1.45	3.93	5.73	1.48	3.22	5.00	0.54	2.74	4.75
CD 0.24 m	2.89	4.68	5.03	1.52	4.41	5.18	1.11	2.36	3.49	0.83	3.30	4.42
σ (m s^{-1})												
CD 0.18 m	0.04	0.08	0.08	0.03	0.08	0.08	0.02	0.05	0.07	0.02	0.06	0.08
CD 0.24 m	0.06	0.07	0.08	0.02	0.10	0.10	0.02	0.04	0.07	0.01	0.06	0.09

4.4.2 Cl^- deposition

In each configuration, the Cl^- deposition was enhanced at the edge of the forest (Fig. 4.5). The 'distance to the edge' and 'edge transition' significantly influenced the Cl^- deposition on the forest ($p = 0.002$ and < 0.001); the effect of 'stem density' was only marginally significant ($p = 0.078$). The effect of 'distance to the edge' interacted significantly with 'edge transition' and marginally with 'stem density' ($p = 0.016$ and 0.065).

The DEI was similar for both crown depths, but edge effects extended further in the sparse forests than in dense ones; Table 4.2). Edge transition had only small and non-univocal effects on the DEI. The MEI values differed significantly between steep and gradual transitions ($p = 0.028$; 1.1 - 2.8 times lower in the case of a gradual edge), while differences between densities or crown depths were not significant ($p = 0.179$ and 0.232, respectively; Table 4.2). Significantly higher deposition on the forest and edge vegetation was found in case of (i) a steep edge ($p = 0.009$) and (ii) a sparse forest ($p = 0.017$), but there was no effect of crown depth ($p = 0.674$; Table 4.2). The deposition in the forest edge zone was, on average, 40 % lower in the dense forests than the sparse forests, 66 % lower with gradual transition than with a steep transition, and 6 % lower for a 0.18 m crown depth than for one of 0.24 m.

The deposition on the gradual vegetation was, on average, 6.0 mg Cl^-. With a steep transition, the deposition in the forest edge zone was, on average, 22.3 mg Cl^- higher than with a gradual transition. Consequently, the presence of a gradual transition generated a decrease in deposition in the forest edge zone which was higher than the deposition on the gradual vegetation (on average, with a factor 8.30 and 1.56 in the sparse and the dense forests, respectively).

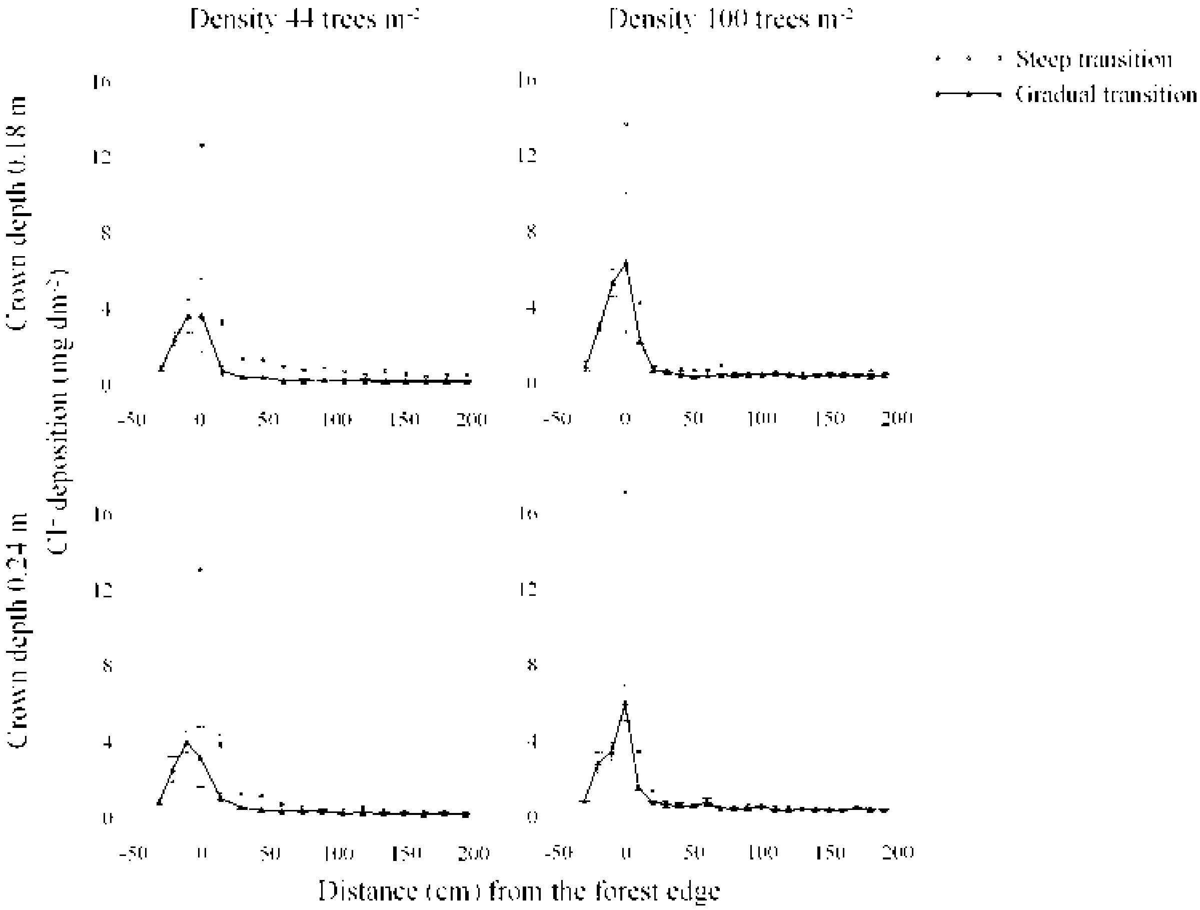

Fig. 4.5: Cl^- deposition (mg dm^{-2} ground area, as a mean of two trees per distance) on the forest and the gradually ascending forest edge as a function of distance to the forest edge. Error bars indicate the standard deviation of the Cl^- deposition.

Table 4.2: The depth of edge influence (DEI, m), the magnitude of edge influence (MEI) (mean of two trees ± standard deviation SD) of the edge effect on the Cl- deposition, and the mean deposition sum (mg) along one tree row of the forest and, in case of a gradual transition, including the gradually ascending forest edge (mean of two tree rows ± SD). CD: crown depth

CD	Density 44 trees m^{-2}						Density 100 trees m^{-2}					
	Steep transition			Gradual transition			Steep transition			Gradual transition		
	DEI	MEI	Sum	DEI	MEI	Sum	DEI	MEI	Sum	DEI	MEI	Sum
	(m)	(-)	(mg)	(m)	(-)	(mg)	(m)	(-)	(mg)	(m)	(-)	(mg)
0.18 m	0.75	26 ± 5	56 ± 13	0.60	24 ± 18	28 ± 5	0.30	25 ± 2	28 ± 3	0.50	20 ± 15	25 ± 4
0.24 m	0.75	48 ± 6	53 ± 11	0.60	17 ± 9	28 ± 4	0.40	40 ± 2	30 ± 2	0.40	17 ± 1	23 ± 2

4.4.3 Relation of deposition with streamwise wind speed and its turbulence

In general, higher wind speed and turbulence increase the amount of deposition per tree (Fig. 4.6), although the relation with turbulence was not significant at z = 0.30 m. The deposition per tree can be predicted by wind speed at z = 0.15 m ($u_{0.15}$) and 0.25 m ($u_{0.25}$) [ln(deposition) = $0.533*u_{0.15} + 0.349*u_{0.25} - 1.017$ ($R^2 = 0.946$)] and by turbulence at z = 0.15 m ($\sigma_{0.15}$) and 0.25 m ($\sigma_{0.25}$) [ln(deposition) = $44.22*\sigma_{0.15} + 13.66*\sigma_{0.25} - 1.157$ ($R^2 = 0.783$)].

4.5 Discussion

4.5.1 Appropriateness of the model forest for this research

From a distance of x = 15 - 20 times tree height h, the wind profile is considered to be fully adjusted to the forest canopy (Meroney 1970), but wind flow at halfway the tree height should already be adapted to the new surface at x = 3.6 h and wind flow within the crowns at x = 6.1 h (Hsi and Nath 1970; Irvine et al. 1997). Consequently, in our study, at x = 1.70 m (or 5.7 h), wind speed at halfway the tree height was already equilibrated to the forest surface, but wind speed at full tree height and above were not. As a result, the value derived from the wind speed profiles at 1.70 m was not correct and different from other studies and the calculated from Jarvis et al. (1976). Nonetheless, the shape of the wind speed and turbulence intensity profiles measured in our study were similar to those measured in full-scale and model forests (Meroney, 1968; Kinnersley et al. 1994; Stacey et al. 1994; Ould-Dada and Baghini 2001). Turbulence intensity within the canopy corresponds very well to the values reported by Kinnersley et al. (1994), but are slightly lower than those reported by Meroney (1968) and Ould-Dada and Baghini (2001). The magnitude of roughness transition corresponds to a transition of arable land (z_0 = 0.1 m, Stacey et al. 1994) to an 18 m high forest (z_0 = 0.075 x height = 1.35 m, Jarvis et al. 1976). When scaled up to an 18 m high full-scale forest, the stem densities would equal 123 and 278 trees ha^{-1} and the crown depths would measure 10.8 and 14.4 m. The similarity of the wind speed and turbulence intensity profiles and the magnitude of roughness transition with those of a full-scale forest indicate that our model forests enabled realistic simulations of forest edge effects.

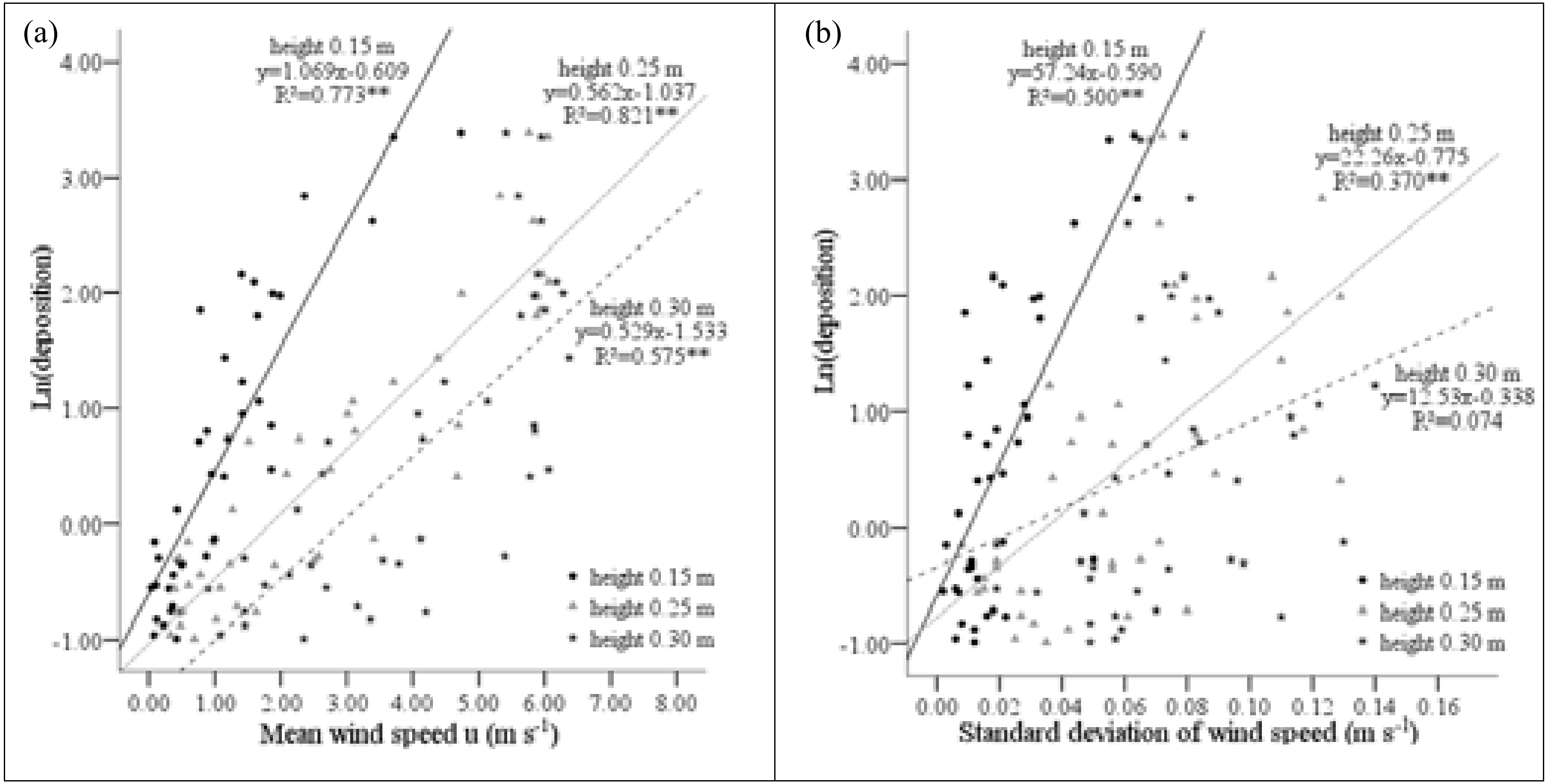

Fig. 4.6: Ln-transformed Cl^- deposition plotted against the mean wind speed (m s^{-1}) and the turbulence (m s^{-1}) at the three measuring heights along the forest canopy, with regression curves per measuring height. (**: $p < 0.01$)

4.5.2 Edge patterns of wind speed and turbulence

Wind speed and turbulence within the canopy were smaller than at the canopy top, which is a typical observation for forest canopies (Gardiner 1994; Stacey et al. 1994; Irvine et al. 1997). Wind speed at halfway the tree height (0.15 m) displayed a sharp decrease at the forest edge, while wind speed in the upper part of the canopy and at the canopy top was still increased in the first 0.10 m of the edge. Meroney (1970) and Irvine et al. (1997), however, found a decrease in wind speed along the entire tree height at the edge. Further downwind, the wind speed decreased exponentially with increasing distance from the edge, which agrees with the findings of Gash (1986), Chen et al. (1995), and Irvine et al. (1997) in full-scale forests and Meroney (1970) in a model forest.

When the wind flow entered the edge, turbulence decreased drastically in the lower part of the canopy and increased considerably in the upper part and at the top of the canopy. Irvine et al. (1997) found a slight increase of turbulence as the flow entered the edge at the canopy top, but also at half way the tree height. Like in our study, further behind the edge, the turbulence continued to increase at the canopy top, but decreased drastically within the canopy.

4.5.3 Edge patterns of Cl^- deposition

The Cl^- deposition on the forests was clearly subject to edge influence: the exponential decrease of deposition with increasing distance to the edge is in accordance with ample throughfall deposition studies [see De Schrijver et al. (2007a) for a review].

The Cl^- deposition upon a tree increased with higher wind speed and turbulence at the tree, in particular with increasing wind speed and turbulence within the canopy (at z = 0.15 and 0.25 m), which is probably the result of the non-uniform distribution of the collecting surface within the canopy (a triangle-shaped tree contour). The relations were weaker for turbulence than for wind speed; this was possibly caused by the low frequency at which the wind speed was measured. In general, turbulence is calculated based upon higher-frequency wind speed measurements, i.e., 5 Hz and more (e.g., Irvine et al. 1997). Although we did not find significantly different values when we extended the number of measurements from 200 to 500 in the pre-tests, our reported values of standard deviation may only give a rough indication of turbulence.

4.5.4 Impact of edge structure on edge patterns

4.5.4.1 Stem density

Irrespective of edge effects, the deposition per tree (thus considering the same effective collecting area) was higher in the sparse forests, which was a result of the higher wind speed within the canopy of the sparse forests, as observed by Gardiner et al. (1997). This indicates that the higher deposition per ground area in the dense forests was a result of the effect of a lower in-canopy wind speed (causing a decrease in deposition) being surpassed by the effect of a higher collecting surface per ground area (causing an increase in deposition). If the LAI would increase further, the effect of decreasing wind speed might counterbalance and even exceed the effect of increasing collecting surface, resulting in a maximum in deposition (velocity) at a certain LAI value, as is suggested by Meyers et al. (1989).

The MEI did not differ significantly between the two stem densities: in the dense forests, the deposition per ground area was higher than in the sparse forests, both at the forest edge and in the forest interior to the same extent. The edge effect on deposition extended further in the sparse forests than in the dense forests. This was caused by (i) the stronger deceleration of wind speed to a forest interior level in dense forests, which was also observed by Dupont and Brunet (2008a) in a large-eddy simulation, resulting in significantly lower wind speed in the forest edge zone within and at the top of the canopy and (ii) significantly lower turbulence within the canopy in the forest edge zone.

In the dense forests, turbulence showed a maximum closer to the edge than in the sparse forests and decreased to a lower level, but around x = 0.90-1.30 m (3-4 h), turbulence rose again. Dupont and Brunet (2008a) introduced the 'enhanced gust zone' (EGZ), a zone which occurs in the upper part of the canopy between x = 2.5- 6 canopy heights, with intense wind gusts and low turbulence, and mention a region of stronger turbulence further downwind of the EGZ. According to Dupont and Brunet (2008a), this EGZ becomes more intense and develops closer to the leading edge with increasing canopy density. It is plausible that in our study this EGZ was manifested in the decreased turbulence between 1 h and 4 h, predominantly in the denser forests, while in the sparse forests, this EGZ was less pronounced and developed further behind the edge.

Our results are in accordance with the findings of Pahl (2000): when LAI increases, the deposition velocity at the edge is enhanced, but the increase of deposition at the edge relative to the interior deposition remains constant. In a modeling study, Wiman and Ågren (1985) observed in dense forests a strong increase in deposition at the edge, but also a larger enhancement of deposition at the edge relative to the interior deposition (MEI), which was not the case in our study. This was caused by an only moderate increase in deposition in the forest interior at LAI = 10 compared to LAI = 1, a result of a strong reduction of wind speed compensating a large deal of the effect of an increase in collecting area.

Draaijers et al. (1994) found a positive relation between the deposition enhancement in the forest edge zone relative to the interior deposition on the one hand and LAI and stem density on the other. In their study, the highest values of LAI occurred in a *Picea abies* (L.) Karst. and a *Pinus nigra* ssp. laricio Maire stand (Draaijers 1993), which suggests that higher values of LAI coincided with a different needle morphology than lower LAI values. In the modeling study by Wiman and Ågren (1985), a forest of *P. abies* (fine needles) yielded higher deposition velocities at the edge but lower deposition velocities in the interior compared to a forest of *Pinus sylvestris* L. (coarser needles). It is therefore plausible that, in the study by Draaijers et al. (1994), the effect of LAI was obscured by an effect of needle morphology.

4.5.4.2 Crown depth

The penetration depth, the deposition enhancement at the edge, and the sum of deposition in the forest edge zone was similar for both crown depths; however, in case of a steep transition, the enhancement at the edge was higher for the 0.24 m crown depth. We hypothesize that the gradual vegetation sealed off the forest edge over the entire tree height, so the trunk space was cut off from the wind flow upwind of the edge. Consequently, crown depth had no or only a minor effect on wind speed and turbulence within the canopy in the case of a gradual edge, as was confirmed by the measurements. With a steep transition, the wind flow can pass undisturbed into the forest. Dupont and Brunet (2008a) observed a sub-canopy wind jet in the trunk space, which was highest in case of a large trunk space. Consequently, a lower wind speed was expected within the canopy in the case of a 0.18 m crown depth, but this phenomenon was not observed. Dupont and Brunet (2008a) also found that vertical wind speed within the canopy at an edge with large trunk space was much lower compared to an edge with a small trunk space. Although not measured, higher values of vertical wind speed occurring within the canopy with a 0.24 m crown depth probably explain the higher deposition enhancement at the edge.

4.5.4.3 Edge transition

The observed edge effects on deposition were significantly influenced by the edge transition type: the presence of a gradual edge vegetation caused a decrease in the deposition in the forest edge zone by 66 %. With a gradual edge, the deposition enhancement at the edge relative to the interior was significantly lower compared to forests with a steep transition, with the largest differences in the sparse forests. Wind speed and turbulence at half of the tree height initially increased when the flow approached the gradual vegetation, but in the last 0.18 - 0.28 m before the edge, both decreased drastically. In the forest edge zone, this resulted in significantly lower wind speeds and turbulences within the lower part of the canopy and higher wind speeds and turbulences at the canopy top. Consequently, the slowing down and deflection of the wind flow by the gradual edge vegetation in front of the forest edge cause the deposition enhancement at the edge to be lower than at steep edges. While the differences in MEI were obvious, the differences in penetration depth between the forests with different transitions were small and non-univocal, which was a reflection of the wind speed at height 0.15 m. As the strongest correlations of deposition with wind speed and turbulence were found for the lower and upper part of the canopy, the observed edge effects on deposition can be fully explained by the deceleration and deflection of the wind flow by the gradually ascending trees. This result confirms the findings of Agster and Ruck (2003) that a more gradual edge offers more shelter for the forest. They found that it provides a more homogenous pressure pattern within the forest compared to a steep edge.

A potential drawback from the application of a gradual edge may be the deposition on the gradual edge vegetation itself. In our study, however, a gradually ascending edge was still profitable when this deposition was taken into account: the decrease of deposition in the forest edge zone caused by the presence of a gradual edge was at least 1.5 times larger than the deposition on the gradual edge vegetation. The largest gain resulting from the presence of a gradual edge occurred in the low density forests, where the decrease in deposition in the forest was about eight times higher than the deposition on the gradual edge vegetation.

4.6 Consequences for edge management

Stem density and edge transition, but also crown depth in case of a steep transition, can significantly influence edge effects on aerosol deposition. However, as Wiman and Ågren (1985) mentioned, other stand and species-specific differences must be considered as well:

tree height and morphology of needles/leaves are also important factors influencing edge effects (Wiman and Ågren 1985; Pahl 2000). Furthermore, in our study, we used coarse particles to simulate atmospheric deposition. The deposition patterns of fine (submicron) particles or gases, which contain potentially acidifying and eutrophying sulfate, nitrate, and ammonium, are less pronounced, but as their deposition is also driven by canopy wind speed and turbulence, the effects of edge structure are expected to be qualitatively similar (Wiman and Ågren 1985; Beier and Gundersen 1989; Draaijers 1993; Wuyts et al. 2008a, chapter 2). Hence, our findings indicate that through an adapted forest edge layout, edge effects on potentially acidifying and eutrophying deposition on the forest as a whole can be mitigated. However, further research is needed to (i) investigate other edge structure characteristics and at a higher number of levels in a wind tunnel study and (ii) validate previous findings in full-scale forests to formulate best-management practices.

4.7 Conclusions

Patterns of wind speed, turbulence, and deposition were significantly influenced by stem density and edge transition, but also by crown depth in case of a steep transition. Although validation in full-scale forests is to be carried out, we can infer from our findings that an altered layout of forest edges as a result of adjusted management should be able to mitigate the edge effects on potentially acidifying and eutrophying deposition on the entire forest.



5 The effect of gradual forest edges on patterns of throughfall deposition

After: Wuyts, K., De Schrijver, A., Vermeiren, F., Verheyen, K. 2009. Gradual forest edges can mitigate edge effects on throughfall deposition if their size and shape are well considered. Forest Ecology and Management 257, 679-687.

5.1 Abstract

For the protection and promotion of biodiversity in forest edges and interiors, forest edge management practices are put forward such as the creation of gradual forest edges (i.e., edges with a gradual increase of vegetation height from open area to forest, e.g., by means of a fringe, a belt, and a mantle). In this study, we tested the mitigating effect of gradual forest edges on the atmospheric deposition of inorganic nitrogen (N) and the potentially acidifying pollutants SO_4^{2-}, NO_3^-, and NH_4^+ (N+S). We conducted field experiments at three exposed forest edges in Flanders and the Netherlands and compared throughfall deposition at steep edges (i.e., edges with an abrupt transition from open area to forest) and at adjacent gradual edges. Along transects perpendicular to the edges, during three months in both winter and summer, throughfall deposition of Cl^-, SO_4^{2-}, NO_3^-, and NH_4^+ was monitored in the forest between 0 and 64 m from the edges and in the gradual edge vegetation. At the smoothest and best fitting gradual edge, the extra N+S throughfall deposition the forest received due to edge effects was lower than at the adjacent steep edge, with, on average, 80 and 100 % in winter and summer, respectively. This was due to a halving of the depth of edge influence and an almost full reduction of the magnitude of edge influence. This decrease in throughfall deposition in the forest was not compensated by the additional throughfall deposition on the gradual edge vegetation itself, resulting in a final decrease in throughfall deposition in the forest edge by 60 % in winter and 74 % in summer. While this result confirms that gradual edges can mitigate edge effects on atmospheric deposition, the results of the other sites indicate the importance of size and shape of the gradual edge vegetation in mitigating edge effects on deposition: due to insufficient height (‘size’) or inadequate shape of the gradual edge vegetation, only small or insignificant decreases in throughfall deposition were observed. Hence, for mitigating edge effects on N+S and N deposition, our results support the recommendation of creating gradual edges at forests with poorly developed, abrupt edges, but

it stresses the importance of a thorough consideration of the shape and size of the gradual edge vegetation in the design and management of gradual forest edges.

5.2 Introduction

Transitions of open land to forests or so-called forest edges can act as local biodiversity hotspots of fauna and flora: they amalgamate species from both forest and open land, but may also harbor characteristic 'ecotonal species' (Yahner 1988; Downie et al. 1996; Menzel et al. 1999; Lloyd et al. 2000; Magura 2002; Ohwaki et al. 2007). Not forest edges per se, but their composition and structure seems of high relevance for biodiversity aspects. Abrupt or hard edges lack the 'ecotonal structures' of highly structured, gradual or soft edges (a herbaceous fringe, a shrub belt, and a forest mantle), which provide a slower transition, between sunlit, warmer, and drier open land and shady, cooler, and more humid forests, and more resources such as food and shelter than abrupt edges (Wermelinger et al. 2007). For various insects, for example, gradual forest edges support a higher species richness than steep edges (Duelli et al. 2002; Wermelinger et al. 2007), and for birds characteristic of edges, abrupt edges act as 'ecological traps' as they attract large populations but suffer increased nest predation rates than gradual ones (Suarez et al. 1997; Flaspohler et al. 2001; Deng et al. 2003). As a result, forest edge management practices for temperate forests are put forward to create 'high quality' edges that provide favorable habitat for edge species (e.g., Temple and Flaspohler 1998; Magura 2002; Wermelinger et al. 2007) and to protect interior habitats in forest remnants from deeply penetrating edge effects (Murcia 1995; Didham and Lawton 1999; Gascon et al. 2000; Cadenasso and Pickett 2001; Weathers et al. 2001). These management choices may also affect the input of nitrogen (N) and the potentially acidifying pollutants NH_4^+, NO_3^-, and SO_4^{2-} (N+S) via atmospheric deposition at edges.

Ample studies have demonstrated the manifestation of so-called edge effects on atmospheric deposition in forests: throughfall deposition at abrupt forest edges is significantly enhanced in comparison with the forest interior [Beier and Gundersen 1989; Draaijers et al. 1994; Weathers et al. 2001; for an overview, see De Schrijver et al. (2007a)]. Field research on the impact of adjusted edge management on edge patterns of atmospheric deposition was until now restricted to the influence of the occurring forest type (Spangenberg and Kölling 2004; Wuyts et al. 2008a, 2008b: chapter 2 and 3) and of closed versus open edges (Weathers et al. 2001). To our knowledge, the impact of edge transition type (i.e., gradual versus steep edges), however, is unexplored in the field. Nonetheless, a wind tunnel study was performed in chapter 4 (Wuyts et al. 2008c), in which the influence of stand density, crown depth, and also gradual edges on deposition, wind speed, and turbulence in the forest edge was studied. Here, we reported that steep forest edges abruptly interrupt the wind flow approaching perpendicular to the edge, while gradual edges can divert the wind flow, hereby reducing wind speed, turbulence, and the level of deposition enhancement at the edge.

Moreover, because of the high species richness at open land-forest ecotones, forest edges in regions with a high deposition load are particularly vulnerable to effects of high N and N+S deposition. Although some positive effects may occur to a given level of N deposition load (e.g., on tree growth and production; Magill et al. 1997; Wallace et al. 2007), excess nitrogen deposition is mainly expressed in negative effects such as loss of characteristic, N-efficient species and dominance of N-requiring species in the forest herb layer, leading to declines in forest biodiversity (Bobbink et al. 1998; Gilliam 2006). Thimonier et al. (1992) found a 20-year vegetation change towards a more eutrophic level to be the largest at forest edges, and particularly at edges exposed to the prevailing winds, which they attributed to enhanced atmospheric deposition at these edges. With acidification, nutrient imbalances in vegetation and vitality decreases may occur via nutrient leaching from the soil and high aluminium concentrations that impede plant ion uptake (Schulze 1989). As a result, acid-tolerant species may become dominant while rare plants, often typical of intermediate pH's, decline (Bobbink et al. 1998; Thimonier et al. 1994). It is likely that the higher nitrogen and sulphur depositions at edges also affect faunal diversity indirectly, through effects on floral diversity. As no vast decrease in N emissions is expected in the short term (Galloway et al. 2003; Cofala et al. 2007), measures should be taken to reduce edge effects on potentially acidifying and N throughfall deposition and their associated negative consequences.

Because, in regions with a high deposition load, mitigating measures are sought to reduce edge effects on N and N+S deposition and the creation of gradual edges is already suggested for protecting and favoring biodiversity at forest edges and interiors, the aim of this study was to assess the potential in forest edges fitted with gradual edge vegetation for diminishing the atmospheric deposition of N and N+S in full-scale forests. Our hypothesis was that gradual edges cause edge effects to be smaller and to extend less deep into the forest than steep edges do. To test this hypothesis, we conducted field experiments at three sites in Flanders and the Netherlands and compared throughfall deposition in paired forest stands with a steep and a gradual forest edge.

5.3 Materials and methods

5.3.1 Site description

We were able to select forest stands at three sites, i.e., at 'Dombergheide' (DH) and 'Neigembos' (NB) in Flanders (eastern part of Belgium) and at 'Neterselse Heide' (NH) in the Netherlands (Fig. 5.1). Site location and soil type of the forest stands, as well as the main tree species and leaf area index (LAI) of the stands and the gradual edge vegetation, are presented in Table 5.1. Fig. 5.2 presents an overview of the shape of the gradual edge vegetation and its dimensions relative to the stand behind it at the three sites. All three stands encompassed a south- to westerly oriented forest edge, which was exposed to the prevailing wind direction and was fitted with gradual edge vegetation. The gradual edge vegetation at the DH site consisted of *Populus tremula* L. trees, of which tree height increased with decreasing upwind distance towards the edge of the mixed *Quercus robur* L. - *Betula pendula* Roth stand, with an inclination angle of 38°. The edge vegetation at the NB site was planted in 1995 as a shrub belt of *Cornus mas* L., *Crataegus laevigata* (Poiret) DC., and some *Corylus avellana* L. and *Euonymous europaeus* L. The shrubs in the belt and the grasses in the fringe were overgrown by *Rubus* sp., hence, the transition of the grassland to the shrubs was smoothened out. Small trees of *Prunus avium* L. and *Ulmus laevis* Pallas shaped the forest mantle, situated between the shrubs and the first trees of the *Fagus sylvatica* L. stand, and leaned over to the shrubs. Overall, the gradual edge vegetation had an inclination angle of 29°. Heavy branches of the front *F. sylvatica* trees reached over to more than half the depth of the gradual edge vegetation. At the NH site, a wide strip of lower *Pinus sylvestris* L. trees was positioned in front of the *P. sylvestris* stand and, in the strip, tree height increased with decreasing upwind distance to the edge at an angle of 16°.

Meteorological data were obtained from the Royal Meteorological Institute of Belgium for the weather station at Ukkel (50°47'52"N, 04°21'31"E). In 2007, annual rainfall amounted to 880 mm, mean annual temperature was 11.5°C, mean temperature of the coldest month (December) amounted to 4.1°C, and mean temperature of the warmest month (June) was 17.5°C. Mean annual relative humidity was 80 % and mean annual wind speed amounted to 3.3 m s^{-1}. Mean temperatures during the winter and summer measuring period were 7.4°C and 16.9°C and mean wind speeds were 4.0 m s^{-1} and 3.1 m s^{-1}, respectively.

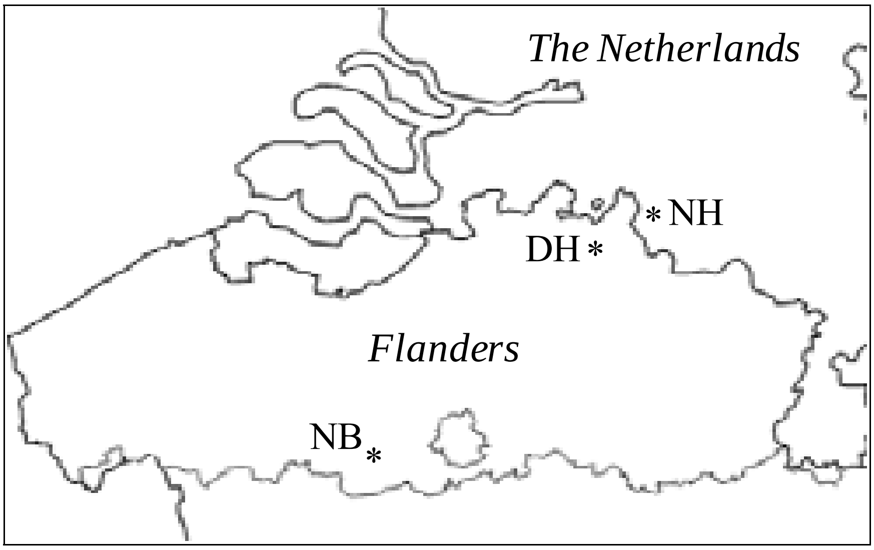

Fig. 5.1: Location of the sites ‘Dombergheide’ (DH) and ‘Neigembos’ (NB) in Flanders and the site ‘Neterselse Heide’ (NH) in The Netherlands

The three sites were situated at about 80-100 km east of the North Sea. Important sources of NO_x and SO_y are (petrochemical) industries at the Antwerp and Ghent Harbors, at 50-100 km west of the DH and NH sites and at 60 km north to north-east of the NB site. Flanders is characterized by a very dense network of roads and highways, which are important sources of NO_x in this region (Van Avermaet et al. 2006). The DH site was located in a region with intensive livestock breeding and was consequently subject to a high level of atmospheric NH_x.

Table 5.1: Site code, latitude and longitude coordinates and soil type (classification FAO-ISRIC-ISSS 1998) of the three sites and mean LAI and main tree species composition (in the tree and shrub layer) of the stand (FS) and the gradual edge vegetation (GEV) in front of the stand. Stand density SN, mean tree height H, dominant tree height Hdom, basal area BA, and stem volume V are given for each tree species (-: not measured).

Site	Location	Soil type		LAI		Species	SN	H	H_{dom}	BA	V
				winter	summer		(m)	(m)	(m)	(m^2 ha^{-1})	(m^3 ha^{-1})
Dombergheide (DH)	51° 21' 00"N	haplic podzol	FS:	0.63	4.41	*Quercus robur* L.	490	14.2	20.7	20	171
	04° 57' 02"E					*Betula pendula* Roth	237	14.8	20.9	6	51
						Prunus serotina Ehrh.	643	6.5		2	10
			GEV:	0.50	2.83	*Populus tremula* L.	3500	5.3	13.3	8	36
Neigembos (NB)	50° 48' 40"N	albeluvisol	FS:	0.63	4.11	*Fagus sylvatica* L.	227	17.4	28.8	41	452
	04° 03' 40"E					*Corylus avellana* L.	382	3.6		1	1
			GEV:	1.11	5.22	*Prunus avium* (L.) L.	338	5.3	8.8	8	31
						Crataegus laevigata (Poiret) DC.	369	5.5	7.3	2	6
						Cornus mas L.	646	4.2	6.0	1	3
Neterselse Heide (NH)	51° 25' 47"N	haplic podzol	FS:	1.32	-	*Pinus sylvestris* L.	393	13.0	16.0	19	125
	05° 11' 44"E					*Betula pendula* Roth	129	6.0		1	4
			GEV:	1.62	-	*Pinus sylvestris* L.	771	8.3	13.1	32	143
						Betula pendula Roth	54	9.5		2	9

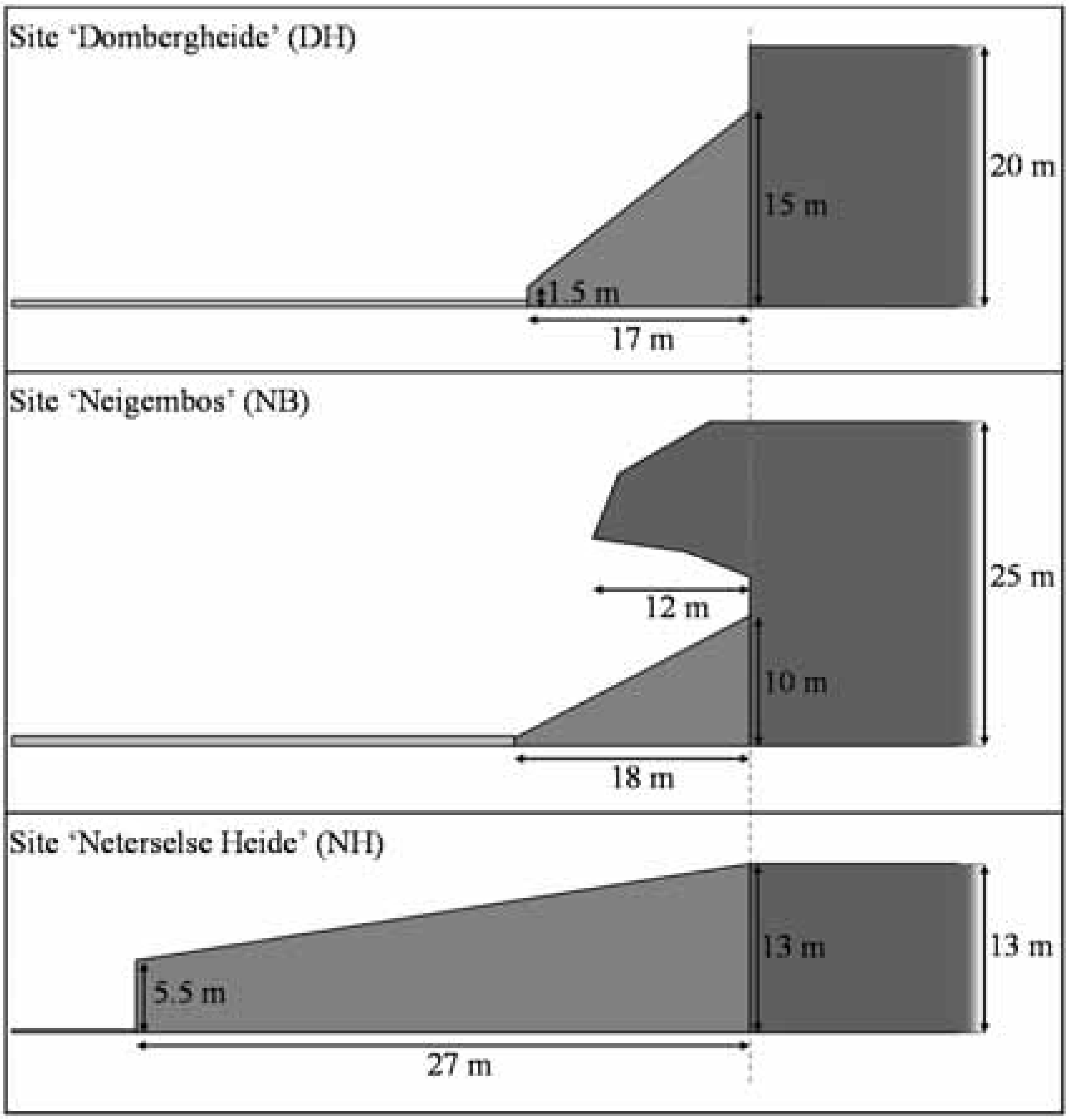

Fig. 5.2: Schematic overview of the shape and dimensions of the gradual edge vegetations (light grey) at the edge of the forests (dark grey) in the three study sites (DH and NB in the north of Flanders and NH in the south of The Netherlands). These gradual edge vegetations were characterized by a gradually ascending vegetation height, from the open area to the edge of the forest stand and, hence, create a more gradual transition in comparison with a steep forest edge.

5.3.2 Experimental setup, sampling, and sample analyses

At each site, experiments were set up to compare a gradual and an adjacent steep forest edge. Along a part of the forest edge at the NH stand, the gradual edge vegetation was absent and a steep edge occurred. At the edge of the DH and NB stands, along a distance of 15-35 m parallel with the edge, the gradual edge vegetation was cut entirely to create a steep transition. At each site, six parallel transects were established perpendicular to the forest edge, i.e. three at the edge with a steep transition (no gradual edge vegetation) and three at the edge with a gradual transition (with a gradual edge vegetation). Along these transects, throughfall deposition collectors were set up (i) within the forest stand, at fixed distances of 0, 2, 4, 8, 16, 32, and 64 m from the edge, and (ii) in front of the forest stand, i.e., within the gradual edge vegetation or up to 10 - 12 m before the steep edge of the forest, at every 2 - 4 m. Also open-field bulk deposition collectors were installed at each of the sites. Similar throughfall deposition and open-field deposition collectors were used as in chapter 3 (Wuyts et al. 2008b). Bottles were partially buried to protect samples from direct sunlight and heat.

Throughfall deposition was collected during three months in winter, from 22 December 2006 to 16 March 2007, and summer, from 13 June to 11 September 2007. At every fortnight, the funnels, bottles, and meshes were replaced by material rinsed with distilled water, sample volume was determined in the field, and a 300 ml subsample was taken for chemical analyses. In the laboratory, the subsamples of two subsequent collections were pooled volume-weighted into monthly samples, which were subsequently filtered (nylon filter, 0.45 µm, Rotilabo, Roth) and analysed for NH_4^+ (photometric determination of a reaction product of NH_4^+ at λ = 660 nm according to the Dutch standard method NEN 6576, Cary 50, Varian) and for Cl^-, SO_4^{2-}, NO_3^-, and PO_4^{3-} (ion chromatography, ICS-90, Dionex). The quality of the analyses was checked by including method blanks and repeated measurements of internal concentration standards. For all ions, a coefficient of variation lower than 5 % (except for NH_4^+: 8.6 %) and a recovery rate higher than 95 % were derived from repeated measurements of certified reference material CRM 409, performed in the study period. The PO_4^{3-} concentration was used to check sample quality because samples with an elevated PO_4^{3-} concentration could have been contaminated by bird droppings (Erisman et al. 2003). At the edge and in the forest interior of the stand at the NH site, the integrity of N in the fortnightly samples was assessed in summer by reanalysing throughfall water after replacement in the field for a second fortnight: NH_4^+ and NO_3^- concentrations were, on average, 0.41 ppm (or 5.5 %) lower after a fortnight in the field.

5.3.3 Data analysis

Monthly ion throughfall deposition fluxes (equiv ha^{-1}) were obtained from the multiplication of monthly sample volume with the ion concentration in that monthly sample, divided by the surface area covered by one funnel. These monthly throughfall deposition fluxes were summed into winter and summer throughfall deposition fluxes.

Repeated-measures analyses (ANOVA) were performed for the three sites and the four studied ions individually, upon the ln-tranformed winter and summer throughfall deposition fluxes of all transects. The factor 'distance from the edge' was treated as a within-subject factor and 'edge transition' (2 levels: steep edge - gradual edge) as a between-subject factor. To avoid the assumption of sphericity, the multivariate approach (Pillai's trace test statistic) was used to test for within-subject effects (O'Brien and Kaiser 1985). The analyses were carried out on the throughfall deposition data at the outer edge (0 m) and the end of the transect (at 64 m from the edge) to detect the overall trend of throughfall deposition in the edge zone of the stands [i.e., the zone of the stand influenced by an edge effect on throughfall deposition; in chapter 3 (Wuyts et al. 2008b), we found edge effects to extend to a distance of up to about 60 m from the edge]. The choice of two measuring points for analysis also caused the multivariate tests for within-subject effects not to be disabled by too low degrees of freedom.

The penetration depth of the edge effect [or forest edge distance (chapter 2) or distance/depth of edge influence (Chen et al. 1992; Harper et al. 2005); DEI] and the level of deposition enhancement at the outer edge [or forest edge enhancement factor (chapter 2) or magnitude of edge influence (Burton 2002); MEI] were calculated for every transect, based on the winter and summer throughfall deposition flux of Cl^-, SO_4^{2-}, NO_3^-, and NH_4^+ individually. The method developed by Beier and Gundersen (1989) was applied to estimate the DEI, while the MEI was calculated as the ratio of the mean throughfall deposition at 0 and 2 m from the edge to the mean of the throughfall deposition at 32 and 64 m. The extra throughfall deposition that reaches the forest floor in the entire edge zone as a result of edge effects ('extra edge flux', area A in Fig. 5.3) was calculated for all transects, similar to 'area A' as described in chapter 2. For every transect, a linear curve was fitted to the ln-transformed throughfall deposition flux in function of the distance to the forest edge, from the edge (0 m) until the end of the edge effect (at DEI): $\ln(y) = ax + b$ (with x: distance from the edge (in m), y: throughfall deposition flux, a: slope, and b: constant). The extra edge flux was then determined as (see Fig. 5.3) area A plus area B (calculated as an integral between 0 and DEI) minus area B. The parameter 'extra edge flux' incorporates both the DEI and the MEI and enables straightforward comparison of edge effects.

Furthermore, the deposition before the edge of the forest relative to the open-field deposition was estimated (area E in Fig. 5.3). This is the deposition the area in front of the forest stand receives a result of the presence of a forest stand with a steep or gradual transition. This deposition fluxes should be taken into account when assessing the effect of a gradual edge vegetation: the presence of a gradual edge vegetation could cause a decrease in deposition flux in the forest edge zone, but this decrease can be counterbalanced by an increase in deposition flux before the edge of the stand due to an increase in collecting surface area caused by the gradual edge vegetation. Therefore, the area-weighted throughfall flux was calculated for the entire length of the gradual edge vegetation (area E plus area OF in Fig. 5.3) and the open-field deposition over the same length (area OF in Fig. 5.3) was subtracted. A linear increase in deposition flux was assumed from the open-field level to the deposition at the first collector. In case of a steep edge, the area underneath the deposition curve was calculated over a distance equal to the length of the gradual edge vegetation of the same site.

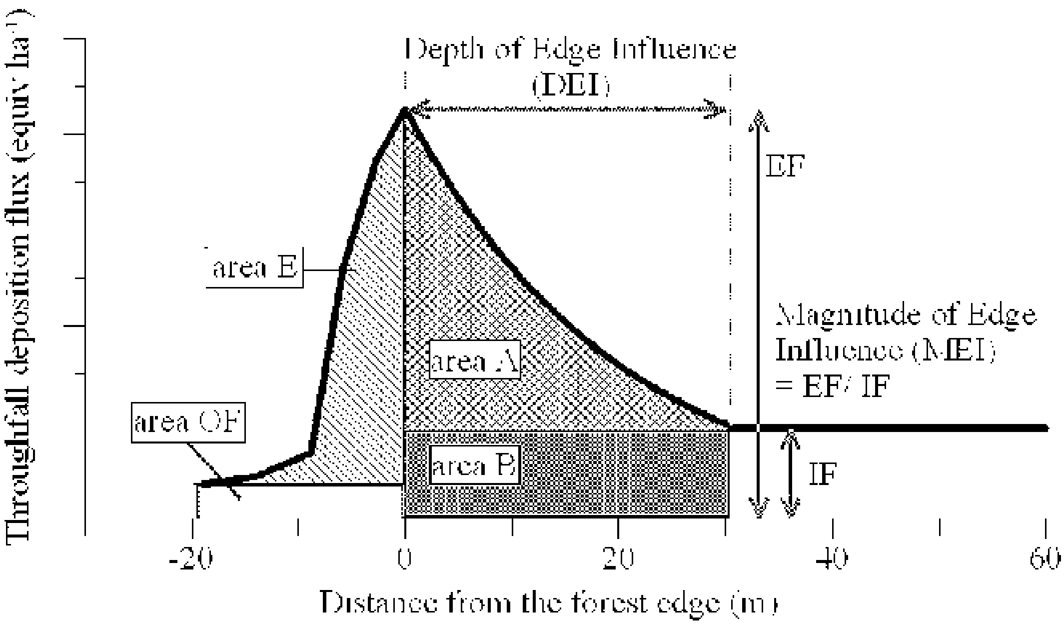

Fig. 5.3: Theoretical throughfall deposition curve along a transect perpendicular to the forest edge. Indicated are the depth and magnitude of edge influence (DEI and MEI) and the areas calculated in this study to asses the total effect of edge proximity on throughfall deposition in and in front of the forest stands (area A and area E). Area A is the 'extra edge flux' and represents the extra throughfall deposition that reaches the forest floor due to the proximity of the forest edge; area E is the extra throughfall deposition in front of the forest stand due to the proximity of the forest edge.

Finally, for the three sites individually, differences in edge effects on N+S and N deposition between a gradual and a steep forest edge were investigated by use of t-tests (two independent samples). We investigated differences in the DEI, the MEI, the area A, and on the area A + area E for winter and summer deposition of SO_4^{2-}, NO_3^-, and NH_4^+ individually. Before analysis, the data groups were tested for normality and homogeneity of variances. In addition, to facilitate comparison with our wind tunnel study in chapter 4 (Wuyts et al. 2008c), we performed a Mann-Whitney test on the DEI, the MEI, the area A, and on the area A + area E for winter and summer throughfall deposition of Cl^-.

5.4 Results

At all sites, the mean winter and summer throughfall deposition fluxes were clearly enhanced at the edges with a steep transition, except for NH_4^+ at the NB site in winter (Fig. 5.4). Remarkably, edge effects did not always cause throughfall deposition to peak at zero distance of the forest edge, at the first tree stem, but peak values also occurred several meters before or behind the edge. The edges with a gradual transition generated edge effects as well. However, for all ions at the NH site, for Cl^-, SO_4^{2-}, and NO_3^- at the NB site, and for Cl^- and SO_4^{2-} at the DH site, these edge effects were less pronounced than those caused by a steep transition (Fig. 5.4). The greatest edge effects were observed at the steep edge of the NH site and for Cl^- in winter. At each site, between 16 and 64 m from the edge, the ratios of NH_4^+ and NO_3^- throughfall deposition to SO_4^{2-} throughfall deposition were similar for the gradual and steep transitions. At the edges of all sites, the ratio NH_4^+:SO_4^{2-} was increased in the case of a gradual transition in comparison with a steep transition. The ratio NO_3^-:SO_4^{2-} as well was higher at the edge with gradual transition than at the edge with steep transition, but only at the Dombergheide site.

At the DH site, only the Cl^- throughfall deposition fluxes were significantly influenced by edge transition (Table 5.2); significant interactions were observed between the effects 'distance to the edge' and ' edge transition', for Cl^-, NO_3^-, and NH_4^+ in winter for Cl^-, SO_4^{2-}, and NO_3^- in summer (Table 5.2). At the NB site, we found a significant effect of transition on the SO_4^{2-} deposition in winter and NH_4^+ deposition in summer, and we detected a significant interaction between distance to the edge and transition for NO_3^- in winter (Table 5.2). At the NH site, we detected a significant effect of distance and transition on the throughfall deposition of all ions and in both time periods, except for NH_4^+ in summer; a significant interaction between distance and transition was found only for SO_4^{2-} in winter (Table 5.2).

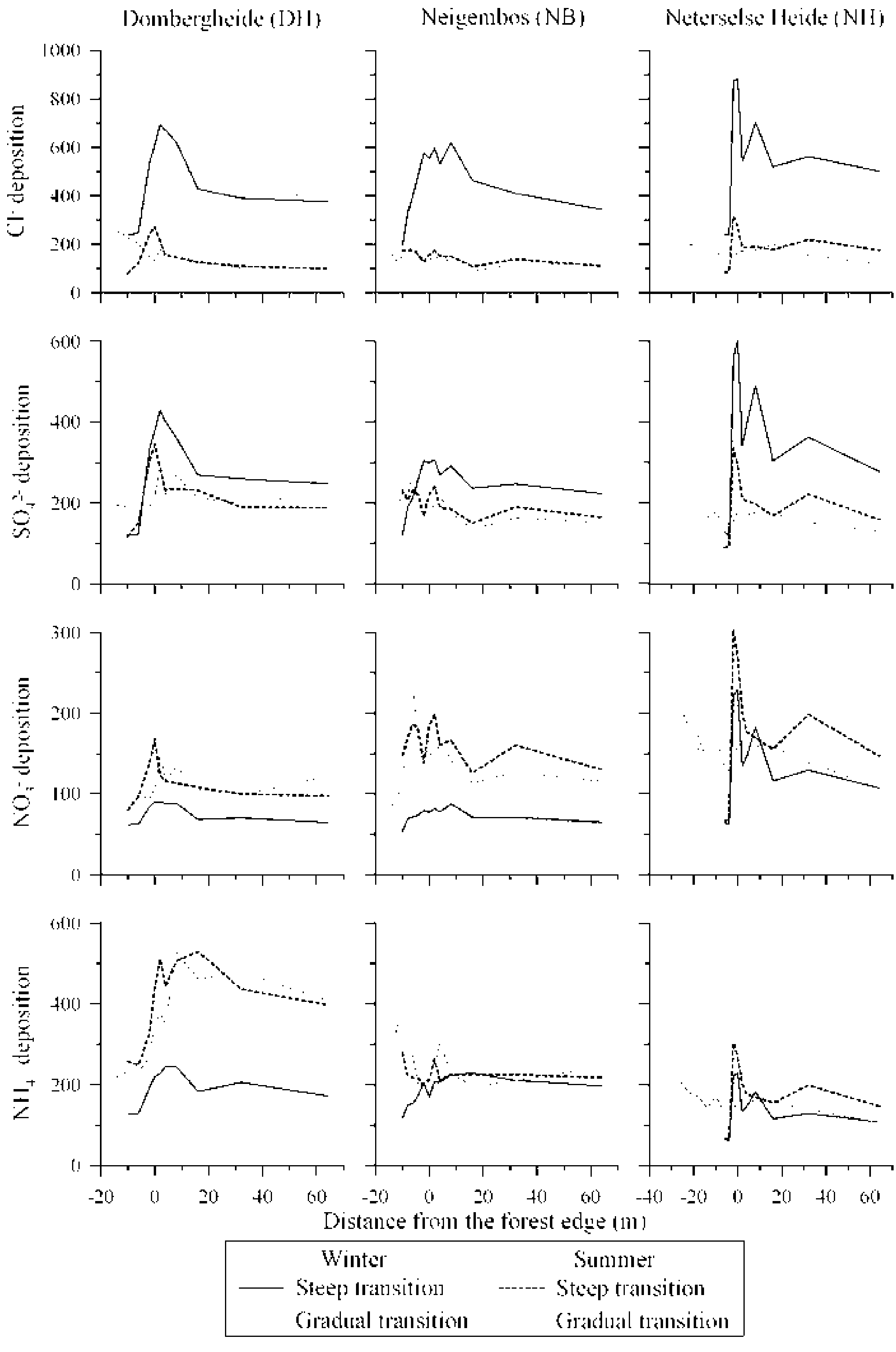

Fig. 5.4: Mean throughfall deposition (in equiv ha^{-1}) of Cl^-, SO_4^{2-}, NO_3^-, and NH_4^+ in summer and winter, along transects across the south- to westerly oriented forest edges of the three studied forest stands. Transects were established at adjacent parts of the forest stands, where the forest edge was fitted with a gradually ascending edge vegetation ('gradual transition') and where vegetation in front of the forest edge was cut or was lacking ('steep transition').

Table 5.2: Repeated-measures ANOVA outcome, indicating the significance of the effect of distance to the forest edge (or edge effect) and of edge transition type (i.e., gradual versus steep transition) on the throughfall deposition of Cl^-, SO_4^{2-}, NO_3^-, and NH_4^+ (underlined: $p < 0.10$; bold: $p < 0.05$). Significant interactions between the distance and transition effects suggest edge effects to depend on edge transition type. Analyses were performed on summer and winter data individually and for each of the study sites (DH: Dombergheide; NB: Neigembos; NH: Neterselse Heide)

	Source of variation	Winter				Summer			
		Cl^-	SO_4^{2-}	NO_3^-	NH_4^+	Cl^-	SO_4^{2-}	NO_3^-	NH_4^+
DH	Within-subject								
	distance	0.187	0.740	0.141	0.671	**0.038**	0.189	0.259	0.692
	distance x transition	<u>0.087</u>	0.106	**0.010**	**0.029**	**0.029**	<u>0.076</u>	<u>0.066</u>	0.567
	Between-subject								
	transition	0.169	0.198	0.272	0.352	0.292	0.393	0.521	0.676
NB	Within-subject								
	distance	**0.000**	**0.011**	**0.001**	0.891	<u>0.062</u>	<u>0.050</u>	**0.045**	0.834
	distance x transition	0.871	0.789	<u>0.080</u>	0.210	0.654	0.977	0.897	0.630
	Between-subject								
	transition	0.267	<u>0.050</u>	0.460	0.761	0.610	0.314	0.181	<u>0.098</u>
NH	Within-subject								
	distance	**0.012**	**0.004**	**0.001**	**0.008**	<u>0.088</u>	**0.025**	**0.024**	**0.021**
	distance x transition	0.352	0.650	0.296	0.812	0.306	<u>0.082</u>	0.165	0.200
	Between-subject								
	transition	**0.001**	**0.010**	**0.005**	**0.023**	**0.036**	**0.026**	**0.011**	0.104

At DH, both lower and higher mean values were observed for the DEI at the gradual transition in comparison with the steep transition (Table 5.3), but, on average, the DEI was about two times shorter in the case of a gradual transition. The MEI showed consistently higher values in the case of the steep transition (on average, 1.53 times higher in winter and 1.43 times higher in summer). Consequently, the area A (see Fig. 5.3) was higher at the steep transition (except for Cl^- and NO_3^- in summer). Area E, however, was lower at the steep than at the gradual edge (except for NO_3^- in summer and winter and for NH_4^+ in summer). At NB, the presence of the gradual transition yielded similar edge effects as the steep transition in winter, according to the DEI, the MEI, and the area A. The edge effects in summer penetrated about

6-12 m deeper and the enhancement was, on average, 1.10 times higher in case of the steep transition, which was reflected in only small differences in the area A. Considerably higher values were found for the area E in the case of the gradual transition, particularly for SO_4^{2-} and NH_4^+. At the steep transition of the NH site, the DEI was 24-42 m longer and the MEI was, on average, 1.22 (in winter) and 1.25 (in summer) times higher than at the gradual transition. This resulted in consistently and considerably higher values for the area A in the stand with the steep transition (factor 5.7 in winter and 8.0 in summer). However, at the gradual transition, the area E was, on average, 4.98 (in winter) and 4.10 (in summer) times higher than at the steep transition.

At the DH and NH sites, the mean values for the SO_4^{2-}, NO_3^-, and NH_4^+ deposition of the DEI, the MEI, the area A, and the area A + area E were higher in the case of a steep transition than in the case of a gradual transition (Table 5.4). Significant differences in the DEI between the two edge transitions were found for DH in winter ($p = 0.009$) and for NH in winter ($p < 0.001$) and summer ($p = 0.001$; Table 5.4). Differences in the MEI and the area A between the edge transitions were significant for DH in winter ($p = 0.005$; 0.036) and summer ($p = 0.009$; 0.025) and for NH in winter ($p = 0.099$; 0.011) and summer ($p = 0.022$; 0.003; Table 5.4). Differences in the area A + area E were only significant for DH in summer ($p = 0.037$). For the Cl^- throughfall deposition, significant differences in the DEI between an abrupt and a gradual edge were found at the NH site ($p = 0.015$), in the MEI at the DH site ($p = 0.009$), and in the area A at the NH site ($p = 0.015$; Table 5.4).

Table 5.3: The depth of edge influence (DEI, m) and the magnitude of edge influence (MEI) of the edge effects on Cl^-, SO_4^{2-}, NO_3^-, and NH_4^+ throughfall deposition and the resulting extra throughfall deposition in and in front of the forest stand due to edge proximity (area A and area E, respectively; Fig. 5.3) at the three sites (DH: Dombergheide; NB: Neigembos; NH: Neterselse Heide).

Site	Period	Ion	Steep transition				Gradual transition			
			DEI	MEI	area A	area E	DEI	MEI	area A	area E
DH	winter	Cl^-	37 (14)	1.88 (0.38)	6277 (3021)	1487 (80)	35 (16)	1.12 (0.18)	868 (784)	1976 (500)
		SO_4^{2-}	53 (11)	1.81 (0.40)	5216 (1213)	952 (16)	48 (16)	1.03 (0.20)	1733 (1654)	1408 (503)
		NO_3^-	53 (11)	1.43 (0.10)	837 (60)	155 (22)	13 (3)	1.01 (0.09)	-94 (58)	150 (76)
		NH_4^+	53 (11)	1.32 (0.10)	1964 (131)	355 (76)	13 (9)	0.88 (0.09)	-31 (187)	588 (201)
	summer	Cl^-	16 (0)	2.50 (0.54)	760 (261)	759 (179)	32 (16)	1.17 (0.04)	759 (406)	1865 (134)
		SO_4^{2-}	28 (18)	1.70 (0.25)	1356 (1033)	834 (230)	11 (5)	1.11 (0.06)	377 (194)	1069 (266)
		NO_3^-	19 (7)	1.49 (0.26)	305 (118)	299 (123)	11 (5)	1.05 (0.03)	194 (109)	213 (83)
		NH_4^+	35 (16)	1.20 (0.07)	2511 (764)	833 (247)	21 (21)	0.89 (0.01)	-600 (600)	341 (120)
NB	winter	Cl^-	43 (11)	1.68 (0.09)	4466 (1682)	3281 (168)	43 (11)	1.56 (0.02)	4119 (917)	3433 (170)
		SO_4^{2-}	21 (5)	1.36 (0.08)	753 (248)	1679 (33)	21 (5)	1.29 (0.03)	851 (304)	2498 (239)
		NO_3^-	37 (14)	1.23 (0.04)	247 (94)	285 (31)	37 (14)	1.21 (0.06)	255 (103)	329 (105)
		NH_4^+	0 (0)	0.97 (0.09)	0 (0)	952 (17)	16 (9)	1.18 (0.16)	431 (272)	1267 (204)
	summer	Cl^-	28 (18)	1.50 (0.06)	678 (283)	1220 (72)	16 (0)	1.26 (0.14)	512 (72)	1278 (105)
		SO_4^{2-}	28 (18)	1.40 (0.09)	783 (267)	1593 (99)	21 (5)	1.28 (0.05)	761 (166)	1877 (89)
		NO_3^-	28(18)	1.48 (0.10)	796 (342)	1290 (103)	21 (5)	1.29 (0.12)	696 (311)	1202 (24)
		NH_4^+	27 (19)	1.09 (0.03)	619 (362)	429 (168)	21 (5)	1.11 (0.22)	790 (125)	890 (82)
NH	winter	Cl^-	43 (11)	1.78 (0.24)	8325 (2816)	2068 (354)	19 (7)	1.38 (0.17)	1656 (808)	9093 (1444)
		SO_4^{2-}	53 (11)	2.19 (0.18)	8049 (2220)	1355 (269)	16 (8)	1.84 (0.23)	1337 (624)	6476 (385)
		NO_3^-	43 (11)	2.14 (0.14)	2206 (789)	491 (97)	16 (8)	1.67 (0.16)	401 (168)	1886 (189)
		NH_4^+	53 (11)	2.50 (0.26)	11823 (3546)	1308 (522)	16 (8)	2.20 (0.43)	1916 (683)	9040 (297)
	summer	Cl^-	53 (11)	1.33 (0.19)	2594 (1062)	834 (64)	13 (10)	1.16 (0.20)	430 (261)	2817 (405)
		SO_4^{2-}	53 (11)	1.59 (0.07)	3384 (891)	558 (29)	27 (19)	1.15 (0.25)	719 (652)	2048 (260)
		NO_3^-	53 (11)	1.60 (0.10)	2898 (837)	525 (28)	16 (9)	1.27 (0.30)	331 (174)	2277 (296)
		NH_4^+	53 (11)	1.47 (0.11)	6900 (1906)	1014 (80)	11 (5)	1.20 (0.14)	558 (407)	5087 (617)

Table 5.4: Mean depth of edge influence (DEI), magnitude of edge influence (MEI), the extra throughfall deposition in the forest stand due to edge proximity (area A), and the extra throughfall deposition in and in front of the forest stand due to edge proximity (area A + area E; see Fig. 2) of SO_4^{2-}, NO_3^-, and NH_4^+ throughfall deposition in winter and summer. Values for the steep transitions and the gradual transitions were compared for each site (DH: Dombergheide; NB: Neigembos; NH: Neterselse Heide) and the significant differences are indicated (based on two independent samples t-test; † $p < 0.10$; * $p < 0.05$).

Site	Period	DEI (m)			MEI (-)			Area A (equiv ha^{-1} m)			Area A+area E (equiv ha^{-1} m)		
		steep	gradual		steep	gradual		steep	gradual		steep	gradual	
DH	winter	53.3	24.9	*	1.50	0.97	*	2672	536	*	3160	1251	
	summer	27.1	14.2		1.47	1.02	*	1391	-9	*	2046	531	*
NB	winter	19.6	24.9		1.19	1.21		333	512		1306	1877	
	summer	27.2	21.3		1.32	1.22		732	749		1836	2072	
NH	winter	49.8	16.0	*	2.27	1.91	†	7359	1218	*	8411	7019	
	summer	53.3	17.7	*	1.56	1.21	*	4394	536	*	5093	3674	

5.5 Discussion

At the steep edges, peak values of throughfall deposition fluxes did not always occur at the very edge of the forest, i.e., at the first tree stem, but before and behind the first tree stem. This was probably the result of tree height peaking several meters behind the very edge or the main part of the crown of the trees on the first tree row being orientated or even leaning over several meters into the lighter open area. This implies that (i) when only throughfall deposition is monitored within the forest stand, from zero distance on, the magnitude of edge effects can be underestimated and (ii) the highest throughfall deposition fluxes may not always occur within the forest edge, which generates the edge effects, but in the field, meadow, or heathland adjacent to it. We took into account the deposition fluxes in front of the forest edge by calculating area E in Fig. 5.3, but in other studies too, the enhanced deposition in front of the edge should be considered when assessing the full consequence of edge effects.

The magnitude and the penetration depth of edge effects on atmospheric deposition in forests depend on the ion considered, meteorological conditions (e.g., wind speed and direction), edge orientation, and edge structure (Draaijers et al. 1994; Weathers et al. 2001; Wuyts et al. 2008a, 2008b: chapter 2 and 3). By comparing edge effects at a gradual edge with those at an adjacent steep edge of the same homogeneous forest stand, we ensured that the edges had the same orientation and were studied within the same time period and, thus, under the same meteorological conditions. As we also investigated the same ions for all edge types, we can assume that differences in edge effects between gradual and steep edges are caused by differences in edge structure.

5.5.1 The 'edge softening' by gradual edges in theory and in the field

In chapter 4 (Wuyts et al. 2008c), we provided 'model-based' evidence of the potential of gradual forest edges for mitigating edge effects on atmospheric deposition under ideal, controlled circumstances in a wind tunnel. In theory, deposition enhancements relative to the forest interior are significantly lower in a forest with a gradual edge than in one with a steep edge: the gradual edge vegetation causes deceleration and deflection of the wind flow, inducing lower wind speed and lower turbulence in the lower part of the forest canopy. Consequently, the extra deposition the edge zone of the forest receives due to edge effects is lower in the case of a gradual transition. Similarly, at the DH site, we observed a significantly smaller increase in Cl^- deposition (MEI) at the edge with a gradual transition and no significant effect on the penetration depth of the edge effect (DEI). We also observed less extra Cl^- throughfall deposition (area A in Fig. 5.3) caused by edge effects at the gradual edges than at the steep edges, but the difference was not significant. Because in this study (i) the wind flow is not continuously directed perpendicular to the edge, and (ii) the shape, size, and composition of the gradual edge were not the same as in the wind tunnel study, the observed effect of edge transition on patterns of Cl^- differed to some extent from the 'theoretical effect' as observed in the wind tunnel. We can conclude that the presence of a gradual edge vegetation can significantly influence the aerodynamic processes that cause edge effects on dry deposition by a decrease in MEI, as indicated by the patterns of Cl^- throughfall deposition (as indicator for total deposition of coarse particles) in this study and of Cl^- dry deposition in our wind tunnel study (chapter 4).

In regard to SO_4^{2-}, NO_3^-, and NH_4^+ throughfall deposition, edge effects can, next to aerodynamic processes (Draaijers et al. 1994; Ould-Dada et al. 2002), also be induced by microclimatic edge gradients (Matlack 1994; Chen et al. 1995). Gradients in temperature and moisture can alter dry deposition of gaseous compounds and/or canopy exchange at edges and, consequently, can strengthen or dilute edge effects on dry deposition resulting from aerodynamic processes (see chapter 2, Wuyts et al. 2008a). At the DH site, for SO_4^{2-}, NO_3^-, and NH_4^+, the presence of the gradual edge decreased the level of deposition enhancement at the edge and, unlike for Cl^-, also the penetration depth of the edge effect. For these ions, the presence of the gradual edge resulted in a significant decrease in the extra edge flux generated by these edge effects. Throughfall deposition of these ions was still lower in case of a gradual edge when the extra deposition in front of the forest edge, on the gradual edge vegetation, was taken into account, although this was statistically significant only in summer. The decrease in DEI for throughfall deposition of SO_4^{2-}, NO_3^-, and NH_4^+ - ions that are deposited as particles and gasses - in the case of a gradual transition can not be a result of changes in aerodynamic processes as then, effects on the DEI of Cl^- throughfall deposition would have been observed. Consequently, we assume that the presence of the gradual edge vegetation also significantly decreases the penetration depth of microclimatic gradients and, hence, of the edge effects on throughfall deposition of SO_4^{2-}, NO_3^-, and NH_4^+. Moreover, the presence of the gradual edge vegetation altered the ratios of NH_4^+ and NO_3^- throughfall deposition to SO_4^{2-} throughfall deposition in the edge, the latter we used as an indicator of total deposition of ions that are deposited as gasses and fine particles, because uptake of S is presumed to be balanced by canopy leaching of SO_4^{2-} (Lindberg and Lovett 1992). At 16-64 m from the edge, edge transition had no influence on these ratios, but at the edge, these ratios were higher in the case of a gradual transition than in the case of a steep transition. When considering SO_4^{2-}, NO_3^-, and NH_4^+ individually, at the steep forest edge of DH, differences in the MEI between SO_4^{2-} on the one hand and NO_3^- and NH_4^+ on the other most probably resulted from enhanced rates of canopy exchange processes at the edge of the forest, such as canopy uptake of NO_3^- and NH_4^+. These differences in MEI were however much smaller in the case of the gradual transition. These results indicate that, at the edge, the presence of the gradual transition probably alters the rate of canopy exchange: at the gradual edge, canopy uptake of NH_4^+ and NO_3^- was still higher at the edge than in the forest interior, but the differences between edge and interior canopy uptake were probably smaller. Consequently, in the forest edge, the decrease in dry deposition of NH_4^+ and NO_3^- (as indicated by SO_4^{2-} throughfall deposition) resulting from the presence of a gradual transition was probably overshadowed by a decrease in canopy uptake of NH_4^+ and NO_3^-.

Our findings agree well with those of Agster and Ruck (2003) on inclined forest edges providing a better shelter for wind throw to a forest stand than steep forest edges do. Didham and Lawton (1999), Cadenasso and Pickett (2001), and Weathers et al. (2001) found field plant seed fluxes into the forest, edge effects on microclimate and vegetation structure, and edge effects on throughfall deposition to penetrate deeper into the forest at open edges (without shrubs, saplings, or other understory vegetation) than at closed edges (characterized by a wall of dense vegetation). Next to a lower penetration depth, Cadenasso and Pickett (2001) also observed a lower magnitude of seed flux at the closed edge than the open one. Weathers et al. (2001), on the contrary, found a higher level of inorganic N and SO_4^{2-} throughfall deposition at the closed edge than at the open edge, a result of the deposition on the lower vegetation that occurred in the closed, unthinned edges but was lacking in the open, thinned edges. In this way, gradual edges provide an even larger potential to mitigate edge effects than closed edges with a low permeability, because they also induce a decrease in the magnitude of edge effects. According to the classification by Harper et al. (2005), we can identify the potential of gradual edges as 'edge softening' and that of closed edges as 'edge sealing'.

5.5.2 The importance of the shape and size of gradual edge vegetation

Comparison of the three forest sites reveals that the shape and size of the gradual vegetation, relative to the forest stand behind it, are of great importance for the impact of gradual edges on the deposition in forests. At the DH site, the presence of the gradual edge yielded a net decrease in throughfall deposition of N and N+S, even when the throughfall deposition on the gradual edge vegetation was taken into account. So, this gradual edge vegetation represented the closest-to-ideal transition to mitigate the edge effects on throughfall deposition of SO_4^{2-}, NO_3^-, and NH_4^+. At the NB site, we found that transition type did not significantly affect the edge effect on throughfall deposition in the forest (except for NO_3^- in winter) and the extra throughfall deposition generated by edge effects in the forest edge zone. The shape of the gradual edge vegetation was similar to the one at the DH site, but the size of the gradual edge relative to the height of the forest stand was inadequate. The gradual vegetation only reached up to 10 m high, leaving the upper 15 m of the first beech trees 'unsheltered' (Fig. 5.2). As a result, the gradual vegetation was not able to deflect the wind flow completely over the forest stand, contrasting with the observations in our wind tunnel study (chapter 4). At the NH site, the presence of the gradual edge vegetation led to a vast and significant decrease in the extra throughfall deposition in the forest stand as a result of edge effects. But, when the deposition on the gradual vegetation was also taken into consideration, the net effect of the gradual vegetation was insignificant. Thus, in the case of the gradual edge, the edge effects were similar to those induced by the steep edge, but were shifted upwind over a distance more or less equal to the depth of the gradual vegetation. The deposition enhancement at the first tree row of the gradual vegetation was slightly smaller than at the steep edge (particularly in summer, see Fig. 5.4), due to the smaller jump in vegetation height at the gradual vegetation. Nonetheless, the shape of the gradual edge was insufficient for mitigation of edge effects on deposition as the change in vegetation height at the gradual vegetation was too abrupt.

5.5.3 Methodological considerations

In Flanders, consciousness of the need for specific edge management is growing but gradual edges are only recently being created, mainly at newly established forests. Hence, most of these recent gradual edge vegetations and/or forests were not sufficiently developed for the purpose of this study. Furthermore, we had stand homogeneity, stand size, and edge orientation conditions for the selection of forest stands. Therefore, the number of stands suitable for this study was restricted, moreover, the selection of replications (i.e., gradual edges with analogous shape and size) was unfeasible. Nonetheless, our results provide a sufficient confirmation of the results of our wind tunnel experiment (chapter 4), which indicated the potential in gradual edge vegetation for mitigating edge effects on throughfall deposition.

Although mean wind speed was higher during the winter period, the influence of transition type was similar in winter and summer: the attenuations of DEI and MEI at the gradual transitions relative to the DEI and MEI at the adjacent steep transitions were similar for winter and summer, except for the NB site, where no consistent decrease in edge effect was observed for the gradual edge. So, the mitigating effect of a gradual edge, when expressed as the relative decrease in DEI and MEI in comparison with a steep edge, did not depend on the measuring period. Given this result, we can presume that, with a year-round sampling campaign, similar relative attenuations of DEI and MEI would have been obtained as those provided by our six-month measuring campaign.

5.5.4 Implications for management

To conserve and promote biodiversity, Magura (2002) and Wermelinger et al. (2007) suggest that at forests with poorly developed, abrupt edges, efforts should be made to create gradual, more diverse edges (i.e., 'high quality edges'). Our results support the idea of creating gradual edges by adjusted forest edge management, as these edges can profoundly influence the extent of edge effects on atmospheric deposition of N and N+S. Gradual edges can be achieved by stimulating natural regeneration or by planting and sowing shrubs and herbs (i) in the open land in front of the steep edge of the forest after, for example, retraction of the fence ('forest expansion') or (ii) in the first meters of the forest stand after cutting trees ('forest receding'). The first management option would be preferential, because the area of forest interior habitat is higher than in the second option, but this management choice is determined primarily by how the forest is embedded in the landscape matrix and by local land use pressure. In contrast with forest type, which affects throughfall deposition in the forest edge (see chapter 2 and 3, Wuyts et al. 2008a, 2008b) and in the forest interior (De Schrijver et al. 2004, 2008), the presence of a gradual edge vegetation can only influence throughfall deposition in the forest edge. However, for the purpose of mitigating edge effects, gradual edge creation is a more straightforward process and easier to implement than forest type conversion.

In addition, the results of the three sites in this study emphasize the importance of the shape and size of the gradual edge vegetation relative to the forest stand behind it for reducing edge effects on atmospheric deposition. Gradual edge vegetation that smoothly and closely links up with the forest stand behind it, as the one at the Dombergheide site, generates the ideal transition to mitigate these edge effects. An edge vegetation that is not completely gradual or smooth, i.e., with a rough transition of vegetation height due to the absence of an herbaceous fringe, a shrub belt, or a forest mantle, is less or even not effective at mitigating edge effects on atmospheric deposition.

6 The effect of LAI and other edge structure characteristics on edge patterns of throughfall deposition

6.1 Abstract

We assessed the effect of edge structure characteristics, and leaf area index (LAI) in particular, on edge patterns of throughfall deposition of Cl^-, nitrogen (N, NO_3^- + NH_4^+), and the potentially acidifying pollutants SO_4^{2-}, NO_3^-, and NH_4^+. The aim of this study was to deduce best-management practices for mitigating edge effects on throughfall deposition. Therefore, we collected throughfall water along transects plotted across the edges of six *Pinus nigra* ssp. *laricio* Maire and *P. sylvestris* L. stands. In addition, one stand of P. nigra ssp. laricio was experimentally thinned to encompass three levels of LAI. LAI was found to relate significantly to the magnitude and/or penetration depth of edge effects on Cl^-, SO_4^{2-}, NO_3^-, and NH_4^+ throughfall deposition, while no such relationships were observed for mean stand height, basal area, and their related structure characteristics, i.e., dominant tree height, crown depth, stand density, and volume. Within the studied LAI range, the level of deposition enhancement at the edge front increased linearly with increasing LAI. The downward trend of the penetration depth of edge effects with increasing LAI or canopy density which has been observed in wind tunnel and simulation studies, was only substantiated in the upper part of the LAI range considered. At lower LAI levels, we found the penetration depth to increase with increasing LAI, but this was probably due to the more complicated demarcation of penetration depths in sparse forests in comparison with dense ones. The entire deposition enhancement in an edge zone of fixed depth (estimated by the integrated forest edge enhancement factor) increased with LAI, but at a LAI close to 2, the increase levelled off for Cl^- and even switched to a decrease for SO_4^{2-}, NO_3^-, and NH_4^+. Although our data set is rather limited (seven forest stands) and is only based on measurements in *P. nigra* and *P. sylvestris* stands, our results imply that LAI is a key driver in the process of edge effect development and that edge effects on throughfall deposition can be reduced by well-considered edge management through manipulation of the LAI.

6.2 Introduction

In forests, throughfall deposition is known to be significantly affected by the collecting surface area, which can be expressed as leaf or plant area index (LAI or PAI) (Meyers et al. 1989; Draaijers 1993; Stachurski and Zimka 2000; Staelens et al. 2006). Dry deposition increases linearly with increasing LAI, but the increase in deposition levels off or even turns into a reduction at a certain level of LAI. In a modeling study, Meyers et al. (1989) found the deposition velocity of HNO_3 to increase with LAI, due to an increasing collecting surface, until a peak value at LAI = 7. At LAI > 7, the effect of the increasing collecting surface was surpassed by the effect of the decreasing in-canopy wind speed and exchange between the atmosphere and the forest canopy, which caused the deposition velocity to decrease with increasing LAI. Next to dry deposition, also canopy exchange is related to LAI, as indicated by the close relationship between PAI and the throughfall deposition of ions partly or mainly originating from canopy exchange (leaching) such as K^+, Ca^{2+}, and Mg^{2+} (Staelens et al. 2006). Hence, thinning of forest stands alters the level of nutrient input via atmospheric deposition as it (temporarily) decreases LAI.

Potentially acidifying and eutrophying throughfall deposition fluxes of N and S are increased at the front of forest edges by up to a fourfold, and this so-called edge effect extends to a distance of up to five times the stand height (e.g., Beier and Gundersen 1989; Draaijers 1993; Weathers et al. 2001; Devlaeminck et al. 2005; De Schrijver et al. 2007a). The level of deposition enhancement at the front of the edge is referred to as the magnitude of edge influence (MEI: Harper et al. 2005), and the penetration depth of edge effects is labelled as the depth of edge influence (DEI; Chen et al. 1995). Edge effects on throughfall deposition are influenced by edge transition type (sensu steep versus gradual forest edges) and forest type (Weathers et al. 2001; Wuyts et al. 2008a, 2008b, 2009b: chapter 2, 3, and 5), indicating the importance of edge structure characteristics. Next to forest type and edge transition type, also stand density or LAI significantly affects dry deposition patterns at edges, as put forward by modeling studies by Wiman and Ågren (1985) and Pahl (2000) and our wind tunnel study in chapter 4 (Wuyts et al. 2008c). In forest edges, the impact of LAI or stand density on deposition can be expected to be more pronounced than in forest interiors. The effect of a decrease in in-canopy wind speeds and turbulent exchange might be lower at edges because edge proximity inherently causes in-canopy wind speed and turbulence to be significantly increased in comparison with the interior (Meroney 1970; Irvine et al. 1997). However, when density or LAI is low, wind flow enters the forest edge to a deeper extent (Dupont and Brunet

2008a). The results of the study by Pahl (2000) and our wind tunnel study (chapter 4; Wuyts et al. 2008c) show that edge effects on deposition extend further in sparse forests than in dense forests, but that no differences occur in the level of deposition enhancement at the front of the edge. Wiman and Ågren (1985), however, found the deposition enhancement at the forest edge to be higher in dense forests than in sparse ones. Draaijers et al. (1994) detected a positive linear relationship between the factor by which deposition was enhanced in the entire edge zone (i.e., the zone at the forest edge in which deposition is enhanced in comparison with the forest interior deposition) on the one hand and LAI or stand density on the other hand. This effect of LAI may, however, have originated from an effect of tree species via an effect of needle morphology (Wiman and Ågren 1985; Wuyts et al. 2008c: chapter 4) as the different LAI levels in the study of Draaijers et al. (1994) coincided with different coniferous tree species. To conclude, the effect of structure characteristics such as LAI has been investigated primarily via simulation studies, and the results of the sole field study are ambiguous.

The aim of this study was to asses the effect of structure characteristics, and LAI in particular, on edge patterns of throughfall deposition of Cl^-, SO_4^{2-}, NO_3^-, and NH_4^+ in the field. Therefore, we monitored throughfall deposition of these ions along transects across edges of seven *Pinus nigra* ssp. *laricio* Maire and *P. sylvestris* L. stands. In addition, one stand of *P. nigra* ssp. *laricio* was experimentally thinned to cover three stand densities with differing LAI. To avoid confounding effects of needle morphology and forest type, the study was restricted to the above mentioned comparable coniferous forest types. We hypothesised that lower stand densities, expressed in lower LAI, give rise to higher penetration depths of edge effects on atmospheric deposition (i.e., higher DEI values) and lower levels of deposition enhancement (i.e., lower MEI values). The results of this study may enable us to deduce best-management practices in terms of the optimal level of stand characteristics, and LAI in particular, for mitigating the inflow of atmospheric N and S into forest edges.

6.3 Materials and methods

6.3.1 Site description

We selected six observational sites (site codes O1 to O6) and one experimental site with two treatments (site codes E1 and E2) in Flanders (the northern part of Belgium) and The Netherlands (Fig. 6.1). All sites were even-aged homogeneous stands of *Pinus nigra* or *P.*

sylvestris that bordered open land at the south- to westerly oriented edge (Fig. 6.1). All studied stands were situated on poor sandy soils, classified as Haplic podzols (FAO-ISRIC-ISSS 1998), in regions with high atmospheric NH_3 concentrations due to intensive livestock breeding. Data from the observational sites have been published or processed previously in function of other research questions in chapters 2, 3, 5, and 6 (Wuyts et al. 2008a, 2008b, 2009b, and submitted), and we refer to these studies for the exact location of the stands and the meteorological and chemical characteristics. In addition, two parallel, adjacent parts of the homogeneous P. nigra stand O3 were thinned in all diameter classes towards a mean basal area of 50 and 40 m² ha^{-1} in August 2005, to obtain two experimental stands, E1 and E2. For each stand, Table 6.1 presents the site codes, the references to the previous studies, the tree species considered, the structure characteristics, and the orientation of the edge.

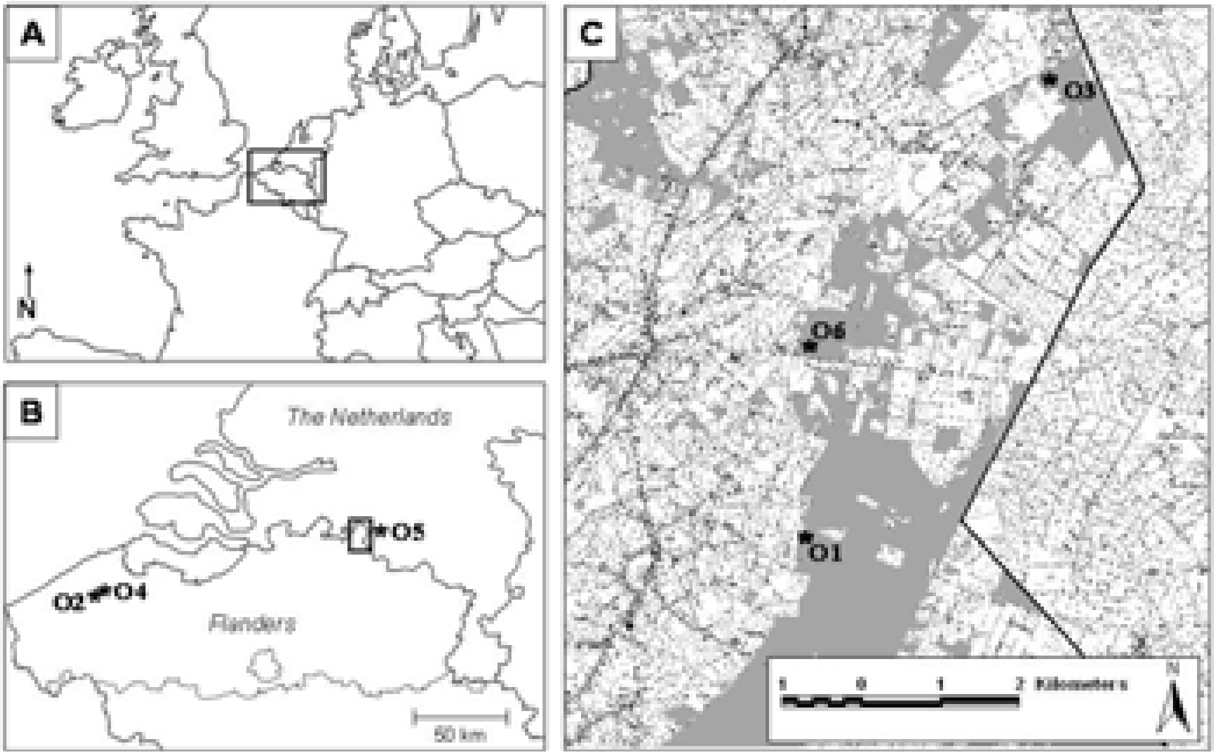

Fig. 6.1: A: Map of Western and Central Europe, with indication of the location of map B. B: Map of Flanders (the northern part of Belgium) and the south of the Netherlands; indicated are the locations of the stands O2 and O4 in the province of West Flanders (B), the stand O5 in the province of North Brabant (NL), and the contours of map C. C: Map with the location of the stands O1, O3, and O6 in the north-east of the province of Antwerp. The experimental stands E1 and E2 were established by thinning parts of stand O3 in August 2005.

Table 6.1: The observational stands considered in this study were used in other chapters; their site codes in those chapters are provided for enabling full retracement. In addition, the dominant tree species and their stand density (SN), basal area (BA), stand volume (V), mean tree height (H), dominant tree height (H_{dom}), and mean crown depth (CD) are presented. The leaf area index (LAI) was determined as a mean value of LAI measured above the throughfall deposition collectors at more than 30 m from the edge. The stands encompassed some variation in edge orientation and throughfall was collected over different time periods from 2003 to 2007. (-: not determined)

Site code	Site code in other chapters	Dominant tree species	SN (ha^{-1})	BA (m^2 ha^{-1})	V (m^3 ha^{-1})	H (m)	H_{dom} (m)	CD (m)	LAI (-)	Edge orientation	Measuring period
O1	CP in ch. 2	*P. nigra* ssp. *laricio*	739	48	414	17.5	18.7	-	3.08(1)	W	15 Jul 2003 - 30 Jul 2004
O2	Pn1 in ch. 3 & 7	*P. nigra* ssp. *nigra* var. *nigra*	388	36	336	19.2	22.0	6.23	2.04	SW	1 Sep 2005 - 30 Aug 2006
O3	Pn2 in ch. 3 & 7	*P. nigra* ssp. *laricio*	1162	55	523	18.0	19.1	5.77	1.88	SW	1 Sep 2005 - 30 Aug 2006
O4	Ps1 in ch. 7	*P. sylvestris*	458	29	287	19.8	23.2	4.62	1.82	SSW	1 Sep 2005 - 30 Aug 2006
O5	NH in ch. 5	*P. sylvestris*	393	19	125	13.0	16.0	4.35	1.32	S	22 Dec 2006 - 16 Mar 2007
											13 Jun 2007 - 11 Sep 2007
O6	Ps2 in ch. 7	*P. sylvestris*	195	20	183	19.4	21.1	6.95	1.03	SSW	1 Sep 2005 - 30 Aug 2006
E1		*P. nigra* ssp. *laricio*	887	50	478	18.8	21.3	6.41	1.31	SW	1 Sep 2006 - 16 Feb 2007
E2		*P. nigra* ssp. *laricio*	700	40	381	18.3	21.2	6.07	1.11	SW	1 Sep 2006 - 16 Feb 2007

Note: (1): LAI was re-measured in September 2008 with hemispherical photographs to avoid errors originating from different measuring techniques, and hence, this value differs from Wuyts et al. (2008a).

All stand characteristics except LAI were determined from a full stand survey along the same transects as those along which throughfall deposition was monitored, from the edge to 64 or 128 m from the edge, and within a range of 10 m at both sides of the transects. The LAI values were determined by means of digital hemispherical photographs (Nikon D70-s with fisheye lens Sigma EXDG Fisheye 8 mm 1:4 D) taken in the year in which throughfall deposition was collected. Photos were taken above the funnel of each throughfall deposition collector in the forest interior of the stands and processed with Gap Light Analyzer GLA 2.0 (www.ecostudies.org/gla). One exception is the O1 stand: at the time throughfall deposition was monitored, its LAI was estimated using the LAI-2000 Plant Canopy Analyzer (LI-COR). In order to allow proper comparison of LAI values between the stands, the LAI of stand O1 was re-measured using hemispherical photographs.

6.3.2 Experimental setup

In each of the observational and experimental stands, throughfall deposition was monitored along a transect perpendicular to the forest edge. Throughfall deposition was measured by means of two (O1) or three (O2-O6, E1, and E2) throughfall collectors at each of distance plots along the transect: eighteen distance plots up to 64 m from the edge (O1), seven distance plots at 0, 2, 4, 8, 16, 32, and 64 m from the edge (O5), or eight distance plots at 0, 2, 4, 8, 16, 32, 64, and 128 m from the edge (O2-O4, O6, E1, and E2). In all stands, identical throughfall deposition collectors were used. The collectors consisted of white polyethylene funnels (diameter 143 mm) positioned on 2 liter, white polyethylene bottles that were partially buried to protect the samples from heat and direct sunlight. In the funnels' necks, a nylon mesh prevented contamination of the sample by plant material and small animals, such as insects and spiders. Stemflow was not considered in this study because of its low contribution to the total nutrient flux to the forest floor in pine forests (Neirynck et al. 2004). Sample collection took place fortnightly to monthly between 2003 and 2007 (see Table 1). In contrast with O1-O4 and O6, where throughfall deposition was measured year-round, throughfall deposition in the O5 stand was monitored during three summer and three winter months and deposition in the experimental stands was sampled during six months from late summer to late winter. On each sampling event, the sample volume was determined for each collector, samples were pooled into a volume-weighted plot sample per distance plot (except for stand O5), and the collectors were replaced by rinsed ones. From these (plot) samples, 300 ml subsamples were taken and transported in a cooled dark box to the laboratory, where they were stored below

5°C. Subsamples from two consecutive fortnights were pooled volume-weighted into monthly samples (except for the fortnightly samples of the O1 stand), which were subsequently analysed for Cl^-, SO_4^{2-}, NO_3^-, and NH_4^+, as described in chapter 2.

6.3.3 Data analysis

For each ion, the monthly (or fortnightly) throughfall deposition flux per distance plot was calculated based on the ion concentration in the pooled monthly (or fortnightly) subsample, the monthly (or fortnightly) sample volume in all collectors of the distance plot, and the surface area covered by the funnels of all deposition collectors per plot. In stand O5, samples were not pooled per distance plot, but deposition fluxes were determined for each collector individually and averaged per distance plot. For each ion and stand individually, the throughfall deposition fluxes summed over the entire measuring period were used to calculate the depth of edge influence (DEI), the magnitude of edge influence (MEI), and the integrated forest edge enhancement (IFEE) factor. The IFEE factor is the level at which throughfall deposition is enhanced in an edge zone of a fixed length and accounts for both DEI and MEI (chapter 2). The DEI was calculated as described by Beier and Gundersen (1989): we delineated two distance groups or parts in the plot of ln-transformed throughfall deposition vs distance from the edge, one group or part displaying a linear relationship with a slope < 0 and the other with a slope close to 0. The MEI was calculated as the ratio of the mean throughfall deposition flux at the first two distance plots of the edge to the mean throughfall deposition flux at the end of the transect. If a forest edge zone could be delineated based on Beier and Gundersen's (1989) method, the IFEE factor was determined. We calculated the IFEE factor as the ratio of the throughfall deposition flux that reaches the forest floor in the first 64 m of the edge to the throughfall deposition flux that would reach the forest floor in the same area in the absence of a forest edge effect. The same methods for the calculations of MEI, DEI, and IFEE were applied for all studied stands.

Edge orientation and meteorological conditions, e.g., wind speed and wind direction, are co-determinants for the magnitude of edge effects (Draaijers et al. 1988; Wuyts et al. 2008a, chapter 2). Throughout the seasons, mean wind speed and the prevailing wind direction change and, consequently, variations in edge effects occur. This implies that, when edge effects are compared, they should be determined within the same period and with the same time-span. Because of the 'evergreen' character of pine species, no significant seasonal variability due to changes in forest characteristics is expected, this in contrast with deciduous tree species. The measuring period of two times three months in winter and summer at the O5 stand was considered to be representative for a year-round measuring period because the wind speed during the measuring campaign was representative for the entire year (Wuyts et al. 2009b, chapter 5). At the experimental stands E1 and E2, throughfall deposition was monitored mainly during autumn and winter, when wind speed is highest and wind direction is mainly south to west (thus perpendicular to the forest edge), and when, consequently, the most pronounced edge effects on throughfall deposition arise (Wuyts et al. 2008a, chapter 2). Therefore, the MEI and IFEE factors measured in the adjacent reference stand during the same six months (from September 2006 to February 2007; O3' in Table 2) were used to linearly recalculate the six-months MEI and IFEE factors of stands E1 and E2 to year-round values based on the previously assessed year-round factors of the same stand (O3, from September 2005 to August 2006). Since rescaling of the DEI was not possible, those values were retained. The variability caused by considering different measuring years was not excluded from the data, but since the variation of mean annual wind speed is small (e.g., mean wind speeds in 2006 and 2007 were 0.4 m s^{-1} lower than the long term mean wind speed according to the Royal Meteorological Institute of Belgium, www.kmi.be), the variability is expected to be low. Also the variability induced by small differences in edge orientation was not omitted from the dataset.

Variation in structure characteristics between stands was summarized by means of principal components analysis (PCA) in CANOCO 4.5 ordination software. The data were log-transformed and the axes scores of the structure characteristics were post-transformed by dividing them by their standard deviation. Because of this post-transformation, ordination diagrams are correlation bi-plots in which the angles between arrows represent correlations, i.e., small angles correspond to high intercorrelations (Lepš and Šmilauer 2003). In this way, independent variables are identified that represent groups of structure characteristics.

Next, linear, inverse, logarithmic, exponential, and quadratic regressions were performed between the independent structure characteristics on the one hand and the DEI, MEI, and IFEE factors on the other, for Cl^-, SO_4^{2-}, NO_3^-, and NH_4^+ individually.

6.4 Results

In all stands, edge effects on throughfall deposition were appreciable for all ions considered, except in the O4 stand, where edge effects only occurred on Cl^- throughfall deposition and not on SO_4^{2-}, NO_3^-, and NH_4^+ throughfall deposition (Fig. 6.2). The forest edge of this stand borders a busy motorway and, although wind speed was noticeably enhanced at the edge relative to the forest interior, it is probable that trucks passing by at less than 5 m from the edge considerably disturbed the expected pattern of wind speed and turbulence as at the forest edge. Because edge effects were considerably altered by an external factor rather than a factor originating from the forest stand itself, the data from the O4 stand were excluded from further analyses.

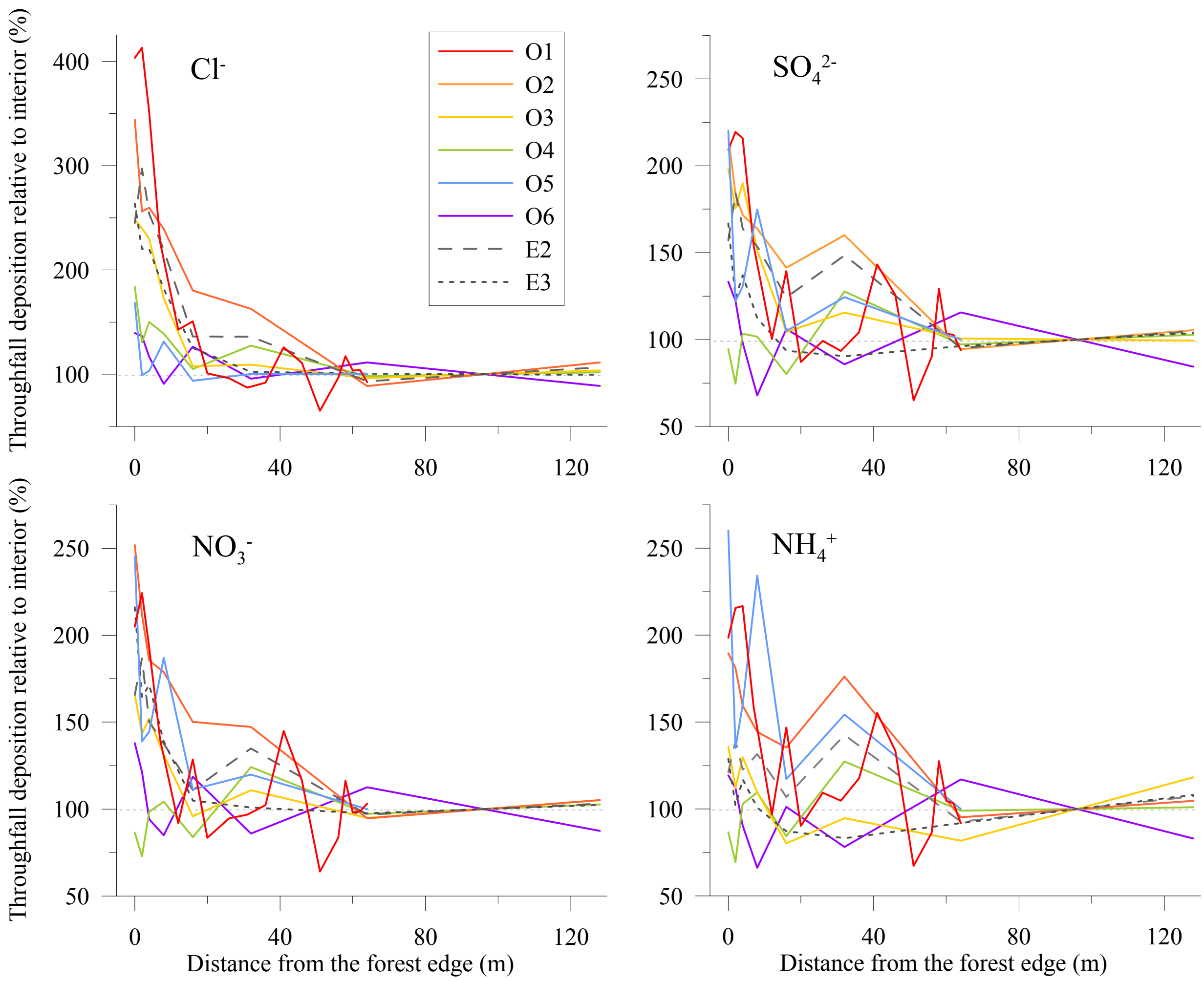

Fig. 6.2: Patters of throughfall deposition of Cl^-, SO_4^{2-}, NO_3^-, and NH_4^+ at the edges of the studied forest stands, expressed as a percentage of the assumed interior deposition (calculated as the mean of the throughfall deposition at 64 and 128 m from the edge, except in O1 and O5; in stand O5, the interior deposition was set equal to the deposition at 64 m and in stand O1, the interior deposition was calculated as the mean deposition at 60 to 64 m from the edge).

The magnitude of edge influence (MEI), the depth of edge influence (DEI), and the integrated forest edge enhancement (IFEE) factors of the edge effects on Cl^-, SO_4^{2-}, NO_3^-, and NH_4^+ throughfall deposition are presented in Table 6.2. The MEI and IFEE factors of the edge effects in the stands O3, E1, and E2 for the six-month measuring period are given in italic font. We encountered some difficulties with the demarcation of the DEI in the sparser forests (O6 and O5) due to the high variability of throughfall deposition along the transects.

Table 6.2: The magnitude of edge influence (MEI; -), the depth of edge influence (DEI; m), and the integrated forest edge enhancement factor (IFEE; -) describing the edge effects on throughfall deposition fluxes of Cl^-, SO_4^{2-}, NO_3^-, and NH_4^+ in the observational (O1-O6) and experimental (E1 and E2) study sites. Values in italic font are assessed from the data of the six-months measuring period (from September 2006 to February 2007), values for E1 and E2 in normal font are rescaled year-round values based on the data of stand O3 and O3'. (-: no clear edge effect was detected, so the IFEE factor was not determined)

Site	Cl^-			NO_3^-			SO_4^{2-}			NH_4^+		
	MEI (-)	DEI (m)	IFEE (-)	MEI (-)	DEI (m)	IFEE (-)	MEI (-)	DEI (m)	IFEE (-)	MEI (-)	DEI (m)	IFEE (-)
O1	4.06	31	1.79	2.10	12	1.12	2.03	12	1.10	2.00	12	1.10
O2	3.00	64	1.95	2.32	64	1.56	2.00	64	1.45	1.85	64	1.40
O3	2.44	64	1.62	1.55	64	1.31	1.87	64	1.40	1.24	64	1.26
O4	1.57	16	1.08	0.80	-		0.85	-		0.78	-	
O5	1.56	48	1.26	1.87	48	1.32	1.89	53	1.42	1.99	53	1.47
O6	1.38	8	1.02	1.30	4	1.01	1.28	4	1.01	1.16	4	1.00
E1	*2.71*	*64*	*1.52*	*1.76*	*16*	*1.10*	*1.71*	*64*	*1.25*	*1.30*	*64*	*1.10*
E2	*2.42*	*32*	*1.28*	*1.90*	*16*	*1.11*	*1.45*	*16*	*1.05*	*1.15*	*32*	*0.98*
O3'	*3.35*	*32*	*1.46*	*2.31*	*64*	*1.35*	*2.01*	*64*	*1.37*	*1.58*	*64*	*1.25*
E1	1.97	64	1.68	1.18	16	1.06	1.59	64	1.28	1.02	64	1.11
E2	1.76	32	1.42	1.28	16	1.07	1.35	16	1.07	0.90	32	0.99

The PCA analysis identified three groups of structure characteristics with significant inter-group correlations, i.e., small angles between variables (Fig. 6.3). The first group included basal area (BA), stand density (SN), and stand volume (V), which were highly intercorrelated. The second group represented mean tree height (H), dominant tree height (H_{dom}), and mean crown depth (CD). Leaf area index (LAI) was not related to the former two groups. Consequently, all seven variables can be represented by the BA, H, and LAI.

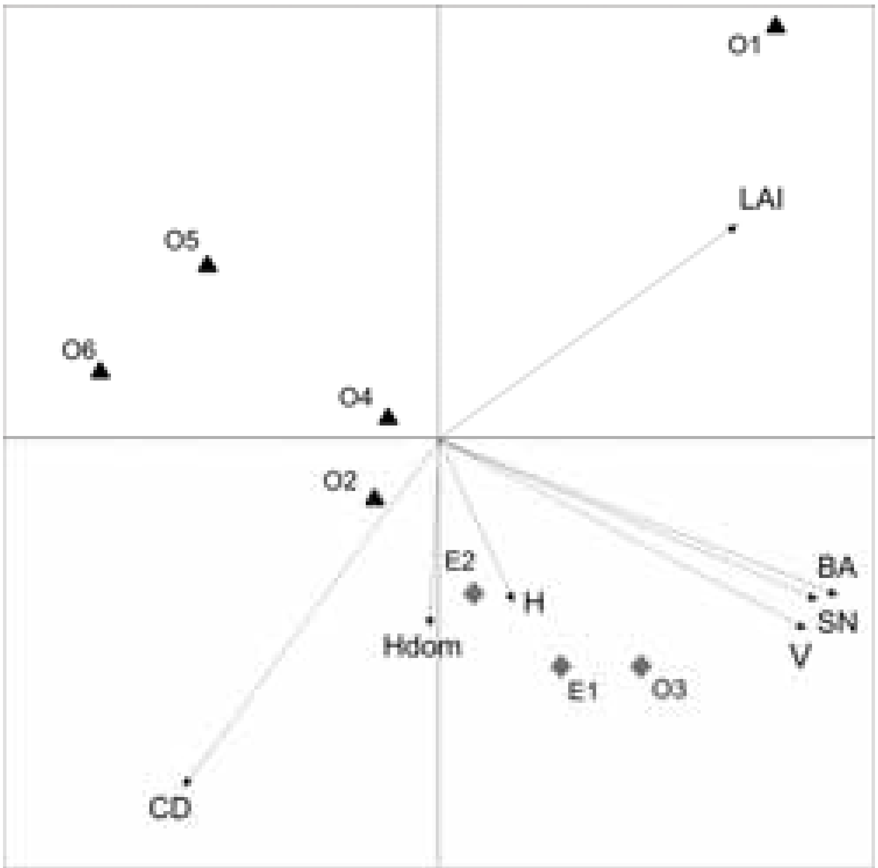

Fig. 6.3: Correlation bi-plot of a PCA analysis on the structure characteristics. The first two ordination axes are shown and account for 58 % and 34 % of the total variability in characteristics between stands, respectively. Grey diamonds represent the three parts of the stand of which two parts were experimentally thinned (E1 and E2) and one part remained unaffected (O3).

We investigated the relationship between these independent variables BA, H, and LAI on the one hand and DEI, MEI, and IFEE on the other hand. For BA, we found no significant relationships with MEI and DEI for any ion; with IFEE, BA showed a marginally significant log-relation ($p = 0.055$) only for Cl^- throughfall deposition. No significant relations between H and DEI, MEI, and IFEE were encountered.The variable LAI, however, showed significant linear and inverse relations with MEI and significant quadratic or inverse relations with DEI and IFEE ($n = 7$; Fig. 6.4; Table 6.3). For all the studied ions, when LAI increased within the range of 1.0-3.0, (i) the MEI increased, while (ii) the DEI increased until LAI approximately equalled 2.0, but tended to decrease again at higher LAI values (Fig. 6.4). Because of the difficult and therefore less reliable demarcation of the DEI in sparse forests, curves were fitted to the DEI values of the stands with LAI > 1.30, i.e., without stands O6 and E2 ($n = 5$; Fig. 6.4; Table 6.4). For these fits, linear curves were chosen because of the low number of stands. The IFEE factor, which incorporates both MEI and DEI, increased with increasing LAI, but at higher LAI values, the increase levelled of (in the case of Cl^-) or even turned into a decrease (in the case of SO_4^{2-}, NO_3^-, and NH_4^+; Fig. 6.4).

The strongest relationships between MEI, DEI, and IFEE on the one hand and LAI on the other hand were observed for Cl^- and SO_4^{2-} throughfall deposition. The relationships of MEI, DEI, and IFEE with LAI differentiated between Cl^- on the one hand and SO_4^{2-}, NO_3^-, and NH_4^+ on the other hand. Firstly, the relationship of the MEI of the edge effects on Cl^- throughfall deposition with LAI was linear while the increase in MEI for SO_4^{2-}, NO_3^-, and NH_4^+ levelled of to a limit of MEI close to 2.5 (Table 6.3). Secondly, at higher LAI levels, the IFEE factor for SO_4^{2-}, NO_3^-, and NH_4^+ decreased with increasing LAI, while the increase in IFEE factor for Cl^- only levelled of and did not yet turn into a decrease (Fig. 6.4).

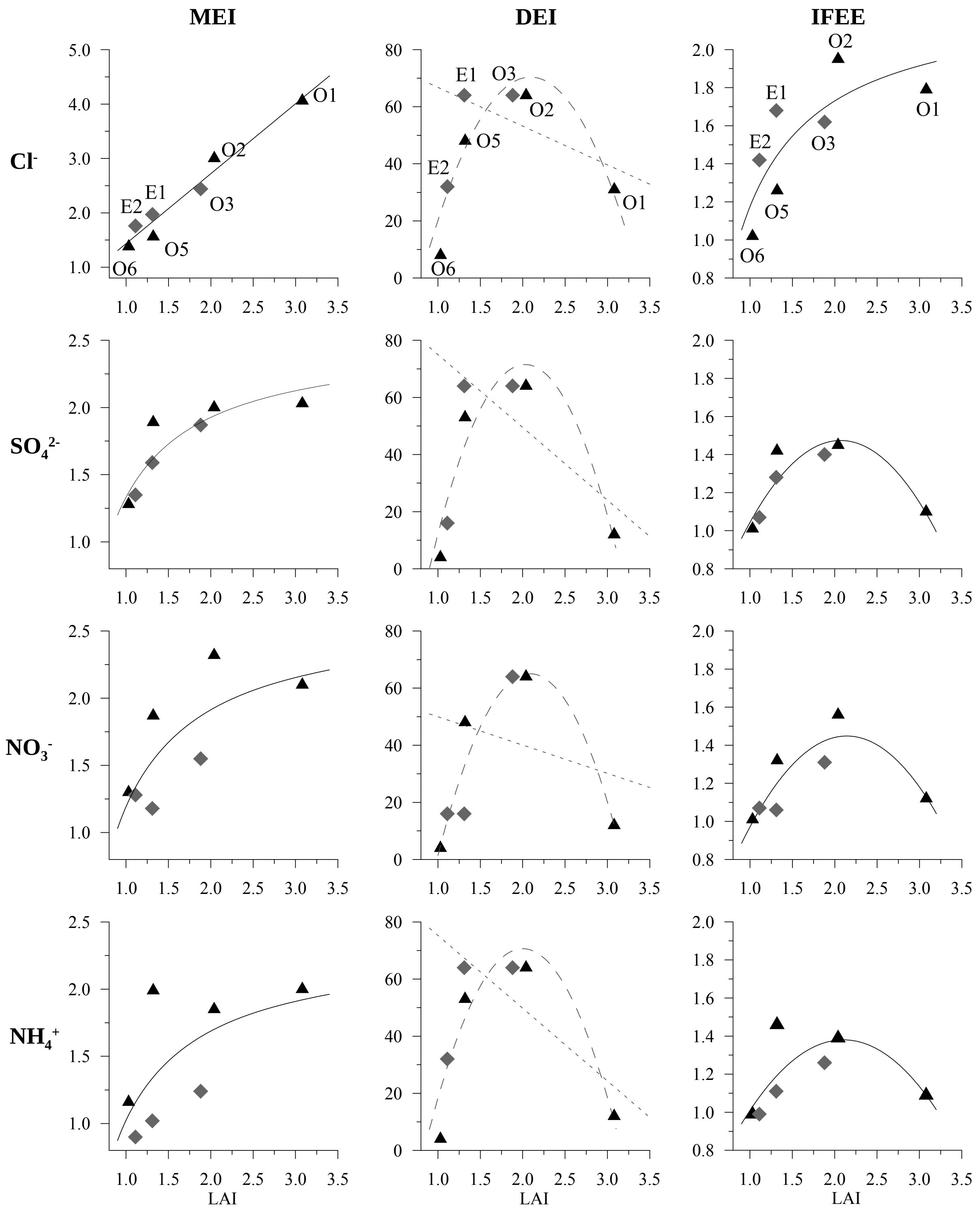

Fig. 6.4: The magnitude of edge influence (MEI), the depth of edge influence (DEI), and the integrated forest edge enhancement factor (IFEE) describing the edge effects on the throughfall deposition fluxes of Cl^-, SO_4^{2-}, NO_3^-, and NH_4^+, plotted against the stands' LAI. Curves indicate the best linear (MEI and DEI), quadratic (DEI and IFEE), or inverse (IFEE) fits to the data. Due to the hampered demarcation of DEI in sparse forests, two fits with LAI are presented: (i) a quadratic fit based on all stands except O4 (n = 7) and (ii) a linear fit based on all stands with LAI > 1.30 except O4 (n = 5). Grey diamonds represent the three parts of the stand of which two parts were experimentally thinned (E1 and E2) and one part remained unaffected (O3).

Table 6.3: Best linear, quadratic, and/or inverse fits to the magnitude of edge influence (MEI), the depth of edge influence (DEI, m), and the integrated forest edge enhancement factor (IFEE) in function of the leaf area index (LAI), for Cl^-, SO_4^{2-}, NO_3^-, and NH_4^+ (n = 7: all stands except O4).

Factor	Ion	R^2	P	Relation
MEI	Cl^-	0.960	<0.001	1.285*LAI + 0.150
	SO_4^{2-}	0.815	0.005	-1.189/LAI + 2.521
	NO_3^-	0.580	0.047	-1.445/LAI + 2.636
	NH_4^+	0.428	0.111	-1.331/LAI + 2.353
DEI	Cl^-	0.762	0.057	-42.12*LAI^2 + 176.08*LAI - 113.64
	SO_4^{2-}	0.812	0.035	-56.05*LAI^2 + 227.49*LAI - 159.31
	NO_3^-	0.876	0.015	-53.83*LAI^2 + 224.71*LAI - 169.47
	NH_4^+	0.808	0.037	-52.20*LAI^2 + 208.93*LAI - 138.42
IFEE	Cl^-	0.656	0.027	-1.109/LAI + 2.285
	SO_4^{2-}	0.823	0.031	-0.376*LAI^2 + 1.557*LAI - 0.138
	NO_3^-	0.749	0.063	-0.365*LAI^2 + 1.564*LAI - 0.227
	NH_4^+	0.527	0.224	-0.305*LAI^2 + 1.283*LAI + 0.031

Table 6.4: Best linear fit to the depth of edge influence (DEI, m) in function of the leaf area index (LAI) based on all stands with LAI > 1.30 (n = 5: all stands except O4, O6, and E3)

Ion	R^2	P	Relation
Cl^-	0.445	0.219	-13.56*LAI + 80.32
SO_4^{2-}	0.669	0.091	-25.48*LAI + 100.47
NO_3^-	0.080	0.645	-9.91*LAI + 59.88
NH_4^+	0.669	0.091	-25.48*LAI + 100.47

6.5 Discussion

6.5.1 Impact of structure characteristics at the edge

The magnitude and penetration depth of edge effects on throughfall deposition in a forest are related to the leaf area index (LAI) of the forest stand. Unlike in the field study by Draaijers (1993) and the simulation study by Pahl (2000), we found no consistent relation between the edge effect measures DEI, MEI, and IFEE on the one hand and the basal area (BA) or the mean tree height (H) on the other hand. Consequently, there were no significant relationships between the edge effect measures and the structure characteristics represented by BA and H, i.e., stand density N, stand volume V, dominant height Hdom, and mean crown depth CD. The lack of correlation with H was probably caused by the relatively small range of stand heights considered, i.e., 17.5-19.4 m with exception of one stand with H = 13 m. In our wind tunnel study (chapter 4, Wuyts et al. 2008c), an increase in stem density altered the penetration depth of the edge effect on Cl^- deposition. However, since stem density was uniformly related to LAI, a decrease in stem density coincided with a similar decrease in LAI and thus, the effect encountered from stem density on edge patterns of deposition was in fact an effect of LAI. In this field study, on the contrary, similar levels of LAI occurred in forests with both high and low BA values and no significant correlation between LAI and BA occurred.

As was mentioned in the material and methods section, the variability caused by considering different years and induced by small differences in edge orientation was not omitted from the dataset. From the smooth trend of the MEI values with LAI and the fairly good fits to the relationship for Cl^- and SO_4^{2-}, ions that can be regarded as indicators of dry deposition because their canopy exchange is considered to be negligible (Ivens 1990; Lindberg and Lovett 1992; Houle et al. 1999), we can assume that the differences in edge orientation and monitoring time had only minor influences on the edge patterns of throughfall deposition.

6.5.2 Effect of LAI on the magnitude of edge influence

Our results indicate that the level at which throughfall deposition is enhanced at the outer edge of the forest increases with increasing LAI. An explanation for these results can be found in the study by Meyers et al (1989) and our wind tunnel study describer in chapter 4 (Wuyts et al. 2008c). From Meyers et al. (1989), we learn that a change in LAI in the forest interior has an impact on two processes that influence deposition on the forest canopy. When LAI increases, the collecting surface area is enlarged, but in-canopy wind speeds are reduced. The latter is supported by wind speed measurements in the field and in model forests in the wind tunnel and by model simulations (Gardiner et al. 1997; Novak et al. 2000; Wuyts et al. 2008c, chapter 4). Dependent on the level of LAI increase, one of the two process shifts dominates the overall outcome on the deposition on the forest canopy. At the forest edge, the same increase in collecting surface area occurs as in the forest interior, but the effect of an increase in LAI on in-canopy wind speeds is less important. Wind speed at the front of the edge (i.e., at the first tree row) seems independent of LAI: wind tunnel studies and numerical simulations by Dupont and Brunet (2008a) and in chapter 4 showed that horizontal and vertical wind speeds at the front edge of forests with differing LAI are similar. Thus, with an increase in LAI, in-canopy wind speeds at the edge are decreased to a lesser extent in comparison with the forest interior, but the collecting surface area increases equally, resulting in a stronger deposition increase at the edge than in the interior and thus in a higher MEI.

The increase in the level of deposition enhancement at the edge front due to higher leaf area or canopy density in our field study was larger than in a numerical simulation study by Pahl (2000) and even opposite to the (insignificant) decrease in deposition enhancement in the wind tunnel study (chapter 4, Wuyts et al. 2008c). An explanation must be sought in the differences between the simulation studies and the field experiments. Firstly, the simulation studies considered only edge effects on dry deposition (DD) originating from aerodynamic processes, while edge effects on DD and canopy exchange (CE) originating from microclimatic gradients were not taken into account. However, since the differences between the studies were also observed for (throughfall) deposition of Cl^-, which originates mainly from DD of coarse particles and is negligibly affected by CE, the absence of edge effects on DD and CE originating from microclimatic gradients in the simulation studies can not explain the smaller effect of LAI on MEI. Secondly, the effect of LAI on MEI can be strengthened when LAI is related to other structure characteristics, such as crown depth. In our field studies however, these confounding effects did not occur. Thirdly, shape and crown depth were different between the model trees used in the simulation studies and the real-life *P. nigra* or *P. sylvestris* trees in our field study. In the wind tunnel study, the canopy of the model trees was triangularly shaped and amounted to 0.6 and 0.8 times the tree height, with the main fraction of the collecting surface occurring in the lower part of the tree's crown. In the study by Pahl (2000), the trunk space was also small (i.e., only one third of the tree height) and provided with undergrowth vegetation. In contrast, real-life pine trees in a forest setting have small crowns (in this study, on average, one third of the mean tree height) and, in contrast with the model trees from the wind tunnel study, the collecting surface is relatively evenly distributed throughout the crown. Numerical simulations by Dupont and Brunet (2008a) indicated that, in forests with a large, sparse trunk space which is typical of pine trees in forest settings, the sub-canopy wind jets occurring in the trunk space is much larger and decays much slower than in forests with a small or dense trunk space. Due to this deeply penetrating sub-canopy wind jet, small vertical wind flows occur from the large, sparse trunk space to the lower part of the tree crowns (Dupont and Brunet 2008a), herewith probably increasing dry deposition on the canopy as far as the sub-canopy wind jet extends into the forest. Consequently, in our *Pinus* forests and in contrast with the above mentioned simulation studies, enhanced (vertical) wind speed and turbulent transport from the trunk space to the lower part of the canopy may cause a significant additional edge effect on deposition which is also affected by LAI. Fourthly, it is plausible that at higher LAI values, the linear increase in MEI with LAI may level off for Cl^-, just like observed for SO_4^{2-}, NO_3^-, and NH_4^+, and that Pahl (2000) and we, in our wind tunnel study (chapter 4), considered edge effects in the range of LAI where the increase in MEI already levels off or even switches to a decrease. In the study by Pahl (2000), the increase in MEI showed some significant levelling off, even at the lowest LAI values considered. In our study, indications for the LAI effect to level off at higher LAI values is observed for SO_4^{2-} NO_3^-, and NH_4^+ in Fig. 6.4. Hence, we can suppose that, at LAI values higher than the ones measured in our study, the canopy becomes this dense that the wind flow is blocked off at the edge front and, after all, the effect of decreasing in-canopy wind speeds will eventually partake at the front of the edge.

The clearest relationships between LAI and MEI were found for Cl^- and SO_4^{2-} throughfall deposition. From the four ions considered in this study, these are the ones regarded as (nearly) inert with respect to canopy exchange processes (Lindberg and Lovett 1992; Stachurski and Zimka 2000). For NO_3^-, and particularly NH_4^+, the relationships of MEI with LAI were probably hazed by the influence of canopy exchange processes. Nonetheless, the patterns of MEI with increasing LAI differed even between Cl^- and SO_4^{2-}. Within the LAI range considered, the increase in MEI with increasing LAI was linear for Cl- while for SO42-, the increase in MEI levelled of to a maximum of about 2.2. At lower LAI values, MEI values were similar for Cl^- and SO_4^{2-}, but at higher LAI values, edge effects generated a MEI that is about two times higher for Cl^- than for SO_4^{2-}. We do not have a thorough, justified explanation for this, but we suggest that this may be attributed to the different origin of dry deposition of these ions: Cl^- is mainly deposited as coarse particle, while S is mostly deposited as gas or fine particle. For coarse particles, it is assumed that in-canopy wind speeds predominantly determine the difference between edge and interior deposition, while for gasses and fine particles, differences between edge and interior are mainly linked to differences in turbulence (Draaijers 1993). In our previous wind tunnel study (chapter 4), an increase in stand density or LAI did not result in significantly different changes in turbulence between the front of the edge and the interior of the edge. Consequently, it seems that for deposition of gasses and fine particles, the effect of a lower wind speed in the interior is weakened by the unaffected turbulence. But as the wind tunnel study only covered two LAI values, this explanation is most hypothetical.

6.5.3 Effects of LAI on the depth of edge influence

In the study by Pahl (2000) and our wind tunnel study (chapter 4), abruptly edged model forests with lower densities (and thus lower LAI) were subject to edge effects that penetrated further in the forest in comparison with higher density forests (with a higher LAI). In our field study, a decrease in DEI with increasing LAI was only observed at higher LAI levels. This downward trend of the DEI with LAI is a result of the stronger deceleration of wind flow entering the forest edge and the accelerated dissipation of turbulence in dense forests in comparison with sparse forests (Dupont and Brunet 2008a; Wuyts et al. 2008c, chapter 4). At lower LAI values, however, our study points to an illogical increase in penetration depth of the edge effect on throughfall deposition with increasing LAI, which was most probably due to methodological restraints. Firstly, the method of Beier and Gundersen (1989), used for the delineation of the penetration depth of edge effects, is a very approximate and subjective method when throughfall deposition is measured in low spatial resolution. Throughfall deposition was measured on a fine spatial scale within the first fifteen meter of the edge, where throughfall deposition alters the most with distance from the edge, but further from the edge, the sampling resolution was too low for an accurate demarcation of DEI. Secondly, the size of the forest stands limited the length of the transects and the delineation of DEI. Thirdly, in very sparse forest stands such as O6 and E3, the correct delineation of the penetration depths of edge effects is hampered and more complex than in dense forests The reason is threefold: (i) the spatial variability of throughfall deposition within the stand is high because the canopy is not continuously closed, (ii) according to Pahl (2000), deposition velocity increases again locally, at about five edge heights (i.e., about 100 m from the edge), due to the occurrence of an eddy, and this increase is more distinct in sparse forests than in dense ones, and (iii) in sparse forests, the contrast between edge and interior deposition is smaller than in dense forests, with the result that, several meters from the edge, deposition is more likely not to be recognized as enhanced relative to interior deposition. Consequently, the quadratic fit to DEI in function of LAI (see Fig. 6.4) is not an acceptable representation of the impact of LAI on the penetration depth of edge effects.

However, not all DEI values in this study are less reliable: in the relatively dense stand O1, throughfall deposition was measured every 2-5 m, so patterns of edge effects were very distinct which enabled an accurate demarcation of the DEI. In addition, in the more dense forests O2, O3, and E2, the demarcation of DEI was unambiguous because of clearly delimited edge effects, but it was less precise than at stand O1 due to the lower sampling resolution in that region. Nonetheless, we are sure that the DEI values in these stands were higher than in the O1 stand. From these results, together with the results of the above mentioned simulation studies, we can conclude that an overall decreasing trend of DEI with increasing LAI would be a more appropriate representation of the impact of LAI on the penetration depth of edge effects. It is however unlikely that the linear relationship between DEI and LAI, as was fitted to the data of the stands with LAI > 1.30 (Fig. 3), is continued at very low and very high LAIs. Instead, we assume that the DEI asymptotically approximates 0 in very dense forests with high LAI and asymptotically approaches ∞ in very sparse forests with low LAI, resembling solitary trees in open areas.

6.5.4 Effect of LAI on the integrated forest edge enhancement factor

The IFEE factors, which encompass both DEI and MEI, linearly increase with increasing LAI, but the pattern of IFEE for Cl^- showed an indication that the increase in the IFEE factor levels off at higher LAI values, i.e., from LAI = 2 onwards. Moreover, for SO_4^{2-}, NO_3^-, and NH_4^+, the increase in IFEE switched to a decrease in IFEE at LAI = 2. This was the result of the larger impact of LAI on MEI for Cl^- than for SO_4^{2-}, NO_3^-, and NH_4^+. It can be expected that, at even higher values of LAI, the increase in MEI will be surpassed by the decrease in DEI, resulting in a declining IFEE factor for Cl^- with increasing LAI.

The repercussions of the above mentioned hampered demarcation of DEI in sparse forests on the IFEE factor are expected to be small: the increase in deposition that is not covered when calculating the IFEE based on the smaller DEI values is relatively small in comparison with the deposition increase in the first meters from the forest edge. Consequently, we can assume that the IFEE values at lower LAI values were underestimated only to a small extent. So, the calculated IFEE values, as overall indicators for edge effects on throughfall deposition, are sufficiently correct estimates for assessing the influence of LAI on the overall increase of deposition in the entire edge zone. Moreover, the curves' shapes in Fig. 4 are as expected. Given that (i) MEI increases linearly with LAI, and the increase might level off at higher LAI values, and (ii) DEI decreases with increasing LAI with asymptotic approximations of 0 at high LAI and ∞ at low LAI, clock-shaped curves can be expected, with a maximum at a moderate value of LAI. This is, however, not in accordance with the results of the study performed by Draaijers (1993), who found the overall increase of deposition in the first five edge heights from the edge (i.e., the WEINTE factor) to be linearly and positively related to LAI. Yet, LAI variation in the study of Draaijers (1993) originated partly from differences between tree species: differences in LAI coincided with differences in needle morphology and tree canopy profile. As a result, the effects of needle morphology (Wiman and Ågren 1985) and canopy profile (Dupont and Brunet 2008a) may have interfered with the effect of LAI. In this study, we limited the confounding of effects by selecting only *Pinus nigra* and *P. sylvestris* forest edges. From the relation of the IFEE factor with LAI, we can conclude that canopy density is a driver in the processes causing edge effects on atmospheric deposition.

6.6 Conclusion

Leaf area index (LAI), as a measure for canopy density, is a key driver in the processes causing edge effects on atmospheric deposition as it determines the magnitude and penetration depth of edge effects. Our results imply that edge effects on throughfall deposition can be reduced by well-considered management of the LAI of a forest stand. We could formulate a general best-management practice based on our results, e.g., forest stands should be kept dense to decrease the penetration depth or very sparse to limit the level of deposition enhancement. However, a general recommendation formulated based on our results only would be insufficiently founded, because our data set is rather limited (seven forest stands) and only based on measurements in *Pinus nigra* and *P. sylvestris* stands. Therefore, we urge for further research on the effect of LAI on edge patterns of throughfall deposition to correctly formulate best-management practices for mitigating edge effects on deposition.

7 Soil nutrient seepage and soil acidification in forest edges

After: Wuyts, K., De Schrijver, A., Staelens, J., Van Nevel, L., Adriaenssens, S., Verheyen, K. Soil acidification and soil seepage in forest edges under a high deposition load. Submitted to Ecosystems.

7.1 Abstract

In forest edges, the input of NH_4^+, NO_3^-, and SO_4^{2-} via atmospheric deposition is enhanced and, consequently, soil acidification and NO_3^- seepage to the groundwater are expected to be higher, the latter at least in nitrogen (N) saturated forests. Patterns of NO_3^- and SO_4^{2-} seepage and soil acidification were assessed along transects across southwesterly oriented edges of eight coniferous and deciduous forest stands. On average, until 60 m from the edge, NO_3^- seepage was enhanced relative to the forest interior at 128 m from the outer forest edge. However, within the first 20 m of the edges, NO_3^- seepage was about 25 kg N ha^{-1} y^{-1} lower than expected from the N input via throughfall deposition. Since the N stock in the mineral soil was significantly enhanced at the edges in comparison with the forest interior, N retention in the soil was shown to be the main process involved. In the upper 0.05 m of the mineral soil, values of pH (KCl) and exchangeable amounts of base cations were higher at the edges than in the interiors, indicating a lower soil acidification rate at edges. However, in deeper soil layers, this edge pattern faded and even lower pH values were observed at the edges of some of the studied stands. Enhanced input of base cations via throughfall deposition, due to edge effects and to a higher LAI, drift of agricultural lime gifts, and higher decomposition rates of the forest floor are identified as the most important causes. Our results indicate that, at edges, factors such as a different microclimate, enhanced tree growth, and increased deposition of base cations alter the relationship that is found in forest interiors between N and S deposition on the one hand and NO_3^- seepage and soil acidification on the other. The results point to the need for further research on the fate of N and on soil acidification in forest edges.

7.2 Introduction

In the interior of nitrogen (N) saturated forests, high levels of N deposition are associated with increased levels of NO_3^- seepage to the groundwater (Macdonald et al. 2002; Kristensen et al. 2004; Vestgarden et al. 2004; De Schrijver et al. 2008), and increased deposition of both N and sulphur (S) accelerates soil acidification (Bredemeier 1989; De Schrijver et al. 1998; Kreutzer et al. 1998). In forest edges, elevated input of N and potentially acidifying ions (NH_4^+ + NO_3^- + SO_4^{2-}) has been demonstrated abundantly, mostly by throughfall deposition studies [see De Schrijver et al. (2007a) for a review]. Consequently, higher levels of NO_3^- seepage and soil acidification in comparison with the forest interior can be expected. Research results on the consequences of elevated N and S deposition in forest edges are, however, ambiguous. Both lower and higher NO_3^- seepage fluxes and concentrations in soil water are detected at edges than further down the edge (Balsberg-Påhlsson and Bergkvist 1995; Kinniburgh and Trafford 1996; Spangenberg and Kölling 2004; Mellert et al. 2008). In Belgium, De Schrijver et al. (1998) reported lower soil pH-H_20 values in the edge of a *Pinus sylvestris* L. stand as a reflection of a significant edge gradient in potentially acidifying deposition. Also Balsberg-Påhlsson and Bergkvist (1995) detected significantly lower pH-H_20 values and higher percentages of Al in the CEC at the exposed edge of a *Picea abies* L stand in Sweden. However, Szibalski and Felix-Henningsen (1999) related higher pH values at the edge of a *P. sylvestris* stand to deposition of dust from the surroundings and to the ameliorating effect of present deciduous trees with better litter quality than pines.

At edges, additional factors may affect the relationship between N and S deposition on the one hand and NO_3^- seepage and soil acidification on the other. Firstly, the proximity of the edge affects air and soil temperature, wind speed, light availability, and soil moisture (Chen et al. 1995; Jose et al. 1996; Marchand and Houle 2006; Heithecker and Halpern 2007). Secondly, throughfall deposition fluxes of the base cations K^+, Ca^{2+}, and Mg^{2+} are enhanced at edges (Spangenberg and Kölling 2004; Wuyts et al. 2008a, chapter 2), possibly counterbalancing the acidifying effects of SO_4^{2-}, NO_3^-, and NH_4^+. Thirdly, several studies report higher growth rates and leaf areas at edges in comparison with forest interiors (McDonald and Urban 2004; Bowering et al. 2006; Sherich et al. 2007), resulting in higher levels of N sequestration at the edge due to higher tree biomass (Spangenberg and Kölling 2004). The extent of edge effects on atmospheric deposition depends, among other factors, on the forest type: deciduous forest types are subject to smaller and shallower edge effects on N and N+S deposition than coniferous ones are (Spangenberg and Kölling 2004; Wuyts et al.

2008a, 2008b, chapter 2 and 3). Hence, we can expect that these differences in edge gradients among forest types are reflected in NO_3^- seepage and soil acidification.

Our research objectives were (i) to study the general patterns of NO_3^- seepage and soil properties linked to soil acidification along forest edges and (ii) to explore the differences in edge effects between deciduous and coniferous forest types. We hypothesise that, in forest edges, the higher input of N and N+S deposition results in a lower cation exchange capacity (CEC), lower pH (KCl) values, and higher levels of NO_3^- seepage, and that edge effects have a larger impact on NO_3^- seepage and soil acidification in coniferous stands than in deciduous ones.

7.3 Materials and methods

7.3.1 Site description

In each of two regions in Flanders (the northern part of Belgium), approximately 135 km apart, we selected four forest stands on poor sandy soils (Haplic podzols; FAO-ISRIC-ISSS 1998) with low buffering capacity for acidifying deposition (De Schrijver et al. 2006; Van Ranst et al. 2002). Both regions are characterized by high deposition fluxes of NH_3 as a result of intensive livestock breeding. For the physical and chemical characterisation of the climate in the two areas, we refer to chapter 3. The four homogeneous forest stands were each dominated by one tree species: (i) pedunculate oak (*Quercus robur* L.; stands Qr1 and Qr2), (ii) silver birch (*Betula pendula* Roth; stands Bp1 and Bp2), (iii) Austrian pine (*P. nigra* ssp. *nigra* var. *nigra* Arnold; stand Pn1) or Corsican pine (*Pinus nigra* ssp. *laricio* Maire; stand Pn2), and (iv) Scots pine (*Pinus sylvestris* L.; stands Ps 1 and Ps2). All stands had an abrupt forest edge oriented towards the prevailing southwesterly winds. Site code, location, and stand characteristics of the studied forest types are presented in Table 7.1. The stands Qr1, Bp1, Pn1, and Ps1 were situated in the western part of Flanders and Qr2, Bp2, Pn2, and Ps2 in the northern part.

Table 7.1: Studied sites (forest stands) with indication of their location, the dominant tree species, and the mean leaf area index (LAI), pH (KCl), and C:N ratio in the forest interior. The mean age, stem density (SN), mean and dominant tree height (H and Hdom), basal area (BA), and stand volume (V) are given for the dominant tree species of each stand. FH is the fermentation and humus layer of the forest floor; 0-5 cm stands for the upper 0.05 m of the mineral soil.

Site code	Location		Dom. tree species	Age	SN	H	H_{dom}	BA	V	LAI		pH (KCl)	C:N	
				(y)	(ha^{-1})	(m)	(m)	($m^2.ha^{-1}$)	(m^3/ha^{-1})	Summer	Winter		FH	0-5 cm
Qr 1	50°52'08" N	03°27'59"E	*Q. robur*	90	187	24.2	26.4	31	343	1.90	0.79	3.35	18.6	18.0
Qr 2	51°24'44" N	05°02'45"E	*Q. robur*	68	135	21.1	22.4	23	221	1.88	0.40	2.88	18.4	19.1
Bp 1	51°09'22"N	03°04'48"E	*B. pendula*	30-40	3628	11.2	18.2	26	194	1.36	0.61	2.93	19.5	26.2
Bp 2	51°25'56"N	05°00'31"E	*B. pendula*	20-30	2715	8.1	13.8	13	74	1.33	0.31	3.03	18.8	22.9
Pn 1	51°08'26"N	03°06'36"E	*P. nigra* ssp. *laricio*	65	388	19.2	22.0	36	336	2.04	-	2.91	24.0	21.2
Pn 2	51°26'37"N	05°05'14"E	*P. nigra* ssp. *nigra*	43	1162	17.3	19.1	55	488	1.80	-	2.89	25.8	28.1
Ps 1	51°10'11"N	03°09'36"E	*P. sylvestris*	80	458	19.8	23.2	29	287	1.82	-	2.95	24.4	25.7
Ps2	51°24'45"N	05°02'39"E	*P. sylvestris*	76	195	19.4	21.1	20	183	1.03	-	2.86	22.0	22.9

7.3.2 Experimental setup and sample analysis

In each stand, a transect was established perpendicular to the forest edge, along which throughfall deposition and soil solution were monitored year-round and soil samples were taken, at eight distances from the edge (0, 2, 4, 8, 16, 32, 64, and 128 m). Sampling and collection of throughfall water was described in chapter 3.

Soil solution was sampled by means of two suction cup lysimeters per distance plot, i.e., at a depth of 0.30 and of 0.90 m. The lysimeters consisted of (i) a PVC tube fitted with a porous ceramic cup and inserted into the soil at an angle of 60° and (ii) an opaque, glass, one-liter bottle connected to the tube via a polyethylene tube and stored belowground to keep samples cool. The lysimeters were inserted into the soil at an angle of 30° with the soil surface and a pressure of -500 hPa was applied. Following installation, they were flushed during four months prior to the actual monitoring. The level of groundwater was monitored with piezometers; in the stands Qr1 and Bp1, soil nutrient seepage was not measured at 0.90 m due to high ground water levels. Throughfall deposition and soil solution sampling took place fortnightly, from 1 September 2005 to 30 August 2006. On each sampling occasion, sample volume in the lysimeters was measured and a subsample was taken to the laboratory. Aliquots of two consecutive sample collections were pooled volume-weighted into monthly samples. Subsequent to filtration (0.45 µm, Rothe), these samples were analyzed for NH_4^+ (by photometric determination of a reaction product of NH_4^+ at λ = 660 nm according to the Dutch standard method NEN 6576 using a Cary 50 spectrophotometer, Varian) and for Cl^-, NO_3^-, and SO_4^{2-} (using ion chromatography with an ICS-90 Dionex system). Dissolved organic N (DON) concentrations were measured in the soil solution of the Pn2 stand, at a depth of 0.30 m, on three sampling occasions between 28 July and 25 August 2005. DON concentrations were determined as the difference between total dissolved N [determined with persulfate oxidation as described by Vandenbruwane et al. (2007)] and dissolved inorganic N (DIN).

In February 2006, at each distance plot, three samples were taken from the fermentation and humus layer (together, FH) of the forest floor (i.e., the ectorganic layer) and from the mineral soil (0-0.05 m, 0.05-0.10 m, and 0.10-0.30 m depth). Forest floor samples were subsequently dried at 70°C and grinded; samples of the mineral soil were dried at 40°C and sieved. All samples from the mineral soil were analysed for pH (1M KCl). Samples of the upper mineral soil (0-0.05 m) were analysed for total N (modified method of Kjeldahl), total C [loss on ignition, four hours at increasing temperature until 450°C; %C = (100 - % of ashes residue) / 2], and, subsequent to a $BaCl_2$ extraction (5.00 g soil in 90 ml 1 M $BaCl_2$ solution), for the exchangeable amounts of K^+, Ca^{2+}, Mg^{2+}, and Al(III) (flame atomic absorption spectrophotometry, SpectrAA-220, Varian). Additionally, samples of the mineral soil at 0.10-0.30 m depth were analysed for total N. Samples of the FH layer of the forest floor were analysed for total N, total C, and total K^+, Ca^{2+}, and Mg^{2+} following oxidation with a $HNO_3/HClO_4$ (1/5) digestion. Next to this sampling for chemical analyses, the biomass of the FH layer was determined at the outer edge (0-20 m from the edge front) and the forest interior (at 128 m from the edge) by collecting all FH material in a square of 0.39 m by 0.39 m. Soil density of the mineral soil was determined using rings with a given volume.

We estimated the N stored in the aboveground biomass fractions stem wood and fresh leaves/needles. For estimating the N stored in stem wood, we determined stem volumes and N concentrations in stem wood. Stem volumes were calculated from the formulas described by Jansen et al. (1996) and diameters at breast height and tree heights, measured at all trees within 10 m from the axis of the transects in spring 2006. For estimating the N stored in fresh leaves or needles, the leaf area index (LAI) was measured above each throughfall collector in 2006 (as described in chapter 3) and the N concentrations in leaves and needles were assessed. Therefore, in June 2008, we sampled leaves and old and new needles by collecting twigs using telescopic secateurs at a fixed height of 12 m above the floor and from the shaded part of the crown of (i) five trees located in the first ten meters of the forest edge and (ii) five trees in the forest interior. Twigs were not collected in the Qr1 stand because the crowns were too high to reach, nor in the Ps1 stand because of safety reasons regarding the bordering motorway. Trees at the edges were sampled at the sheltered side opposite from the edge to avoid overestimation of the N concentrations by sampling sunlit leaves or needles. From the same trees, also stem cores were taken, at 0.30 m above the root collar. For analysis, bark was excluded and only fully intact stem cores, bored through the entire stem, were selected. Wood and leafy material were dried at 60°C, grinded, and subsequent to total destruction analysed for total N content (modified method of Kjeldahl).

The quality of the chemical analyses was checked by including method blanks and repeated measurements of internal standards and certified reference samples. Repeated measurements of certified reference water samples (CRM 409) performed during the study period yielded for all ions coefficients of variation less than 5 % and recovery rates higher than 90 %.

7.3.3 Element fluxes, N stock, and soil properties

Throughfall deposition fluxes (equiv ha^{-1} y^{-1}) were calculated in chapter 3. Soil nutrient seepage fluxes (equiv m^{-2} y^{-1} or equiv ha^{-2} y^{-1}) at a given depth were determined by multiplying the water seepage flux (l m^{-2} y^{-1}) at this depth with the mean ion concentration in the soil solution at this depth (equiv l^{-1}). The water seepage fluxes were estimated with the Chloride Mass Balance method (Eriksson and Khunakasem 1969; De Schrijver et al. 2008; Boxman et al. 2008), based on the assumption of mass conservation between the input of Cl^- (via throughfall deposition) and the flux of Cl^- through the soil. Recent studies indicate that Cl^- release or retention may occur in the soil (Öberg and Sandén 2005; Bastviken et al. 2007). Therefore, we checked the water seepage fluxes with Na^+ (flame atomic absorption spectrophotometry, SpectrAA-220, Varian) as a tracer at Qr2 and Pn2. The water seepage fluxes calculated with Cl^- were, on average, 1.18 (for oak) and 1.13 (for pine) times higher than those estimated with Na^+ and these differences were marginally significant (paired samples t-test, $p = 0.077$). Nonetheless, the Na^+ and Cl^- concentrations in the soil solution displayed similar patterns throughout time and distance from the edge, so the water fluxes calculated with Cl^- as a tracer were sufficient to study edge effects.

The N stock in leaves or needles was approximated by multiplying the mean N concentration (kg N kg^{-1}) by the LAI (m^2 m^{-2}, converted to m^2 ha^{-1}) and the mean specific leaf or needle weight (dry weight per leaf or needle surface area). Leaf surface area was measured with LI-3000 Portable Area Meter (LICOR) and needle surface area was estimated from needle length, and this yielded next mean specific leaf/needle weight values: *Quercus*: 0.033 kg m^{-2}; *Betula*: 0.045 kg m^{-2}; *P. nigra*: 0.070 kg m^{-2}; *P. sylvestris*: 0.052 kg m^{-2}.

The N stock in tree stem biomass was estimated by multiplying the mean N concentration (kg N kg^{-1}) by the stem volume (m^3 ha^{-1}) and a wood density value. Wood density values were derived from literature (*Quercus*: 700 kg m^{-3}; *Betula*: 660 kg m^{-3}; *Pinus*: 520 kg m^{-3}; Rijsdijk and Laming 1994; Fraanje 1999). At the forest edges, these wood densities were corrected for possibly higher growth rates by multiplication with the ratio of mean weight per length of stem core at the edge to that in the interior, i.e., 0.944 for *Quercus*, 0.990 for *Betula*, 0.894 for *P. nigra*, and 0.981 for *P. sylvestris*. Finally, by dividing the N in the stem wood by the mean tree age (in years), we obtained a rough estimation of the annual N sequestration in stem wood (kg N ha^{-1} y^{-1}).

For the mineral soil, the ratios C:N and Al(III):Ca were determined. The N stocks (kg N ha^{-1}) in the upper 0.05 m and between 0.10 and 0.30 m depth were calculated based on the measured N concentration and soil density.

7.3.4 Statistics

By means of a repeated-measures ANOVA (RMA), the effects of within-subject factor 'distance to the forest edge' (8 levels) and between-subjects factor 'forest type' (4 levels: *Quercus*, *Betula*, *Pinus nigra*, and *P. sylvestris*) on the variables of soil nutrient seepage and soil characteristics were tested. Statistics on soil properties were performed without the data from the Qr1 stand because historic land use maps indicate a short period of grassland practice around 1900, which is confirmed by higher forest interior pH values (+ 0.3-0.5 pH units; Table 7.1) in the upper mineral soil in comparison with the other stands. We found, however, no considerable effects of this former land use on the soil seepage of N. We calculated the mean throughfall deposition and soil seepage fluxes of NO_3^- + NH_4^+ (DIN) and SO_4^{2-} in the first 32 m of the forest edge weighted by the share of each distance plot in an edge zone of 32 m (i.e., 1, 2, 3, 6, 12, and 8 m for the first to sixth distance plots, resp.). Repeated-measures ANOVA was applied to identify and separate the influence of 'edge proximity' (2 levels: forest edge 0-32 m and forest interior at 128 m) and of deciduous vs coniferous forest types on the throughfall deposition and soil seepage of DIN and SO_4^{2-}. This simplification of data enabled us to make more general conclusions.

For each of the stands, the difference in the N concentrations in stem wood and in leaves/needles between the edge and the forest interior was tested with an independent samples t-test. Moreover, a repeated-measures analysis was applied on the N stock in stem biomass, leaf/needle biomass, and in the mineral soil and on the N sequestration in stem wood and leafy material, to separate the influence of 'edge proximity' (2 levels: edge and interior) and of deciduous vs coniferous forest types.

Furthermore, differences between edge and interior were tested for the deciduous and coniferous stands individually with non-parametric related samples tests (Wilcoxon signed ranks test). All statistical analyses were performed using SPSS 15.0.

7.4 Results

7.4.1 Throughfall deposition and soil nutrient seepage flux

We found a significant effect of forest type on the NO_3^- + NH_4^+ (dissolved inorganic nitrogen; DIN) and SO_4^{2-} soil seepage fluxes at 0.30 m depth (p = 0.001 and <0.001, respectively; Fig. 7.1). The DIN and SO_4^{2-} soil seepage flux at 0.90 m was not significantly influenced by the factor 'forest type'. The distance from the forest edge did not significantly influence DIN and SO_4^{2-} soil seepage fluxes at 0.30 and 0.90 m depth. We found no significant interaction between the factors 'forest type' and 'distance from the forest edge'.

The throughfall deposition and soil seepage fluxes of DIN and SO_4^{2-} averaged over the first 32 m of the edge were consistently higher than the fluxes in the interior, and differences were marginally significant for the soil seepage fluxes at 0.30 m of DIN (RMA: p = 0.084) and SO_4^{2-} (p = 0.066) and significant for the DIN soil seepage fluxes at 0.90 m (p = 0.005). When considering coniferous and deciduous stands separately, the differences between the first 32 m of the edge and the interior were only marginally significant for the DIN soil seepage flux at 0.90 m in the coniferous stands (p = 0.068; n = 4). In the first 32 m of the edges, the soil seepage fluxes of DIN at 0.30 m were 105 % (in the deciduous stands) to 71 % (in the coniferous stands) higher than in the forest interior. In absolute fluxes, the edge effect gave rise to a larger increase in DIN soil seepage (at 0.30 m) in the deciduous stands (8.6 kg N ha^{-1} y^{-1}) than the coniferous stands (24.9 kg N ha^{-1} y^{-1}; Fig. 7.2). At 0.90 m soil depth, the increase of DIN soil seepage in the first 32 m of the edge relative to the interior is 9.6 kg N ha^{-1} y^{-1} (or 131 %) and 15.1 kg N ha^{-1} y^{-1} (or 76%) in the deciduous and coniferous stands, respectively (Fig. 7.2). For SO_4^{2-}, the edge effect on the seepage flux at 0.30 m was smaller in the deciduous stands than in the coniferous ones, in relative numbers (12 and 55 % higher edge seepage for deciduous and coniferous stands, respectively) as well as in absolute numbers (1.8 and 9.0 kg SO_4^{2-}-S ha^{-1} y^{-1}, respectively). At 0.90 m depth, the increase in SO_4^{2-} soil seepage in the first 32 m of the edge was similar in absolute fluxes (3.3 and 3.6 kg S ha^{-1} y^{-1} for deciduous and coniferous, resp.), but relative to the interior, the increase was larger in the deciduous stands (45 %) than in the coniferous ones (16 %).

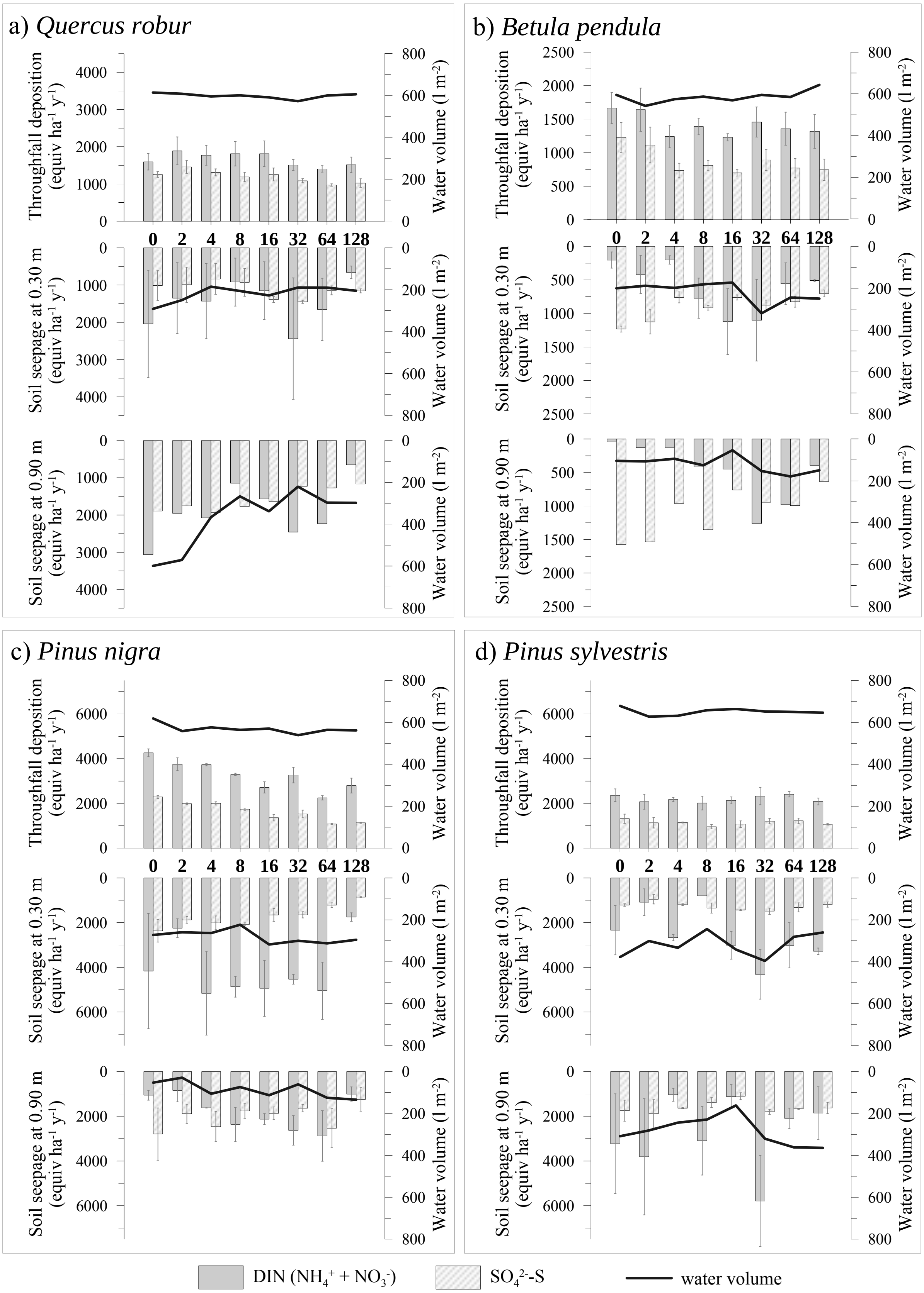

Fig. 7.1: Annual throughfall volume and water percolation volume fluxes (mm ha^{-1} y^{-1}) and throughfall deposition and soil seepage fluxes at 0.30 and 0.90 m depth of DIN (NO_3^- + NH_4^+) and SO_4^{2-}-S (equiv ha^{-1} y^{-1}) along transects across the forest edges, averaged per forest type: (a) *Q. robur*, (b) *B. pendula*, (c) *P. nigra*, and (d) *P. sylvestris*. Error bars indicate standard errors.

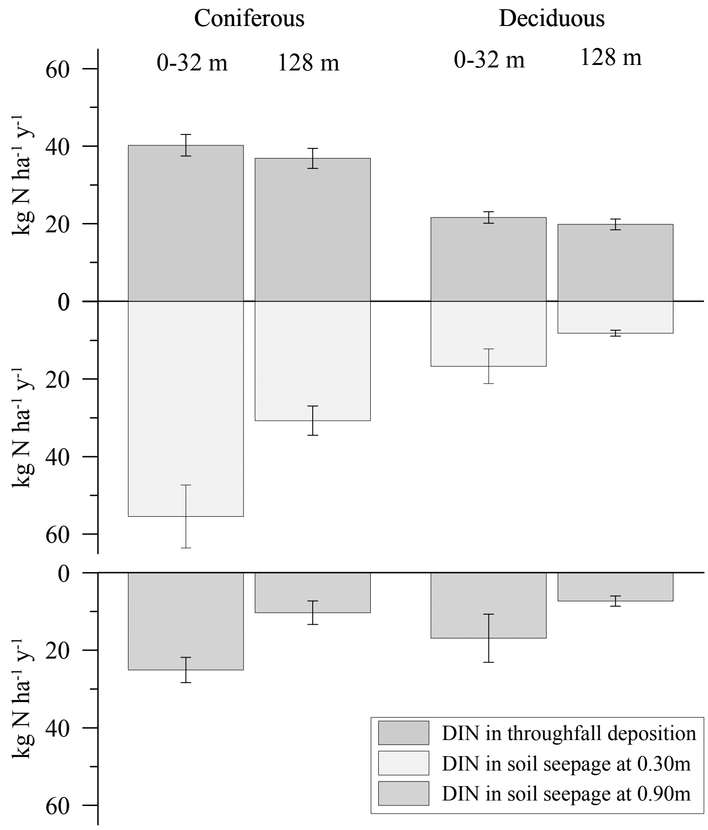

Fig. 7.2: Annual DIN ($NH_4^+ + NO_3^-$) throughfall deposition and soil seepage fluxes at 0.30 and 0.90 m depth (kg N ha^{-1} y^{-1}). All data are weighed mean values for the first 32 m of the edge and at 128 m from the edge, for the coniferous and deciduous forest types individually. Error bars indicate standard errors.

Significant linear relations were found between the ln-transformed DIN soil seepage at 0.30 m and at 0.90 m ($R^2 = 0.508$; $p < 0.001$; slope = 0.951; n = 48) and between the soil seepage of SO_4^{2-} at 0.30 m and at 0.90 m ($R^2 = 0.407$; $p < 0.001$; slope = 0.843; n = 48).

During the three sampling occasions in Pn2, the DON concentrations in the soil solution (at 0.30 m depth) were enhanced in the first 5 m of the edge relative to 32-64 m behind the edge, on average by a factor of 1.5.

7.4.2 Forest floor and mineral soil

The factor ‘distance from the forest edge’ significantly affected the C, N, K, Ca, and Mg concentrations in the FH layer of the forest floor, as well as its C:N ratio (Fig. 7.3; Table 7.2). The N and C concentrations and the C:N ratio were lower while the K, Ca, and Mg concentrations were higher at the edges. A significant effect of the factor ‘forest type’ was observed on the N and Ca concentrations and the C:N ratio of the FH layer (Table 7.2). At the outer edges (0-20 m), on average 4.55 ± 1.62 kg m^{-2} of dry FH layer was collected, which was significantly different from the dry weight of the FH layer in the forest interiors which amounted to 6.35 ± 1.90 kg m^{-2} (non-parametric independent samples t-test: $p = 0.016$; $n = 25$; Table 7.3).

Table 7.2: Significance (p value) of edge effects (factor 'distance to the forest edge' or 'distance') and forest type effects on the pH (KCl), the C, N, K, Ca, Mg, and Al concentrations, and the C:N and Al:Ca ratios of the FH layer (fermentation and humus layer of the forest floor) and the mineral soil, according to the repeated-measures ANOVA outcome (bold: $p < 0.05$).

	FH layer			Mineral soil			
	Distance	Forest type	Distance * forest type	Soil depth (m)	Distance	Forest type	Distance * forest type
pH(KCl)				0-0.05	**0.001**	0.282	**0.048**
				0.05-0.10	0.103	0.060	**0.012**
				0.10-0.30	0.398	**0.004**	**0.022**
C	**0.001**	0.336	0.070	0-0.05	**0.019**	0.135	0.329
N	**0.002**	**0.008**	0.086	0-0.05	**0.001**	**0.014**	0.517
C:N	**0.016**	**<0.001**	0.287	0-0.05	0.118	**0.015**	0.782
K	**0.028**	0.162	**0.016**	0-0.05	**0.014**	**0.001**	0.331
Ca	**0.037**	**0.038**	0.414	0-0.05	**<0.001**	**<0.001**	0.146
Mg	**<0.001**	0.658	**0.018**	0-0.05	**<0.001**	**0.002**	0.461
Al				0-0.05	0.354	**<0.001**	**0.037**
Al:Ca				0-0.05	0.062	**0.005**	0.417

In the upper mineral soil (0-0.05 m), the N and C concentrations and the concentrations of exchangeable K, Ca, and Mg were significantly influenced by 'distance from the forest edge', and were higher at the edge (Table 7.2; Fig. 7.3). The factor 'distance from the forest edge' marginally affected the Al:Ca ratio, which was lower at the edges. The C:N and Al:Ca ratios, the N concentrations, and the concentrations of exchangeable K, Ca, Mg, and Al were significantly influenced by the factor 'forest type' (Table 7.2). Furthermore, for Al, a significant interaction was found between the factors 'distance from the forest edge' and 'forest type'.

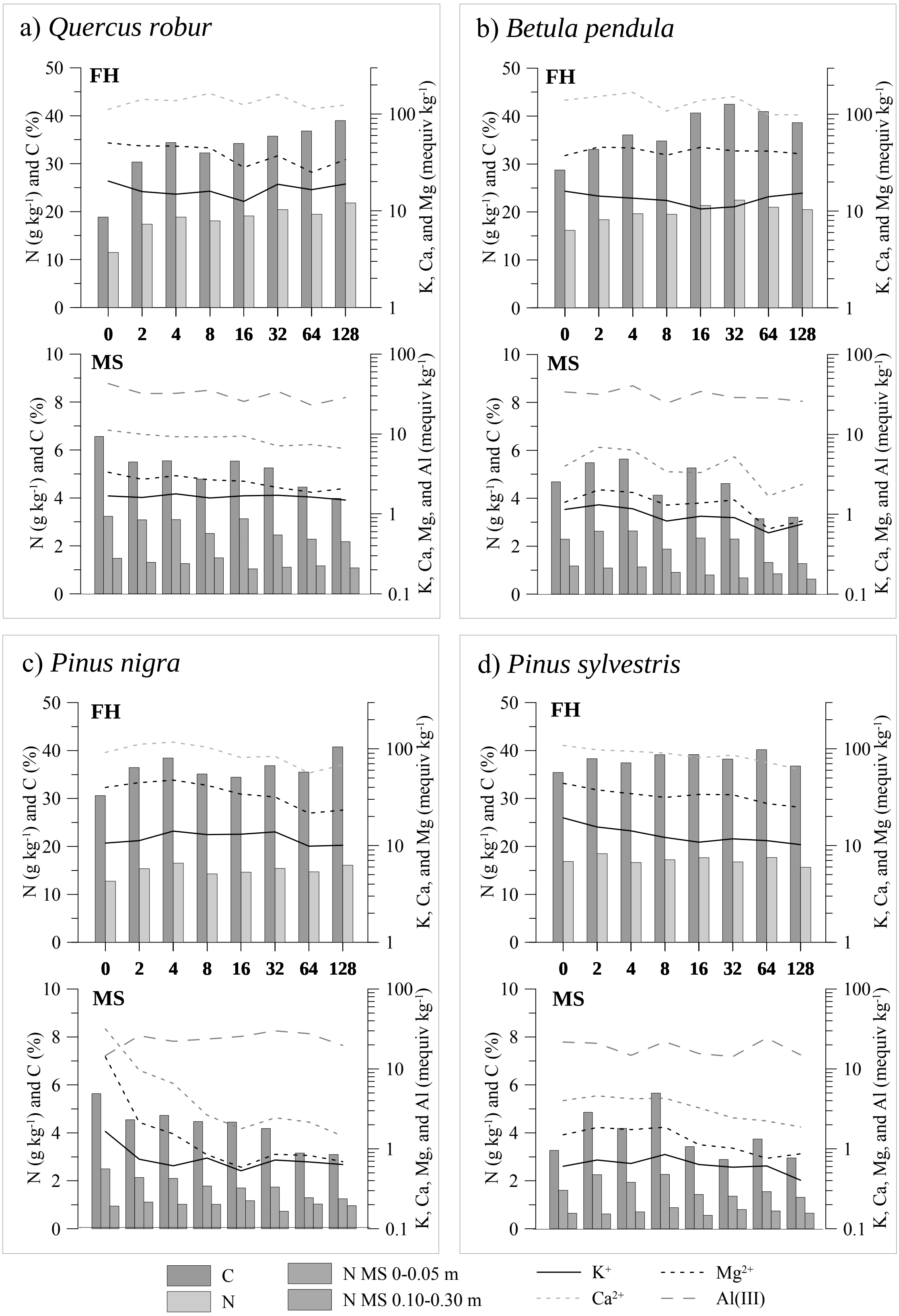

Fig. 7.3: N (g kg^{-1}), C (%), K, Ca, and Mg (mequiv kg^{-1}) concentrations in the FH layer of the forest floor (FH) and N (g kg^{-1}) and C (%) concentrations and exchangeable concentrations of K, Ca, Mg, and Al(III) (mequiv kg^{-1}) in the upper 0.05 m of the mineral soil (MS) along transects across the forest edges, averaged per forest type: (a) *Q. robur*, (b) *B. pendula*, (c) *P. nigra*, and (d) *P. sylvestris*. Also given is the N concentration (g kg-1) in the MS at 0.10-0.30 m depth.

Table 7.3: Dry weight (kg m^{-2}) of the FH layer (fermentation and humus layer of the forest floor) in the first 0-10 m of several edges and forest interiors (at 128 m from the edge) under study. The edge to interior (E to I) weight ratio is also given.

Site code	Edge	Interior	E to I ratio
Qr2	9.03	10.33	0.87
Bp1	5.94	9.32	0.64
Pn1	11.41	15.11	0.76
Pn2	5.76	7.16	0.80
Ps1	7.37	12.45	0.59

The pH(KCl) of the upper 0.05 m of the mineral soil was subject to a significant edge effect (Table 7.2), with higher values occurring at the edges (Fig. 7.4). In deeper layers of the mineral soil, this edge effect fades out as indicated by Table 7.2. From Fig. 7.4, it appears that, at a soil depth of 0.10-0.30 m, the 'positive' edge effect on the pH in the upper mineral soil was inversed in a 'negative' one, with lower pH values at the edge. Although this 'negative' edge effect was not significant (Table 7.2), it was nevertheless appreciable in the *B. pendula* and *P. nigra* stands (Fig. 7.4). Significant interaction between edge effects and forest type occurred for all soil septh ranges considered in this study.

At the edge, soil density was significantly lower than in the forest interior, i.e., 987 ± 118 kg m^{-3} vs 1118 ± 108 kg m^{-3}, on average. In the upper 0.05 m of the mineral soil, significantly higher N stocks were found in the first 10 m of the edge compared to the forest interior (RMA: $p = 0.003$); when the deciduous and coniferous stands were considered separately, the pattern was consistent but less significant (deciduous and coniferous: $p = 0.068$, $n = 4$; Fig. 7.5). The N stock of the upper mineral soil was, on average, 376 kg ha^{-1} (or 50 %) higher at the edge than in the forest interior. Differences between edge and interior were larger in the deciduous forest types (427 kg ha^{-1}) than in the coniferous ones (325 kg ha^{-1}). The N stock in the mineral soil at 0.10 to 0.30 m soil depth was 258 kg ha^{-1} (or 14 %) higher in the first 10 from the edge than in the interior, but the differences were not significant (RMA: $p = 0.232$) and depended upon the forest type (Fig. 7.5). When considering the deciduous stands separately, the difference in N stocks between edge and interior was marginally significant ($p = 0.068$, $n = 4$) and amounted to, on average, 594 kg ha^{-1}. In the coniferous stands, no significant differences between edge and interior in N stock at 0.10-0.30 m depth were observed ($p = 0.465$; $n = 4$). A comparison of the N stocks in the mineral soil between coniferous and deciduous stands over the whole transect length revealed no significant differences, nor at 0-0.05 m depth (RMA: $p = 0.542$), nor at 0.10-0.30 m depth ($p = 0.967$).

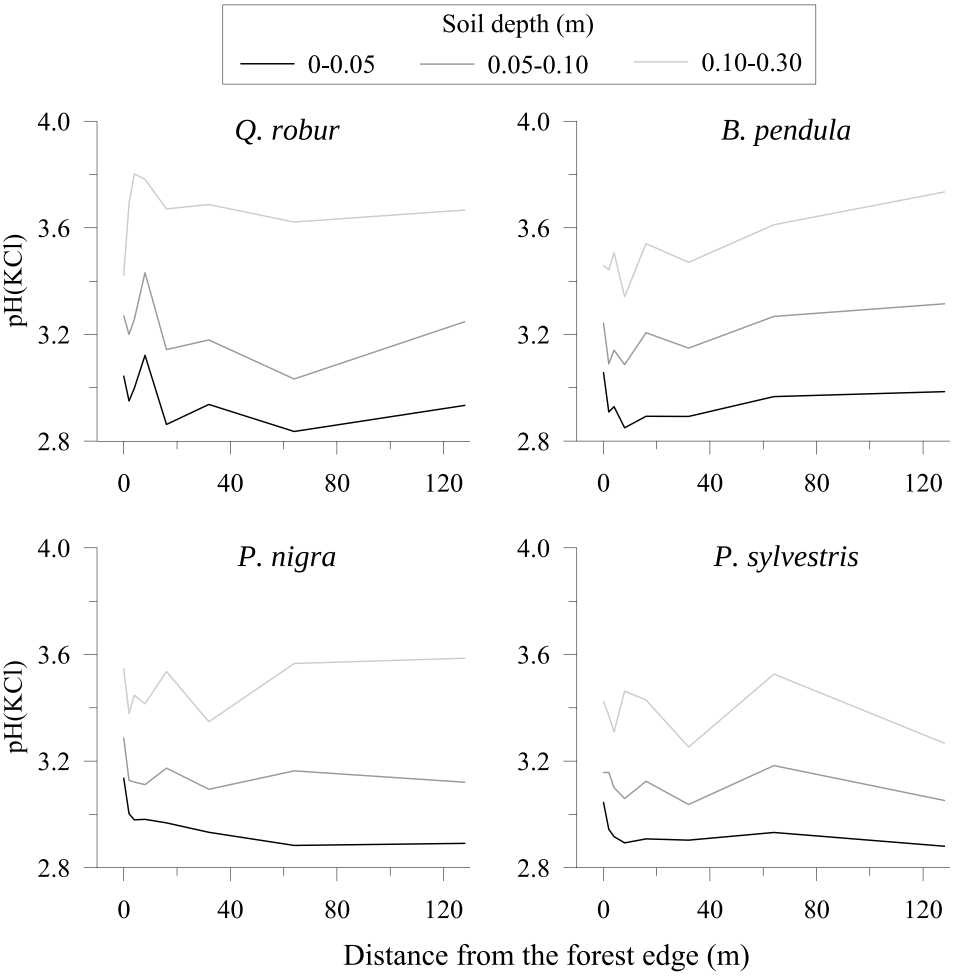

Fig. 7.4: Mean pH (KCl) of the upper 0-0.05, 0.05-0.10, and 0.10-0.30 m of the mineral soil along transects across the forest edges per forest type.

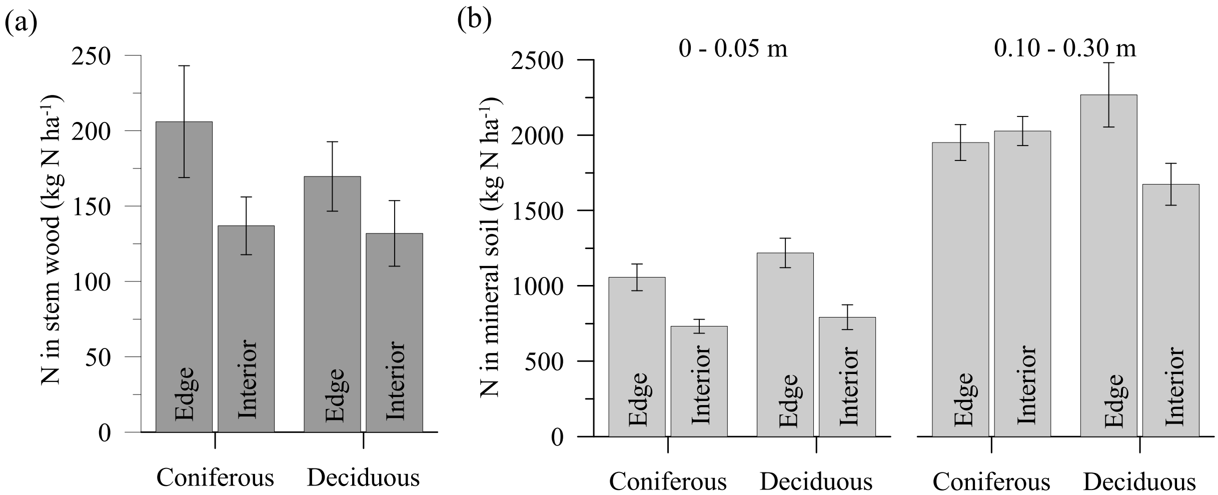

Fig. 7.5: N stock (kg N ha^{-1}) in (a) stem wood and (b) the mineral soil at 0 to 0.05 m and at 0.10 to 0.30 m soil depth. All data are mean values for the first 10 m of the edge and at 128 m from the edge, for the coniferous and deciduous forest types individually. Error bars indicate standard errors.

7.4.3 N in stems and fresh leafy material

In general, LAI values and stem volume were consistently higher in the first 10 m of the edges (Fig. 7.6). The N concentrations in stem wood and leafy material at the edge were not significantly different from the concentrations in the forest interior, except for the higher concentration in stem wood at the edge of the Bp2 stand ($p = 0.006$) and the lower concentration in stem wood at the edge of the Pn2 stand ($p = 0.006$; Fig. 7.6).

According to Table 7.4, the N stock in the leaves/needles was, on average, 3.3 kg N ha^{-1} (or 21 %) higher in the first 10 m of the edge (on average 19.5 kg N ha^{-1}) than in the interior (on average 16.2 kg N ha^{-1}). In the deciduous stands, the N stock in the edge was, on average, 5.1 kg N ha^{-1} higher (with differences up to 9.8 kg N ha^{-1} in Bp2), but in the coniferous stands, the mean difference in N stored in the fresh needles was only 1.5 kg N ha^{-1}). However, the differences between edge and interior were not significant (RMA: $p = 0.194$; deciduous: $p = 0.285$, $n = 3$; coniferous: $p = 1.00$; $n = 3$; Table 7.4).

In all stands, except in the Ps2 stand, the N stock in stem biomass was higher in the first ten meter of the edge than in the forest interior (Table 7.4). On average, an additional amount of 53 kg N ha^{-1}, or roughly estimated 0.96 kg N ha^{-1} y^{-1}, was stored in the stems in the first 10 m of the edge in comparison with the interior (Fig. 7.5a), and these differences in N stock and the annual N uptake were marginally significant (RMA: $p = 0.097$ and 0.065, respectively). In the coniferous stands, differences in the annual N uptake in stem wood between edge and interior were not significant ($p = 0.144$; $n = 4$), while in the deciduous stands, differences were only marginally significant ($p = 0.066$; $n = 4$; Fig. 7.5a). Both the mean N stock and annual N uptake in stem wood were not significantly different between the deciduous and coniferous stands (RMA: $p > 0.5$, $n = 4$).

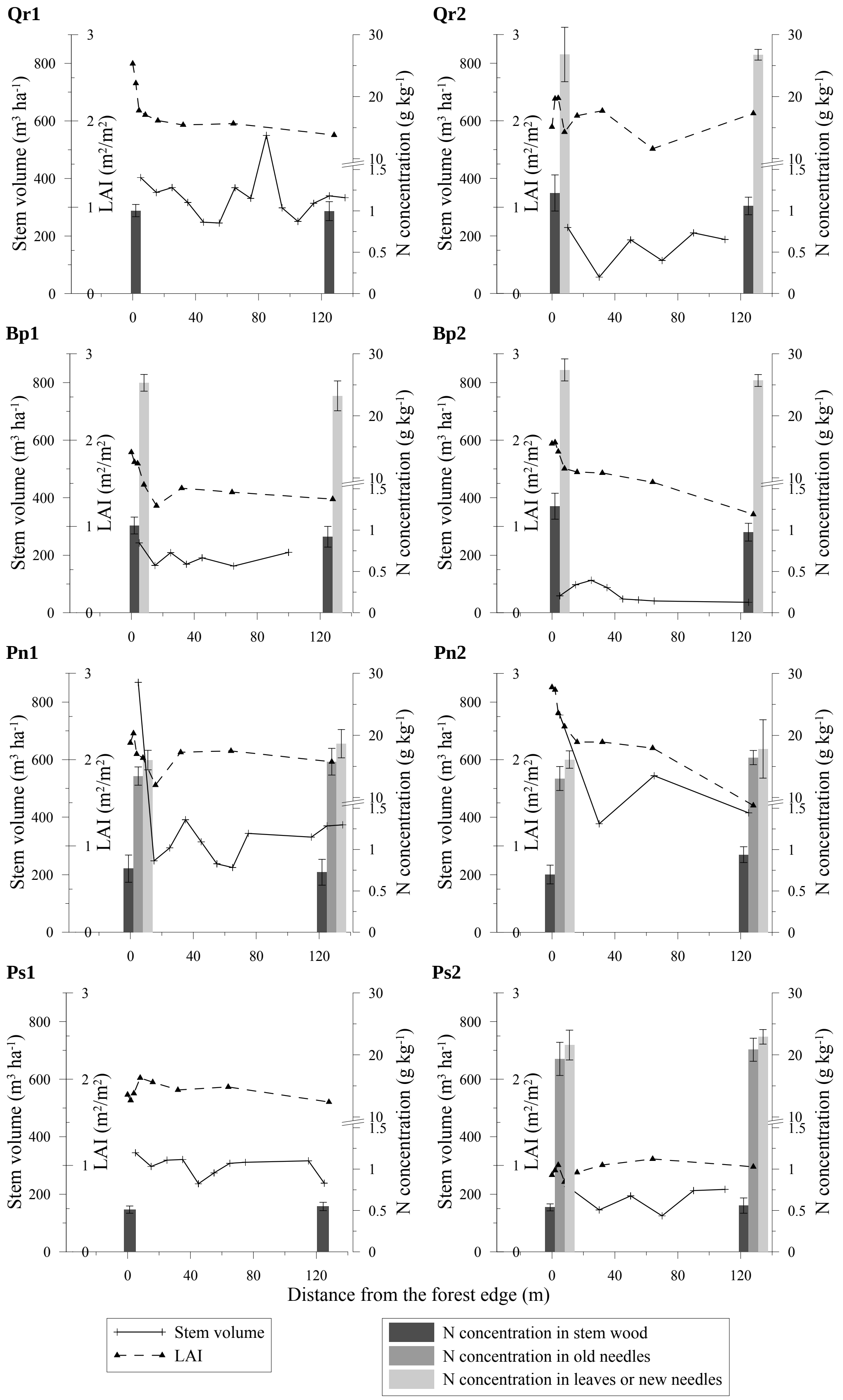

Fig. 7.6: Stem volume (m3 ha^{-1}), LAI, and N concentrations (g kg^{-1}) in tree stem wood, leaves and old and new needles along transects across the southwesterly oriented edges of the studied forest stands. Codes above each one of the plots refer to the site codes in Table 7.1. Error bars indicate standard deviation.

Table 7.4: The N stock in tree stem wood and in fresh leaves or needles, (both in kg N ha^{-1}) at the forest edge (0-10 m from the edge, E) and the forest interior (128 m from the edge, I) of all eight forest sites under study (see Table 7.1 for explanation of site codes; -: not measured).

Site code	Tree stem wood			Fresh leafy material		
	Edge	Interior	E-I	Edge	Interior	E-I
Qr1	287	232	35	-	-	-
Qr2	184	139	45	18.4	18.5	-0.1
Bp1	173	127	46	19.3	13.7	5.7
Bp2	48	23	24	22.9	13.1	9.8
Pn1	393	162	231	23.4	25.0	-1.7
Pn2	255	234	21	23.4	15.6	7.8
Ps1	106	78	27	-	-	-
Ps2	70	73	-3	9.5	10.9	-1.4

7.5 Discussion

In this chapter, we only discuss edge effects on soil nutrient seepage and soil acidification. Our results on the effect of forest type on soil nutrient seepage and acidification are in line with previous studies performed in the interior of similar forest types (Augusto et al. 2002; Herrmann et al. 2005; Reich et al. 2005; De Schrijver et al. 2007b, 2008) and we refer to these studies for discussion on this issue.

7.5.1 Edge effects on soil nutrient seepage

In comparison with the forest interior, enhanced soil seepage fluxes of dissolved inorganic nitrogen (DIN) and SO_4^{2-} are found in a 32 m edge zone, at the edges of both coniferous and deciduous forests. Other than for N, the pattern of SO_4^{2-} via throughfall deposition was univocally reflected in the SO_4^{2-} soil seepage as peak values arose at the edge fronts, behind which seepage decreased with increasing edge distance. Detailed patterns of DIN soil seepage, however, differed remarkably from those of DIN throughfall deposition. In the *Q. robur*, *B. pendula*, and *P. nigra* stands, maximum values of DIN and throughfall deposition fluxes occurred at the front of the edges (0-2 m from the edge) (see chapter 3). In the *P. sylvestris* stands, edge effects on throughfall deposition of DIN and SO_4^{2-} were small, because the edge of the Ps1 stand borders a very busy motorway - trucks passing by at < 5 m from the edge considerably disturb the expected pattern of wind speed and turbulence across forest edges - and because of the very low stand density in the case of the Ps2 stand. The peak values of DIN soil seepage did, however, not occur at the edge front but several meters behind it, even up to 32 or 64 m from the edge. In the first 0 to 20 m of the edge, remarkably low DIN soil seepage occurred. This pattern of DIN soil seepage was observed in seven out of eight studied forest stands. At the outer edge of 20 m, on average, 25 kg N ha^{-1} y^{-1} of the DIN input via throughfall deposition is not 'recovered' in the DIN soil seepage flux at 0.90 m depth (Fig. 7.7). In the first meters of an exposed beech edge, Kinniburgh and Trafford (1996) reported very low NO_3^--N concentrations in the pore water beneath the rooting zone; however, concentrations were increased at 30-100 m from the edge, after which they decreased again. These results and those of Spangenberg and Kölling (2004) agree with our findings on soil seepage of DIN. Several processes can cause the DIN soil seepage to be lowered at the outer edge (i.e., in the first 0-20 m of the edge, but particularly in the first 10 m of the edge): (1) enhanced N uptake by trees, (2) enhanced N immobilisation in the soil, and (3) enhanced emission of NO and N_2O. These processes are discussed subsequently.

N uptake by trees

Spangenberg and Kölling (2004) hypothesised that NO_3^- seepage was lower at outer edges than several meters behind it, because more N is sequestered in trees and understory biomass. Higher levels of growth rate, stand basal area, crown depth, and leaf area are reported at forest edges in comparison with forest interiors (McDonald and Urban 2004; Bowering et al. 2006; Sherich et al. 2007). Also in our study, the stand volume and the LAI were generally higher in the first 10 m of the edge. And although the N concentrations in the stem wood and the leaves/needles were not significantly different between edge and interior, the N stock in biomass was higher in the first 10 m of the edge. Annual N sequestration (roughly estimated from the N stock) was, on average, 0.96 + 1.47 kg ha^{-1} higher in the first 10 m of the edge than in the forest interior. Hence, root N uptake is expected to be higher in the first 10 m from the edge than further behind it. We did not measure N sequestration in the branches and belowground biomass, but we expect its contribution in the total tree N pool to be comparable to that of stems and leafy material (Neirynck et al. 1998; Alriksson and Eriksson 1998) and its response to edge proximity to be similar. Even if higher N sequestration in branches and roots of a few kg N ha^{-1} y^{-1} would be taken into account, an increase in N sequestration in tree biomass could not solely explain the discrepancy between DIN throughfall deposition and soil seepage of up to 25 kg N ha^{-1} y^{-1} at the outer edge.

Belowground N immobilisation

A second possible explanation for the lower DIN soil seepage at the outer edges might be found in a significantly increased N immobilisation in the mineral soil at the edge. We did not measure annual sequestration in the soil, but the N stock in the upper 0.05 m of the mineral soil was, on average, 556 kg N ha^{-1} higher in the first 10 m of the edge than in the interior. At 0.10-0.30 m soil depth, no significant differences in N stock were observed in the coniferous stands between edge and interior, but in the deciduous stands, an additional amount of, on average, 594 kg N ha^{-1} was stored at the edges. Processes that are believed to be involved in belowground N retention are abiotic and biotic (microbial and mycorrhizal) fixation of NO_3^- and NH_4^+ in or on soil organic matter (Magill et al. 1997; Aber et al. 1998; Sjöberg and Persson 1998; Davidson et al. 2003; Micks et al. 2004; Morier et al. 2008). Dissimilatory reduction of NO_3^- to NH_4^+ (DNRA) is also involved in N immobilisation, but seems to be mainly of importance in unpolluted, N-limited forests (Bengtsson and Bergwall 2000; Huygens et al. 2007).

According to Persson et al. (2000), soil N only increases through circulation of plant material: the extra inorganic N is taken up by plants and is returned to the soil as organic N with N-enriched litter. We presume that (i) an increased rate of decomposition occurs at the edges within the first ± 64 m from the edges and (ii) an increased rate of biomass input through litterfall takes place within the outer edge of the stands (0-20 m), causing N immobilisation in the mineral soil to be higher in the outer edges than at ± 20 to 128 m from the edge.

Firstly, input of base cations to the forest floor is enhanced at edges due to edge effects and higher LAI at edges (chapter 2; Beier and Gundersen 1989). Also microclimatic gradients of soil temperature and moisture occur along the forest edge (Chen et al. 1995; Jose et al. 1996; Marchand and Houle 2006). At the edges, higher soil temperature and higher Ca concentration in the litter may increase decomposition and mineralisation rates (Reich et al. 2005; Hobbie et al. 2007). The higher decomposition rates were confirmed by additional measurements of the dry weight of the FH layer of the forest floor: on average 4.55 ± 1.62 kg m^{-2} of FH layer was collected at the edges, which was significantly lower than the 6.35 ± 1.90 kg m^{-2} of FH layer in the interiors. Consequently, at the edge, less N and organic matter is retained in the forest floor (i.e., the ectorganic layer) than in the forest interior.

Secondly, we expect that the flow of organic material via the forest floor to the mineral soil is higher at the outer edges (0-20 m) than at 20-128 m. Although the N stocks in fresh leaves/needles were not significantly different between edge and interior, the biomass of fresh leaves, and thus also the flux of organic material to the forest floor, was higher in the outer edge than further down the edge and in the interior. The higher organic matter content of the mineral soil at the edges indicates a higher potential for fixation of NO_3^- and NH_4^+ in or on soil organic matter. Also, in the outer edge of a *P. nigra* stand on similar soils and during the same measuring period, Vandenbruwane (2008) found a higher flow rate of DON and DOC from the forest floor to the mineral soil and detected that more DON and DOC were retained in the upper 0.70 m of the mineral soil in the edge (12.7 kg N ha^{-1} y^{-1}) than in the interior (6.6 kg N ha^{-1} y^{-1}). Vandenbruwane (2008) furthermore refers to N enrichment of the DOM at the edge, but also an increase in DOC adsorbed to the mineral soil may occur.

Next to retention in the soil (Kaiser and Zech 2000), DON can be leached from the soil profile (Qualls et al. 2002). Despite the higher retention in the mineral soil, Vandenbruwane (2008) also found DON soil seepage to be higher in the outer edge (5.0 kg N ha^{-1} y^{-1}) than in the forest interior (3.3 kg N ha^{-1} y^{-1}). In our study, during a snapshot sampling in the Pn2 stand, these findings were confirmed: DON concentrations in soil seepage were about 1.5 times higher in the outer edge than at 32-64 m behind it and we found no significantly less water seepage fluxes at the edge. This indicates that, although the DIN seepage to groundwater was lower, DON seepage may be elevated in the outer meters (0 to 20 m) of forest edges.

NO and N_2O emissions

A third process possibly involved at the outer edges is an increased N loss through gaseous NH_3, NO, N_2O, and N_2 emissions. We can assume that NH_3 volatilization was negligible due to the low pH of the soils (Bowden 1986; Magill et al. 1997). Production of NO and N_2O, primarily driven by microbial processes including nitrification and denitrification (Wrage et al. 2001; Ambus et al. 2006), depends on soil temperature, moisture, pH, and N input (Carnol and Ineson 1999; Butterbach-Bahl et al. 2002; Schindlbacher et al. 2004; Ambus et al. 2006; Kesik et al. 2006; Pilegaard et al. 2006). From the gradients in N input via throughfall deposition and microclimate (Chen et al. 1995), emission fluxes of NO and N_2O are expected to be elevated at edges.

Outline of throughfall vs seepage patterns in forest edges

To get a global insight in the patterns of DIN throughfall deposition vs soil seepage at 0.90 m soil depth, Fig. 7.7 provides a generalizing overview based on averaged throughfall deposition and soil seepage fluxes from all stands but Qr1, Bp1, and Ps2. In the stands Qr1 and Bp1, soil nutrient seepage was not measured at 0.90 m due to high ground water levels; in the stand Ps2, thinning about three years prior to the start of our measuring campaign caused N soil seepage to be temporarily enhanced (von Wilpert et al. 2000; Weis et al. 2006). The figure is used to illustrate the divergence between the patterns of DIN throughfall deposition and seepage at the edge and the occurrence of three zones from edge to interior: from 0 to 20 m, from 20 to 60 m, and from 60 m to the interior.

In the first 20 m of the edge, i.e. the outer edge, a void between throughfall deposition and soil seepage occurs, and mainly increased N retention in the mineral soil accounts for it. Within this outer edge, organic matter input from higher amounts of leaf litter fall occurs. From 20 to 60 m from the edge, approximately all N reaching the forest floor via throughfall deposition and litterfall leaves the ecosystem via soil nutrient seepage, which we ascribed to low net N retention in the forest floor (due to enhanced decomposition rates) and the mineral soil (due to lower organic matter input in comparison with the outer edge). From literature, it can be deduced that microclimatic gradients of soil temperature and moisture can occur across the forest edge up to about 60 m from the edge (see Fig. 1.6, chapter 1), causing increased decomposition and mineralisation rates (Hobbie et al. 2007). This area of the forest edge comes close to a N (over)saturated forest ecosystem: the N coming into the system almost entirely flushes through and leaves the system.

At more than 60 m from the edge, some significant N retention reoccurs, predominantly in the forest floor and possibly also in the upper mineral soil, indicating that N retention mechanisms are not completely exhausted under this high N deposition load. However, because not all possible N pathways were actually measured in this study, we stress the importance of further research to verify our assumptions.

Dise et al. (1998), Gundersen et al. (1998), and Macdonald et al. (2002) suggested the use of the C:N ratio of the forest floor or upper soil layers to predict NO_3^- seepage. In our study, the C:N ratios of the forest floor in the outer edges (i.e., the first 20 m of the edge) were lower than at 32-128 m from the edge, which would indicate that the outer edge is more prone to NO_3^- seepage. Therefore, we can conclude that, for forest edges, the C:N ratio alone is not a sufficient predictor for N saturation and NO_3^- seepage in forest edges. Also Brumme and Khanna (2008) stress that additional factors other than C:N ratio such as N sinks in plants and soil should be considered in studies on N seepage.

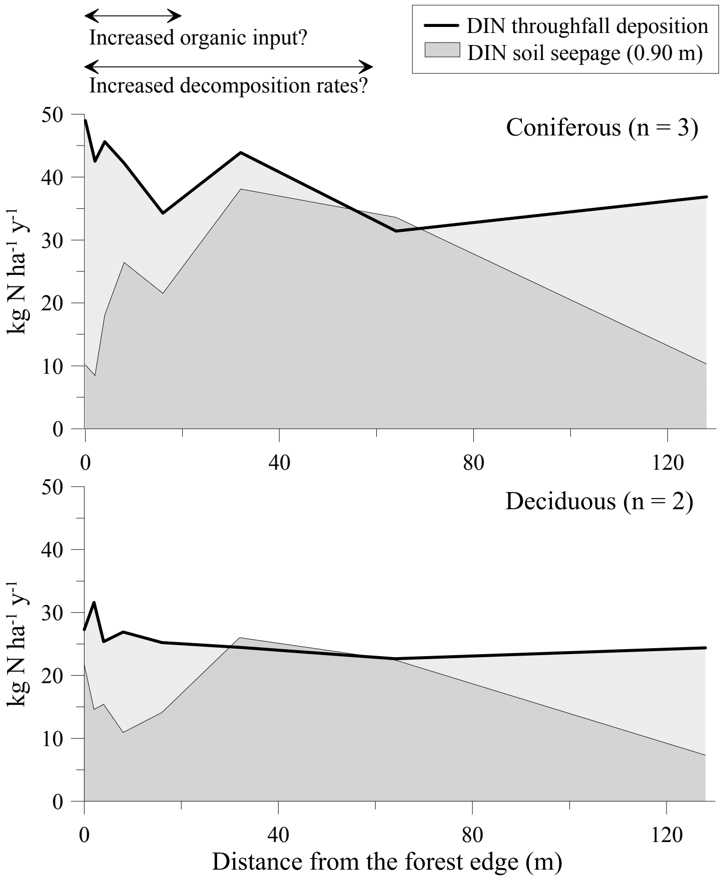

Fig. 7.7: Patterns of dissolved inorganic N (NO_3^- + NH_4^+) in throughfall deposition and soil nutrient seepage at 0.90 m soil depth along transects across the forest edges. The data are mean values from the deciduous stands Qr2, Bp2 and the coniferous stands Pn1, Pn2, and Ps1. Indicated are the edge zones influenced by the factors that possibly relate to the seemingly increased N retention in the soil at the edges.

In absolute fluxes, the increase in DIN soil nutrient seepage at 0.30 m at the edge vs the interior was higher in the coniferous stands than in the deciduous ones (24.9 and 8.6 kg N ha^{-1} y^{-1}, respectively). Also at a depth of 0.90 m, differences between edge and interior were greater in the coniferous stands (15.1 kg N ha^{-1} y^{-1}) than in the deciduous ones (9.6 kg N ha^{-1} y^{-1}). However, the edge effects on DIN soil seepage were not found to be significantly different between coniferous and deciduous forest types, although, when considered separately, the increase in DIN soil seepage at 0.90 m depth at the edges was marginally significant in the coniferous stands but not in the deciduous stands. The latter may be the results of the fact that soil seepage at 0.90 m was measured only in two deciduous forest stands. Also for SO_4^{2-} soil seepage at 0.30 m depth, the increase at the edges relative to the interiors was greater in the coniferous stands (on average, 9.0 kg S ha^{-1} y^{-1}) than in the deciduous ones (on average, 1.8 kg S ha^{-1} y^{-1}), although this effect was not significant. At

0.90 m depth, on the contrary, this difference between coniferous and deciduous stands in increase of SO_4^{2-} seepage at the edges disappeared. This may also be the result of the lacking of soil seepage measurements at 0.90 m depth in two deciduous forest stands. We can conclude that our study gives a sufficient indication for a greater increase in DIN and SO_4^{2-} seepage at forest edges of coniferous forests than at edges of deciduous forests stands, but that further research is needed to investigate whether the observed difference between coniferous and deciduous forest types in DIN and SO_4^{2-} seepage enhancements at edges is a significant and general pattern.

7.5.2 Edge effects on pH and CEC

Due to (i) increased atmospheric input of the potentially acidifying pollutants NH_4^+, NO_3^-, and SO_4^{2-} (Draaijers et al. 1994; De Schrijver et al. 2007a; Wuyts et al. 2008a, 2008b, chapter 2 and 3) and (ii) higher tree growth rates (McDonald and Urban 2004; Sherich et al. 2007), increased rates of soil acidification can be expected at forest edges in comparison to interiors, as reported by Balsberg-Påhlsson and Bergkvist (1995), Jose et al. (1996), and De Schrijver et al. (1998). Our study on throughfall deposition in the same stands (chapter 3) revealed that the first 2 m of the edges received an additional potentially acidifying throughfall deposition of, on average, 686 equiv ha^{-1} y^{-1} in the *Betula* and *Quercus* stands and 2522 equiv ha^{-1} y^{-1} in the *P. nigra* stands compared to the forest interior. At the edges, we observed, however, higher pH (KCl) values and exchangeable amounts of K^+, Ca^{2+}, and Mg^{2+} in the upper 0.05 m of the mineral soil, indicating lower soil acidification rates at the edge than the interior. However, in the deeper soil horizons, this ‘positive’ edge effect faded out, and at 0.10-0.30 m soil depth, lower pH values were observed at the edge in comparison with the forest interior of the *Betula* and *P. nigra* stands. Possible reasons for these diverging patterns at 0-0.05 and 0.10-0.30 m depth are twofold. Firstly, in these type of sandy soils, most of the NH_4^+ entering the soil via throughfall deposition disappears from the soil solution between 10 and 35 cm soil depth (De Schrijver et al. 1998). Reasons are nitrification reactions, uptake by plants, or fixation at the forest floor or mineral soil, which are all proton producing processes. Also at our stands, NH_4^+ compromised only less than 10 % of the total DIN seepage flux at a depth of 0.30 m. Secondly, at the edges, additional proton buffering may occur, mainly restricted to the upper centimetres of the mineral soil. Three probable causes for the higher pH(KCl) values and higher concentrations of exchangeable K^+, Ca^{2+}, and Mg^{2+} in the upper mineral soil at the edges are discussed next, listed according to their presumed relevance.

Increased input of base cations (Bc) via throughfall deposition

Spangenberg and Kölling (2004), Devlaeminck et al. (2005), and Wuyts et al. (2008a, chapter 2) reported that, next to potentially acidifying pollutants (N+S), also throughfall deposition of K^+, Ca^{2+}, and Mg^{2+} (Bc) is enhanced due to edge effects extending to a distance of 8-30 m from the edge. These higher throughfall deposition fluxes at edges are the result of an aerodynamic edge effect, a higher LAI at the edge (Beier and Gundersen 1989), or increased canopy exchange (leaching) of base cations. From our study in chapter 2, it was calculated that, at the outer edge of a *B. pendula* stand, the input of Bc via throughfall deposition was 286 equiv ha^{-1} y^{-1} higher than in the interior, while in a *P. nigra* stand, the difference amounted to 1095 equiv ha^{-1} y^{-1}. These data indicate that almost half of the extra input of N+S at edges is already neutralised by an increased throughfall deposition of Bc.

Increased input of Bc and organic matter via mineralisation of the forest floor

The Bc input through litterfall is probably higher at the edge than in the interior, because (i) the FH layer of the forest floor had higher concentrations of K^+, Ca^{2+}, and Mg^{2+} at the edges than the interiors and (ii) we can assume that the litter flux to the forest floor is higher, based on the higher LAI values, at the edges. Moreover, given the assumed higher decomposition rates at the edge, we expect that the rate at which Bc cycle through the system is higher at the edge than in the interior, i.e. at the edge, the Bc are (temporarily) retained in the forest floor to a lesser extent than in the interior. Finally, the soil organic matter content and CEC of the mineral soil was higher at the edges, implying a higher capacity for Bc retention in the mineral soil.

Drift of lime fertilizer from agricultural applications

Due to strong sidewinds or imprecisely set equipment during agricultural applications, significant drift of lime fertilizer or bentonite clay may occur up to several meters into the forest. Szibalski and Felix-Henningsen (1999) linked higher pH (KCl) values at the edge of a *P. sylvestris* forest, next to the presence of deciduous trees, also to increased deposition of lime fertilizer and dust from the surrounding fields.

As the first two factors are presumable also forest type dependent, this may explain the interaction between the edge effect on pH(KCl) at 0-0.05 m and 0.05-0.10 m depth and the effect of forest type. Further research is needed to identify the relative importance of these four probable causes.

7.6 Conclusions

Due to increased throughfall deposition fluxes, the SO_4^{2-} and NO_3^- soil seepage fluxes are higher in edges than in interiors of both deciduous and coniferous forests, with the largest increase in NO_3^- seepage in coniferous forest edges. In the outer edge, i.e., the first 20 m, however, soil seepage fluxes of NO_3^-, and DIN in general, are strongly reduced in comparison with 20-60 m behind the edge. We assume that increased N retention in the soil and enhanced seepage of dissolved organic nitrogen are the main processes involved in the local decline in NO_3^- seepage. From higher pH (KCl) values and higher exchangeable amounts of base cations in the upper mineral soil at the edges, we conclude that, though the input of the potentially acidifying ions NH_4^+, NO_3^-, and SO_4^{2-} is higher, the rate of soil acidification is lower at edges than in interior zones because of the higher input of base cations. However, in the deeper soil horizon (0.10-0.30 m depth), the increased potentially acidifying input at edges comes to expression in lower pH values. We stress the need for further research on the fate of N and on soil acidification in forest edges.

8 General discussion and conclusions

The major aims of this thesis were (i) to assess the effects of forest type and edge structure on patterns of throughfall deposition of nitrogen (N) and potentially acidifying ions (N + S) in forest edges and (ii) to improve the insight into how contrasting forest ecosystems on sandy soils respond to the elevated deposition in forest edges in terms of nitrate seepage to the groundwater and soil acidification.

In Flanders, about 35 % of the total forested area consists of coniferous forests, mainly homogeneous plantations of *Pinus sylvestris* L. (Scots pine) and *P. nigra* ssp. *laricio* Arnold (Corsican pine) on sites naturally dominated by deciduous species (Bos en Groen 2001). Most of these pine plantations are located on the sandy soils of the Campine ecoregion in the north-eastern part of Flanders and, for a minor part, on sandy soils in West and East Flanders (Bos en Groen 2001). Due to the low base saturation (5-10 %; Neirynck et al. 2002) and the small amounts of weatherable silicate minerals (Van Ranst et al. 2002), these sandy soils are particularly sensitive to acidification (De Schrijver et al. 2006). On these poor soils, *Quercus robur* L. (pedunculate oak) and *Betula pendula* Roth (silver birch) are the intended tree species in the first successional stages of closer-to-natural forest ecosystems. Therefore, for the purpose of providing implementable management suggestions for the sandy region in Flanders, our research on the effect of forest type and edge structure was focused primarily on the four above-mentioned tree species located on sandy soils. The first part of this chapter provides a general and integrated discussion of the most relevant results of each of the six previous chapters. In the second part, we translate our results into suggestions for edge management (‘best management practices’) for the purpose of mitigating edge effects on atmospheric deposition and its negative consequences. At the end of this chapter, we provide directions for further research.

8.1 Effect of forest type on edge patterns of throughfall deposition

In the second chapter of this thesis, we compared a *B. pendula* stand with an adjacent *P. nigra* ssp. laricio stand on the issue of edge effects on throughfall deposition considered on a fine spatial resolution at steep, southwesterly-oriented edges. The edge front (i.e., the first 2 m of the edge) of the birch edge received 30 % more N and N+S deposition than the forest interior, while in the pine stand, the increase in N and N+S deposition was more than 100 %. Next to a higher level of deposition enhancement at the edge front, in the pine stand, also higher penetration depths of the edge effects were observed. As a result, the edge effect in the pine stand generated an extra input of N and N + S throughfall deposition, which was 12 and 9.4 times higher than the extra input in the edge zone with birch. Moreover, the higher extra input of N+S deposition in the pine edge, in comparison with the birch edge, was not 'compensated' by a higher extra input of base cations: the difference in extra N+S input between the forest types was twice as high as the difference in extra input of base cations. Our results thus indicate that the additional potentially acidifying effect of N+S throughfall deposition generated by edge effects in pine forests is twice as high as for birch forests. Altogether, **the first 50 m of the forest edge of the pine stand received more than twice as much nitrogen and potential acidifying ions as the first 50 m of the birch stand, but in both stands, the first 50 m of the edge obtained approximately the same amount of base cations**.

Significant short-term temporal variation in edge effects occurred in both stands, particularly for Na^+ and the base cations. For N and N+S throughfall deposition, edge effects ranged between an increase of 60 % and a decrease of 10 % in deposition in the first 50 m of the edge. This short-term variation was similar for both stands and was linked directly to variation in wind speed and direction, confirming the assumption of Draaijers (1993). However, the seasonal variation in forest edge patterns of N and N+S deposition was different for the birch and the pine stand: the birch stand displayed a large divergence in edge effects between the dormant season (with significant edge effects) and the vegetation season (without significant edge effects), while edge effects in the pine transect were similar in both seasons. Therefore, **differences in edge effects on N and N + S deposition between the pine and birch stand originated primarily from differences in edge effects during the growing season**.

In the third chapter of this thesis, we investigated whether the above-described differences in edge patterns between birch and pine can be accepted as a general finding of forest type affecting edge effects on throughfall deposition. In each of two regions, characterized by a high air concentration of NH_3 due to intensive livestock breeding, we studied three forest stands with a steep forest edge oriented towards the prevailing southwesterly winds, dominated by *Q. robur*, *B. pendula*, or *P. nigra*. Once more, the pine stands displayed larger edge effects on throughfall deposition of NO_3^- and SO_4^{2-} than did the deciduous oak and birch stands because of higher throughfall deposition enhancement at the edge and/or a larger penetration depth. **The edge effects in the oak and birch stands gave rise to an extra input of N throughfall deposition which was, on average, 90 % lower than the extra input of throughfall deposition received by the pine stands. The extra input of N+S throughfall deposition was, on average, 85 % lower in the oak and birch stands than in the pine stands**. The differences in edge effects between the forest types were larger than the differences between regions induced by meteorological dissimilarities. These results thus confirm that the differences in edge effects between the birch and pine stand in chapter 2 can be considered a consistent effect of forest type. **The fact that larger edge effects on N and N+S throughfall deposition occur in pine stands compared to oak and birch stands implies that differences in throughfall deposition input between pine and oak/birch stands are higher than previously suggested by studies on throughfall deposition in the forest interior alone**, e.g., the studies by De Schrijver et al. (2004) and Herrmann et al. (2006).

Since LAI (leaf area index), stand height, stem density, silhouette area density (i.e., the frontal crown area per 100 m²), and the morphology of needles or leaves have been shown to influence edge effects (Draaijers 1993; Wiman and Ågren 1985; Pahl 2000; chapters 4 and 6), **it is probably the complex of these factors that causes the magnitude and the penetration depth of edge effects to be higher in coniferous forest types than in deciduous ones.** Our dataset is, however, too small to specify which factors are the most important ones. For differences in interior throughfall deposition between coniferous and deciduous stands, the difference in tree height and the absence of leaves in deciduous stands during the dormant season are identified as important factors (Draaijers 1993; Houle et al. 1999; Augusto et al. 2002). For edge effects, however, these factors seem of relatively low importance since (i) in chapter 3, average tree height was higher in the oak stands than in the pine stands and (ii) in chapter 2, differences in edge effects between the birch and pine stand originated mainly from differences in the leafed (vegetation) season. It should be kept in mind that the birch edges in our study were relatively young in comparison with the pine and oak stands, which may have led to an overestimation of the forest type effect size. In general, as our birch stands grow older, particularly height and to a smaller extent also LAI can be expected to increase. But as the tree height seems of little importance for the extent of edge effects and as the difference in height between the birch stands in our study and a 65-year old birch stand (H_{dom} ~ 23 m; Jansen et al. 1996) is small, we expect the lower age of the studied birch stands to only minorly affect our general conclusion on the effect of forest type on edge patterns of throughfall deposition.

To substantiate the importance of forest type in edge effects on deposition, we assessed the exceedance of critical loads for regions of Flanders differing in the proportion of edge area and of coniferous forest in the total forest area (Wuyts et al. 2009a). Firstly, we calculated critical load exceedance (for protection of biodiversity and for root protection; CL N and CL N+S, respectively) for regions with a similar proportion of forest edge area (about 70 %), but a different share of coniferous forests (Flemish Brabant: 14 %, West-Flanders: 34 %, and Antwerp: 67 %). From these calculations, we inferred that **the impact of edge deposition on the exceedance of critical load (CL) values increases with an increasing proportion of coniferous forests**, i.e., from 112 to 556 equiv ha^{-1} y^{-1} for CL N+S and from 125 to 383 equiv ha^{-1} y^{-1} for CL N. Secondly, in regions with a higher proportion of coniferous forests, e.g., Limburg (with 50 % forest edge in the total forest area), the impact of edge deposition on total throughfall deposition and CL exceedance was larger than in regions that are much more fragmented such as East-Flanders and Flemish Brabant (where 87 and 71 % of the forest area is forest edge, respectively). For the exceedance of CL N+S, we expect the impact of forest type in the influence of edge effects on CL exceedance to be similar if the extra input of base cations in the forest edge is taken in consideration because the first 50 m of the edges of forest types would receive approximately the same amount of base cations.

8.2 Effect of edge structure on edge patterns of throughfall deposition

By means of one wind tunnel study and two field studies, we investigated the impact of edge structure on edge patterns of (throughfall) deposition. In contrast with the full-scale field experiments, the wind tunnel study (chapter 4) enabled us (i) to assess the influence of multiple factors individually under controlled circumstances and on a small time scale; (ii) to avoid the high level of variation associated with throughfall deposition measurements; and (iii) to explain the observed impact of edge structure. Scale model trees were shaped and configured to obtain eight stand structure configurations, encompassing a combination of stem densities, crown depths, and edge transition types. The effect of the latter was assessed by contrasting steep transitions (i.e., edges with an abrupt transition from open area to forest) with gradual transitions (i.e., edges with a gradual increase of vegetation height from open area to forest). Patterns of wind speed, turbulence, and simulated dry deposition of a Cl^- aerosol were significantly influenced by edge transition type and stem density, and, to a small extent, also by crown depth in the case of the steep transition.

8.2.1 Effect of gradual edges on edge patterns of throughfall deposition

In the wind tunnel study, **a gradual transition** or forest edge shaped by a gradual increase of vegetation height (Fig. 4.2) **induced a decrease in dry deposition in the forest edge zone (i.e., two tree heights from the edge of the forest model) by 66 %. With a gradual transition, the deposition enhancement at the edge front relative to the interior was significantly lower compared to forests with a steep transition, with the largest differences in the sparse forests.** Wind speed measurements revealed that deflection of the wind flow by the gradual edge vegetation and drastic deceleration of wind speed before the front of the edge of the forest caused the deposition enhancement at the edge to be lower than at steep transitions. Differences in penetration depth between the forests with different transitions were small and non-univocal. Keeping in mind that (i) deposition patterns of fine (submicron) particles or gases, which contain potentially acidifying and eutrophying sulfate, nitrate, and ammonium, are less pronounced than those of coarse aerosols such as Cl^- and that, (ii) in this wind tunnel study, only edge effects on dry deposition originating from aerodynamic processes are considered, we can conclude that gradual edges have the potential to mitigate edge effects.

Nonetheless, our findings needed validation in full-scale forests. Accordingly, in chapter 5, we tested the ‘theoretical’ mitigating effect of gradual forest edges on throughfall deposition of Cl^-, N, and S by collecting throughfall deposition at adjacent steep and gradual edges at three forest stands. **At the smoothest gradual edge, which closely linked up with the forest behind it, the extra N and S throughfall deposition the forest received due to edge effects was lower than at the adjacent steep edge.** Next to a reduction in deposition enhancement at the edge front (as observed in the wind tunnel), also a halving of the penetration depth occurred at the smooth edge. Our results indicate that **a gradual transition does not solely affect the aerodynamically induced edge effects on dry deposition, but it can also**

significantly decrease the penetration depth of microclimatic gradients causing edge effects on dry deposition of gases and canopy exchange.

In addition, this study points to **an important prerequisite for the mitigation of edge effects on deposition: the gradual edge vegetation should be triangular-shaped in cross-section and its dimensions should be in proportion with the height of the forest stand behind it, so that the edge vegetation smoothly links up the open area and the forest.** In the case of insufficient height or inadequate shape of the gradual edge vegetation, only small or insignificant decreases in throughfall deposition were observed. Given the underlying processes of deflection and deceleration of wind flow by the gradual transition, it is clear that the shape and size of the gradual edge vegetation in front of the forest are of high importance. Naturally developing forests in the immediacy of a shoreline exposed towards the prevailing winds (e.g., in Belgium, The Netherlands, and France, the shoreline of the North Sea, the English Channel, and the Atlantic Ocean is oriented towards the prevailing southwesterly winds) are relentlessly exposed to high wind speeds. These forests show a tendency for developing naturally ascending edges, an advantageous configuration in which rows of smaller trees protects the adjacent tree rows from high wind speeds and wind stress. This implies that a gradual transition at edges indeed gradually deflects the wind flow and protects the forest stand behind it.

A potential drawback from the application of a gradual edge may be the deposition on the gradual edge vegetation itself. In the wind tunnel study, a gradually ascending edge was still profitable when this deposition was taken into account: **the mitigation of deposition in the forest edge zone caused by the presence of a gradual edge was at least 1.5 times larger than the deposition on the gradual edge vegetation**. The largest gain resulting from the presence of a gradual edge occurred in the low density forests, where the mitigation of deposition in the forest was about eight times higher than the deposition on the gradual edge vegetation. **Similarly, at the closest-to-ideal gradual transition of the field study, the mitigation of the input of N and S deposition caused by edge effects was not compensated by higher N and S deposition in front of the edge of the forest, i.e., on the gradual edge vegetation itself.** In the case of the steep transition, the extra N and S deposition in the stand and in front of the stand caused by edge proximity was, during the measuring campaign, three times higher than in the case of the gradual transition.

While the input of N+S throughfall deposition can be reduced by the presence of gradual edge vegetation, the effect on net potential acidifying input (N+S input minus input of base cation) was not assessed. We did not measure base cation deposition in chapter 5, but from chapter 2 we can deduce that (i) throughfall deposition fluxes of Mg^{2+} show similar edge patterns as Cl^-, but (ii) the deposition enhancement at the edge front and the penetration depth of edge effects on Ca^{2+} and particularly K^+ are smaller than those for Cl^- and approximate those for N and N+S. Because the mitigating effect of a gradual transition depends on the density or LAI of the forest stand (chapter 4) and with LAI and because the differences in edge effects between Cl^- (and probably also the base cations) on the one hand and N and S on the other hand alter with LAI, **it is probable that the effect of a gradual edge vegetation on the net potentially acidifying input interacts with LAI**. We can only make an estimation of the balance of N+S versus base cation input via throughfall deposition for the stand we considered in our field study. At the gradual transition, the deposition enhancement in the first 64 m of the edge (the IFEE factor) was decreased to the same extent for Cl^- and SO_4^{2-} (from 1.23 to 1.07 for Cl^- and from 1.28 to 1.08 for SO_4^{2-}), while for NH_4^+ and NO_3^-, edge effects were reduced almost completely. Given that, in general, the input of base cations via throughfall deposition is much less than the input of N and S via throughfall deposition, particularly in coniferous forests [chapter 2 and De Schrijver et al. (2004, 2008)], **it is likely that the reduction of N+S input via throughfall deposition is higher than the reduction in base cation input via throughfall deposition**. This implies that layout of gradual edge vegetation at steep edges would lead to a lower net potentially acidifying input via throughfall deposition, and with it, a lower rate of soil acidification.

8.2.2 Effect of stem density and leaf area index (LAI) on edge patterns of throughfall deposition

In the dense model forests of the wind tunnel study, the dry deposition in the forest edge zone (i.e., the first two tree heights from the edge of the forest model) **was 40 % lower than in the sparse model forests because the edge effect extended further in the sparse forests than in the dense ones.** This was caused by (i) the stronger deceleration of the wind speed to a forest interior level in dense forests, which was also observed by Dupont and Brunet (2008a) in a large-eddy simulation, and by (ii) the significantly lower turbulence within the canopy of the forest edge of the dense forests. From the processes involved, it is clear that this so-called effect of stem density in the wind tunnel study (chapter 4) is in fact an effect of canopy density, which was uniformly related to stem density as canopy silhouettes were identical. In full-scale forests, the relationship between stem density and LAI depends on tree species and stand structure development. So, **given the underlying processes of smaller penetration depths in dense forests, not stem density but canopy density is of direct relevance**. If the canopy silhouette is a constant feature, LAI is a sufficient and easy-to-measure indicator for canopy density.

In chapter 6, we checked the effect of LAI in full-scale forest stands, at observational sites and experimentally thinned full-scale forest stands of *P. nigra* and *P. sylvestris*. **Leaf area index (LAI), as a measure for canopy density, was found to be a key driver in the processes causing edge effects on atmospheric deposition as LAI determines both the magnitude and penetration depth of edge effects.** The level at which throughfall deposition is enhanced at the edge front of the forest increases with increasing LAI, but in contrast with Cl^-, this increase levelled of at an LAI of 2 onwards for SO_4^{2-}, NO_3^-, and SO_4^{2-}. The relation of the penetration depth with LAI was ambiguous due to the difficult demarcation of penetration depths in sparse forests. However, the increasing trend of penetration depth of edge effects on wind speed and deposition with LAI, which was observed in our wind tunnel study and in simulation studies by Pahl (2000) and Dupont and Brunet (2008a) was substantiated for higher LAI values. The overall effect of LAI on the N and S deposition in a fixed edge zone of 64 m was quadratic in shape: with increasing LAI, the deposition enhancement in the entire edge zone increased at low LAI values, stagnated at LAI = 2, and decreased at higher LAI values. This is a reasonable concept: **in very sparse forests, edge effects penetrate very far into the stand but the level of enhancement at the edge front is negligibly small, while in very dense forests, edge effects give rise to high deposition enhancement factors at the edge front, but penetrate less deep into the forest**.

In order to understand the impact of LAI on potential soil acidification in forest edges, the incoming acidifying pollutants (N+S) should be balanced against the incoming base cations. As mentioned above, Mg^{2+} throughfall deposition - only to a small extent influenced by canopy exchange - would be subject to similar edge patterns as Cl^-, but edge patterns of Ca^{2+} and particularly K^+ throughfall deposition would approximate those of N and N+S throughfall deposition (chapter 2). So, for base cations, we can roughly expect the pattern of IFEE with LAI to lie between the pattern for Cl^- and those for N and S. For Cl^-, the enhancement of deposition in a fixed edge zone of 64 m (IFEE) started leveling off at LAI = 2 and can be expected to peak at an LAI between 3 and 4 (Fig. 6.4). For LAIs < 1.5, the enhancement factors of N and S throughfall deposition in the first 64 m versus the forest interior were lower than for Cl^-. But from LAI = 2 onwards, the enhancement factors for N and S increased to a lesser extent than for Cl^-, and thus, presumably, also for the base cations, or even showed a decrease. Knowing that the interior throughfall deposition is more than three times higher for N+S deposition than for the base cations [chapter 2, De Schrijver et al. (2004, 2008)], this implies that, **the higher the LAI, the smaller the difference between the input of N+S and the input of base cations**. We summarised this theory in Fig. 8.1, in which the mean throughfall deposition of N+S and the base cations in an edge zone of 64 m was calculated, based on the forest interior deposition in the pine stand of chapter 2 and the modelled IFEE factors from chapter 6 of (i) Cl^- for Ca^{2+} and Mg^{2+} and (ii) N and S for K^+, N, and S throughfall deposition. At LAI ~ 3.5, we expect the smallest difference between N + S deposition and the base cations: the input of N and S throughfall deposition in the first 64 m from the edge would be almost equal to the interior deposition (IFEE close to 1.0), while the input of base cations via throughfall deposition in the first 64 m from the edge would be about two times higher than in the interior. Hence, it is probable that (i) the higher the LAI is, the higher the proportion of potentially acidifying input is that is ‘neutralised’ by the input of base-cations and that (ii) at LAI ~ 3.5, the first 64 m of the edge is subject to a lower net acidifying input (the input of N+S minus the input of base cations) than the forest interior, although not all the input of N+S throughfall deposition is neutralised by the input of base cations via throughfall deposition.

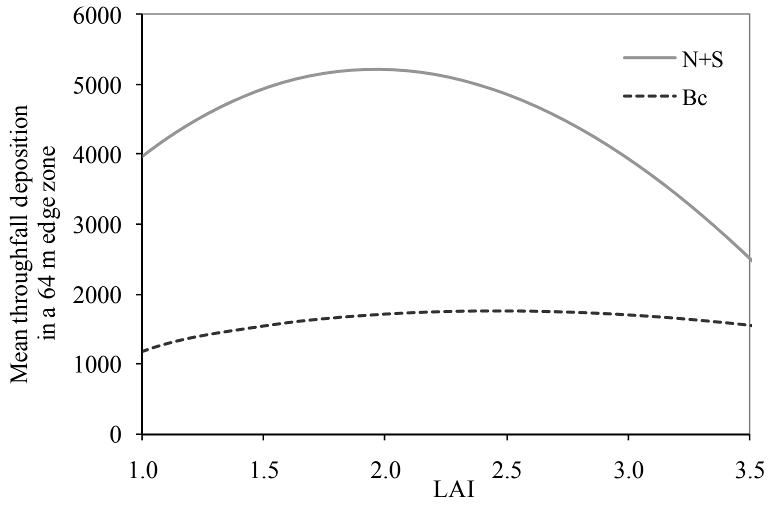

Fig. 8.1: Hypothetical response of the mean throughfall deposition of potentially acidifying ions (N+S) and the base cations (Bc) in the first 64 m from the edge to a changing LAI of a pine stand

8.2.3 Effect of other edge structure characteristics on edge patterns of throughfall deposition

The effects of other edge structure characteristics than edge transition type and LAI were considered in the wind tunnel study in chapter 4 and the field study in chapter 6. **In the wind tunnel study, crown depth was found to affect to a small extent the increase in dry deposition caused by edge effects.** This was not confirmed by our field study in chapter 4, which was most probably due to the relatively small range of crown depth in which the edge effects were assessed. Similarly, in field study of chapter 4, we did not find any relation between edge effect measures and stand height, basal area, or stand volume because of the same reason. It can be expected, however, that **mean tree height is relevant for the dimensions of edge effects as the difference between the average tree height and the average height of the vegetation in front of the forest determines, via the roughness length, the magnitude of roughness transitions** (here, forest edges). Draaijers (1993) found the penetration depth of edge effects to depend on the mean tree height or stand height, as he expressed the penetration depths as a multiple of tree heights.

8.3 Edge effects on throughfall deposition vs edge effects on wet deposition, dry deposition, and canopy exchange

In this thesis, we only considered edge effects on throughfall deposition, with exception of the wind tunnel study in chapter 4. The observed edge effects on throughfall deposition may originate from edge effects on wet deposition, dry (+ occult) deposition and/or canopy exchange.

Edge effects on throughfall water volume in our stands were either not observed or restricted to the first meters of the edge, which agrees with the findings of Hasselrot and Grennfelt (1987), Draaijers et al. (1988), Devlaeminck et al. (2005), and Herbst et al. (2007). Klaassen et al. (1996) also observed that distance to the forest edge had little effect on throughfall water volume: due to enhanced wind speed close to the edge, the higher evaporation rate at edges was counterbalanced by the lower water storage capacity. Hence, increased throughfall water volumes at the front of a windward edge are the result of more rain penetrating the edge from the side. It is therefore unlikely that edge effects penetrating as far as 60 m from the edge are the result of increased wet deposition.

Devlaeminck et al. (2005) calculated the canopy exchange of the base cations at the edge of a beech forest based on the canopy budget method by Ulrich (1983). Na^+, Cl^-, K^+, Ca^{2+}, and Mg^{2+} are deposited mainly as coarse particles (> 1 µm), and, in the canopy budget model, it is assumed that particles containing these ions are subject to the same deposition processes and have more or less the same deposition velocities. Devlaeminck et al. (2005) found canopy exchange (leaching) of K^+ and Ca^{2+} to be lowered within the first 30 m from the edge. We also estimated the canopy leaching of K^+, Ca^{2+}, and Mg^{2+} based on the data in chapter 2 and on Ulrich (1983). Since dry deposition of larger particles such as Na^+, Cl^-, K^+, and Ca^{2+} are presumed by the model to have similar edge effects, the small, or even negligible, edge effects on Ca^{2+} and particularly K^+ throughfall deposition in comparison with Na^+ and Cl^- likely resulted from a smoothing influence of canopy exchange. Because throughfall deposition of K^+ originates to a large extent from canopy leaching (De Schrijver et al. 2004; Staelens et al. 2006), at the edge, the higher canopy exchange (uptake) rates almost completely counterbalanced the presumed higher dry deposition levels. Ca^{2+} and Mg^{2+} displayed more pronounced edge effects on throughfall deposition than K^+, which can be explained by the larger contribution of dry deposition to the throughfall flux of Mg^{2+} and Ca^{2+} in comparison with K^+ (Draaijers et al. 1997; De Schrijver et al. 2004). In contrast with these estimations, Hasselrot and Grennfelt (1987) found K^+ throughfall deposition corrected for sea-salt, as a measure for K^+ canopy leaching, not to be subject to significant edge effects. Beier and Gundersen (1989) suggested that edge effects on leached ions are related to changes in LAI, which would imply higher exchange rates of base cations such as K^+ at the edges of the stands in chapter 2 (LAI was increased in the first 9 m of the edge). Nonetheless, the results of chapter 2 argue for lower canopy leaching of base cations or no influence of edge proximity on canopy leaching, as the edge patterns of the base cations resemble more the edge patterns for Cl^- and Na^+ as the contribution of canopy leaching to the throughfall deposition decreases (from K^+ to Ca^{2+} and Mg^{2+}). Hence, **we can conclude that the increased input of base cations via throughfall deposition does not originate from increased canopy leaching at the edge, and can be considered as an external extra potential buffering of acidification**.

The canopy budget method developed by Ulrich (1983) and extended by de Vries et al. (1998) becomes less reliable when it comes to canopy exchange (uptake) of NH_4^+. Furthermore, in the canopy budget model, canopy exchange of NO_3^- is assumed to be negligible, while it has been frequently pointed out that NO_3^- can absorbed by the canopy, although to a lesser extent than NH_4^+ (Lovett and Lindberg 1984, 1993; Neary and Gizyn 1994; Houle et al. 1999;

Stachurski and Zimka 2000, 2002). Sulphur is assumed to act as a conservative element with respect to the canopy, as stomatal uptake of SO_2 is thought to be balanced by canopy leaching of SO_4^{2-} (Lindberg and Lovett 1992; Draaijers et al. 1997; Stachurski and Zimka 2002). SO_4^{2-}, NO_3^-, and NH_4^+ are mainly deposited as gases and sub-micron aerosols, and deposition of these ions is thus expected to be driven by the same processes, in contrast to Na^+, Cl^-, K^+, Ca^{2+}, and Mg^{2+}. Hence, edge patterns of SO_4^{2-}, NO_3^-, and NH_4^+ throughfall deposition were much alike, although the enhancement factors (MEI and IFEE) were usually smaller for NO_3^- and, particularly, for NH_4^+. The latter is the only ion in all studies of this thesis that occasionally shows deposition 'enhancement' factors close to or even below 1, i.e., no or 'negative' edge effects with lower throughfall deposition fluxes at the edge than in the forest interior (e.g., at the Dombergheide and Neigembos sites in chapter 5). This is not the pattern one expects when lower canopy exchange (uptake) of NH_4^+ is assumed at the edge. Instead, no edge effect on canopy uptake of NH_4^+ or higher uptake of NH_4^+ at edges can be assumed. The same assumption can be made for canopy uptake of NO_3^-, but as NO_3^- is taken up by the canopy to a lesser extent than NH_4^+, its edge patterns differ with those of SO_4^{2-} to a lesser extent than those of NH_4^+ do. **To conclude, we can presume that the increase in N and N+S input via throughfall deposition at edges is most probably an underestimation of the actual edge effect induced increase in N and N+S input to the entire ecosystem.**

8.4 Edge patterns of N seepage and soil acidification

In most edges considered in this thesis, edge effects on atmospheric deposition caused the input of N and potentially acidifying ions to be increased within the first 8 to 64 m from the edge. In chapter 7, we investigated whether these increased deposition fluxes were reflected in increased rates of dissolved inorganic N (DIN; mostly NO_3^-) seepage to the groundwater and of soil acidification at edges of both deciduous and coniferous forests. We indeed found, in both deciduous and coniferous forests, **higher SO_4^{2-} and NO_3^- soil seepage fluxes within the forest edge up to 80 m from the edge than in the forest interior**. However, the patterns of DIN seepage across the forest edge were not a straightforward reflection of the N input via throughfall deposition: in the outer edge, i.e., the first 10 to 20 m from the forest edge, soil seepage fluxes of NO_3^-, and DIN in general, were strongly reduced in comparison with 20 - 60 m behind the edge. From additional measurements of N stock in the soil and N uptake by tree biomass, **increased N retention in the soil is assumed to be the main process involved in the local decline in NO_3^- seepage within the first 20 m from the edge**. We suspect that edge gradients in microclimate, in deposition of base cations, in biomass production, and in decomposition of the forest floor material are the causes of the increased N retention capacity in the soil at the outer edge. Firstly, higher soil temperatures and higher Ca concentrations in the litter increase decomposition rates (Reich et al. 2005; Hobbie et al. 2007) at the edge, which was confirmed by additional measurements of the biomass in the fermentation and humus layer of the forest floor. Consequently, the N retention in the forest floor (i.e., the ectorganic layer) was lower at the edge and presumably up to 60 m from the edge than in the forest interior. Secondly, we expect that the flow of organic material via the forest floor to the mineral soil is higher at the outer edges (0 - 20 m) than at 20 - 128 m. The biomass of fresh leaves, and thus presumably also the flux of organic material to the forest floor, was higher in the first 10 m of the edge than further down the edge and in the forest interior. Thus, potentially more N is adsorbed onto the organic matter and is retained in the soil (Kaiser and Zech 2000), which is confirmed by measurements by Vandenbruwane (2008). Next to adsorption, dissolved organic nitrogen (DON) can also be leached from the soil profile (Qualls et al. 2002). In the first 10 m of the edge, Vandenbruwane (2008) observed higher levels of DON soil seepage than in the forest interior. It is clear, however, that further research is needed to verify our assumptions on the fate of N in the forest edge.

From higher pH (KCl) values and higher levels of exchangeable base cations in the upper 0.05 m of the mineral soil at the studied edges, we concluded that, although the input of the potentially acidifying ions NH_4^+, NO_3^-, and SO_4^{2-} is higher, the rate of soil acidification is lower at edges than in interior zones. However, **in the deeper soil horizons, this edge effect with enhanced pH values at the edge faded out, and at 0.10 to 0.30 m soil depth, even lower pH values were appreciable in the edges of the *B. pendula* and *P. nigra* stands in comparison with the forest interiors**. The possible causes for these patterns of higher pH (KCl) values and levels of exchangeable K^+, Ca^{2+}, and Mg^{2+} in the upper mineral soil at the edges are increased input of base cations at the edge and the proton producing processes related to NH_4^+ deposition occurring below the upper centimeters of the mineral soil. Firstly, data from the field study in chapter 2 indicated that almost half of the extra input of N+S at edges is being neutralised by the increased input of base cations (Bc, $K^+ + Ca^{2+} + Mg^{2+}$) at edges. This increase in Bc deposition can be the result of aerodynamically induced edge effects and/or higher LAI values. Moreover, the study on the impact of LAI on edge effects in pine stands pointed out that, with increasing LAI, the proportion of potentially acidifying input ‘neutralised’ by the input of Bc at the edge increases to a point at a relatively high LAI level where the edge is subject to a lower net acidifying input (the input of N+S minus the input of Bc) than the forest interior. Secondly, input of Bc and organic matter is assumed to be higher at the edge than in the forest interior due to increased decomposition and mineralisation rates. Thirdly, due to strong winds or imprecisely set equipment during agricultural applications, significant drift of lime fertilizer or bentonite clay may occur up to about 20 m into the forest. The latter is inherent to the high land use pressure characterising Flanders.

Furthermore, the soil seepage data confirm the relevance of forest type: **the largest increase in NO_3^- seepage was observed in the edges of coniferous forests**. Surprisingly, edge effects on soil acidity were most pronounced in the pine stands, i.e., the increase in pH (KCl) at the edges was the highest for the pine stands. From the data in chapter 2, however, we calculated that the difference between the extra input of potentially acidifying ions and the extra input of base cations was about 9 times higher in the pine stand than in the birch stand, which implies that edge effects in pine stands would give rise to higher acidification rates than edge effects in birch stands. From our study, we cannot explain this incongruity, but one (and probably both) of the birch sites had formerly been planted with *Picea abies* (L.) Karst., and the low pH (KCl) values at the edge can be a legacy of the acidifying effects of the *P. abies* (Augusto et al. 2002; Reich et al. 2005).

8.5 Implications for edge management

The overall aim of this thesis was to provide edge management suggestions for the purpose of mitigating edge effects on atmospheric deposition and consequently negative ecosystem responses such as nitrate seepage, soil acidification, and loss of biodiversity. Since edge effects on the input of N and S via throughfall deposition are affected by forest type and edge structure, we are able to make suggestions for edge management for the purpose of mitigating edge effects on atmospheric deposition. Although strong and global emission reductions are a necessary prerequisite for the protection of ecosystems, no vast decrease in N emissions is expected in the short term (Galloway et al. 2003; Cofala et al. 2007), pointing to the necessity of applying mitigating measures. Moreover, model predictions for 2070-2100 by Haugen and Iversen (2008) and Debernard and Røed (2008) reveal that annual mean wind speeds and the occurrence of high daily wind speed will increase in the region of Belgium due to climate change. Given that the extent of edge effects increases with increasing wind speed (chapter 2), the need for mitigating measures will grow with time. Therefore, **we argue for a well-considered design and layout of existing and new forest edges. The following suggestions provide a basis for a thorough consideration of different edge management options.**

Firstly, the more pronounced edge effects on N and S deposition in pine stands in comparison with oak and birch stands strengthen the idea, based on the higher level of throughfall deposition, NO_3^- seepage, and soil acidification in the forest interior (De Schrijver et al. 2004, 2007b, 2008; Herrmann et al. 2005, 2006), that **conversion of high-density pine plantations into deciduous forest types with oak and birch will reduce the input of eutrophying and acidifying ions to forests. Conversion of pine plantations to oak or birch forests would decrease the extra N and N+S throughfall deposition caused by edge effects by 90 % and 85 %, respectively.** The choice of the silvicultural strategy of the conversion process has distinct effects on the biogeochemical cycling throughout the process (Weis et al. 2006; De Schrijver et al. in press; Gielis et al. in press). In addition, at forest edges, tree stability is also of great importance when weighing up the pros and cons of different conversion scenarios, such as clear-cut or continuous cover shelterwood- or group-cut (Stacey et al. 1994; Gardiner et al. 1997, 2005).

Secondly, our results indicate that **gradual forest edges have the potential for mitigating edge effects on N and S deposition, mainly via a reduction of the level of deposition enhancement at the edge front**. Consequently, **we suggest the lay-out of gradual transitions at steep edges with, e.g., an herbaceous fringe, a shrub belt, and a forest mantle**. This can be achieved by stimulating natural regeneration or by planting and sowing shrubs and herbs (i) in the open land in front of the steep edge of the forest after, for example, retraction of the fence ('forest expansion') or (ii) in the first meters of the forest stand after cutting the trees ('forest receding'). From the view of forest habitat, the first management option would be preferential, because the area of forest interior habitat is higher than in the second option, but this management choice is determined primarily by how the forest is embedded in the landscape matrix, by local land use pressure, and by ecological, economical, and recreational priorities. Our management suggestion is in line with the one provided by Magura (2002) and Wermelinger et al. (2007) to conserve and promote biodiversity, i.e., efforts should be made to create gradual, more diverse edges (i.e., 'high quality edges') at forests with poorly developed, abrupt edges. The layout of gradual edges would be most profitable at edges where the most pronounced edge effects occur. In Western Europe, these are edges oriented towards the south- to west, which are at the same time the warmest and lightest kind of edges, and thus encompass the highest potential for developing gradual edges with high species diversity. Also, according to and Agster and Ruck (2003) and Dupont and Brunet (2008b), gradual edges and, particularly, edges with an angle of 45° provide better shelter for wind damage to the forest behind it than steep edges do. **In addition, we emphasize the importance of the shape and size of the gradual edge vegetation relative to the forest stand behind it for reducing edge effects on atmospheric deposition. Gradual edge vegetation that smoothly and closely links up with the forest stand behind it**, such as the one at the Dombergheide site in chapter 5 (Fig. 5.2), **generates the ideal transition to mitigate these edge effects.** If a sufficiently smooth gradual edge is intended, than an edge angle of maximum 45° should be considered, which implies that a minimum distance of one tree height should be reserved for the layout of the gradual edge vegetation. For reasons of nature conservation, it is often suggested that horizontally undulating gradual edges provide more diverse environmental conditions with a higher associated species diversity than horizontally straight gradual edges. This layout most probably also affects wind flow at the edge. Whether this would lead to significant changes in deposition patterns at the edge remains to be studied.

Thirdly, the leaf area index, as a measure for canopy density, determines the penetration depth of edge effects and the level of deposition enhancement caused by edge effects in pine stands. Edge effects on throughfall deposition are smaller in sparse (with low LAI) and in dense forests (with high LAI) than in forests with intermediate LAI levels: **forest stands should be kept dense to decrease the penetration depth or very sparse to limit the level of deposition enhancement.** But as, for the base cations, this peak in-between sparse and dense forests probably occurs at higher LAI levels than for N and S deposition, for the purpose of mitigating edge effects, **we suggest managing pine forests so as to obtain the highest canopy density possible**. This does not imply that pine stands should be kept at a high level of stem density, but thinning measures should be started with in an early stage and carried out frequently, so that deep and large tree crowns are created. We, however, point to the fact that this suggestion is based on a limited data set collected only in *Pinus nigra* and *P. sylvestris* stands.

To sum up, we suggest, with increasing impact on the edge patterns of deposition but decreasing ease of implementation: (i) early and frequent thinning of forests, (ii) layout of gradual edge vegetation at steep edges, and (iii) conversion of coniferous pine plantations to deciduous forest types. In contrast with forest type, which affects throughfall deposition in the forest edge (chapters 2 and 3) and in the forest interior [chapter 3 and De Schrijver et al. (2004, 2008)], the presence of gradual edge vegetation can only influence throughfall deposition in the forest edge. However, **for the purpose of mitigating edge effects, gradual edge creation is a more straightforward process and easier to implement than forest type conversion**.

These management suggestions were formulated based on measurements (i) at edges oriented towards the prevailing southwesterly winds, where edge effects on atmospheric deposition are the most pronounced (Pahl 2000), (ii) mainly on sandy soils (podzols), which are the most sensitive to soil acidification and nitrate seepage (Van Ranst et al. 2002; Rothwell et al. 2008). Nonetheless, the above mentioned management suggestions also apply for edges with an edge orientation other than the prevailing wind direction, but their mitigating influence is expected to be much smaller as edge effects are already small or even negligible at these less exposed edges. Furthermore, in Flanders, the regions of sandy soils contain the majority of the coniferous pine plantations (Geudens et al. 2006), where the potential of forest type conversion as mitigating measure is assessed. For forests on richer soils, the mitigating effects on deposition will be similar, but edge patterns of nitrate seepage and soil acidification will probably be different.

A recurring concern regards the impact of a large-scale application of edge effect mitigating measures on air quality and deposition to the surrounding ecosystems. According to the Flemish report on the state of the environment (Van Steertegem 2007), in 2006, the potentially acidifying emission amounted to 9 575 million equivalents. Let us assume the largest decrease in N+S deposition enhancement possible by applying mitigating measures on the entire forested area of Flanders, i.e., from an IFEE factor of 1.50 to 1.00 for coniferous forests and from 1.20 to 1.00 for deciduous forests with the abstraction of edge orientation. Forests in Flanders are characterized by 60 % of edge and 40 % of interior zone in the total forested area (De Schrijver et al. 2007a) and by 56 % of deciduous and 44 % of coniferous forest area, and are subject to an interior deposition of 3000 and 5000 equivalents ha^{-1} y^{-1}, at most, for deciduous and coniferous forests, respectively (De Schrijver 2007, chapters 2 and 3). Based on these data, we estimated that an overall application of mitigating measures would lead to an average annual decrease of 998 equivalents per ha, at most, or 148 million equivalents per year for the entire region of Flanders (surface area: 13 522 km²; forest index: ± 11 %). **Approximately 8 % of the actual Flemish annual emission of potentially**

acidifying pollutants is captured by forests, and a hypothetically complete reduction of edge effects would reduce the proportion of the annual emission captured by forests to 6.5 %. These estimations indicate that the impact of a hypothetical large-scale application of edge effect mitigating measures on national or regional air quality would be very small, which confirms the hypothesis by Erisman and Draaijers (1993) that edges do not influence deposition on a national or regional scale. On a smaller spatial scale (local or landscape scale), this is a more complex matter as several other factors than air concentration should be taken into account, such as changes in effective roughness of the landscape surface (Veen et al. 1996). Wiman and Ågren (1985) simulated aerosol depletion and deposition in forests along transects across the edge, and found the pollutant concentration in the forest interior not to be influenced by the extent (deposition enhancement and penetration depth) of edge effects. Apparently, air flows are sufficiently mixed so that air concentrations are only decreased in the direct vicinity of forest edges. But a strong turbulent flow across rougher surfaces transports gases and aerosols towards the land surface more efficiently than quieter flows across an aerodynamically smoother surface (Veen et al. 1996). Thus, forest edges must not be considered as considerable pollutant diluters for the surrounding landscape. We conclude that a local increase in deposition on ecosystems (such as heathland) in the vicinity of a forest of which edge layout is altered to mitigate edge effects within the forest is expected to be negligible. It should be stressed, however, that this topic needs further investigation.

8.6 Suggestions for further research

Although this thesis has increased our understanding of the factors that influence edge patterns of throughfall deposition and of how these patterns of N and N+S throughfall deposition are reflected in nitrate seepage and soil acidification, several issues deserve and need further research. Firstly, more research is needed to substantiate and fully explain our findings on nitrate seepage and soil acidification at forest edges. Also, experiments should be set up in forests on richer soils and/or dominated by other tree species than the ones considered in this thesis to validate our findings. Our assumptions on the processes involved in the fate of N and in soil acidification at forest edges need thorough verification. Existing models or decision trees that predict the feasibility of nitrate leaching based on N input and soil characteristics (Dise et al. 1998; Gundersen et al. 1998; MacDonald et al. 2002; Kristensen et al. 2004; Brumme and Khanna 2008; Rothwell et al. 2008) need adaptation for forest edges, since forest edges are increasingly important features of landscapes worldwide.

Secondly, in terms of ecosystem responses, we only considered nitrate seepage and soil acidification, while the causal relation between increased N and potentially acidifying deposition in forest edges on the one hand and tree vitality, composition of understory vegetation and lichens, and abundance and composition of myccorhizal fungi on the other hand are largely unidentified. Furthermore, the impact of the management suggestions provided on these ecosystem responses is unknown. Finally, ecosystem recovery has been the subject of study (e.g., by clean rain roof experiments of the EXMAN, NITREX, RAIN, and Roof projects; Tietema et al. 1998; Beier et al. 2003; Corre and Lamersdorf 2004; Boxman and Roelofs 2006), but only in the interior of forests. The question remains whether the lower input of N and potentially acidifying deposition due to well-considered edge management encompasses a potential for ecosystem recovery and, if so, on what time scale it can be expected.

Summary

Due to forest fragmentation, forest edges as ecotone boundaries between forest and open area such as heathland, pasture, or agricultural land are increasingly dominant features in landscapes around the world. In Flanders, the northern part of Belgium, forests are strongly fragmented: the relative amount of forest edge in the total forested area amounts to almost 60 % when considering an edge depth of only 50 m. Forest edges have a vast influence on the flux of nutrients or pollutants from the atmosphere towards the forest ecosystem. Due to these so-called edge effects, throughfall deposition of nitrogen (N) and potentially acidifying pollutants [sulfate (SO_4^{2-}), nitrate (NO_3^-), and ammonium (NH_4^+)] is enhanced from the front of the edge to up to more than 100 m from the edge, by up to a fourfold compared with the forest interior. On top of that, Flanders suffers from N and SO_4^{2-} deposition levels that are among the highest in Europe. In the interior of forests, high levels of N (NH_x + NO_y) and sulphur (S, SO_x) deposition are associated with N saturation, increased levels of nitrate seepage, soil acidification, and eventually vitality decreases, floristic shifts, and declines in forest biodiversity.

These facts argue for mitigating measures that reduce the input of N and potentially acidifying pollutants in forest ecosystems. In order to formulate recommendations for the design and management of forest edges for the mitigation of edge effects, we tried to get insight into (i) the impact of forest type and edge structure on the extent of edge effects on throughfall deposition, and (ii) the impact of enhanced input of N and potentially acidifying pollutants on two ecosystem responses at edges, i.e., nitrate seepage and soil acidification. For this reason, we sampled throughfall deposition, soil solution, and soil and determined stand structure characteristics along transects perpendicular to edges of contrasting forest stands, from the edge front up to a distance of 128 m from the edge. We focused on forest ecosystems on sandy soils, which are most vulnerable to nitrate seepage and soil acidification, and in regions with intensive livestock breeding and consequently high NH_3 emissions. In addition, a wind tunnel study was performed in which we simulated edge effects in model forests with differing structure.

Two field studies demonstrated that the throughfall deposition of N and potentially acidifying deposition in pine stands are subject to edge effects that penetrate the forest to a deeper extent and cause a higher level of deposition enhancement at the edge front than in deciduous oak and birch stands. The additional input of N and potentially acidifying pollutants in the oak and birch stands was 10 to 15 times lower than in the pine stands. The difference between a pine and an adjacent birch edge in additional input was higher for the potentially acidifying pollutants than for the base cations. This indicates that the higher additional input of potentially acidifying pollutants in pine edges is not compensated by a higher input of 'potentially neutralising' base cations, so higher rates of soil acidification can be expected at pine edges in comparison with birch edges.

By means of a wind tunnel study and two field studies, the impact of edge structure on edge patterns of (throughfall) deposition was explored. In the wind tunnel study, a gradual forest edge (i.e., a gradual transition of vegetation height in front of the forest edge) deflected the wind flow and decelerated wind speed in front of the forest. Consequently, it induced a decrease in the level of deposition enhancement at the edge front in comparison with a steep forest edge, particularly in sparse forest models. A field study, in which throughfall deposition patterns were measured in adjacent steep and gradual edges, confirmed the wind tunnel results on the lower level of deposition enhancement at gradual edges, but also pointed to a decrease in penetration depth of edge effects. With a steep transition, the additional N and S deposition induced by edge effects in the forest was, on average, three times higher than the additional N and S deposition in the forest and the gradual edge vegetation itself in the case of a gradual transition. In addition, the field study pointed to an important prerequisite in terms of shape and size of the gradual edge vegetation for the mitigation of edge effects on N and potentially acidifying deposition.

In the wind tunnel study, edge effects penetrated further into a sparse model forest than in a dense forest, which was caused by the stronger deceleration of wind speed and turbulence in the dense forest. This effect of canopy density, expressed as leaf area index (LAI), was verified in a field study on Corsican and Scots pine stands with differing LAIs. LAI was found to be a key driver in the processes causing edge effects on deposition as it determined both the level of deposition enhancement at the edge front and the penetration depth of the edge effects. The overall increase in deposition in the entire edge zone of forests displayed an optimum at an intermediate LAI level, with small penetration depths in forests with high LAI and low levels of deposition enhancement in forests with low LAI.

In both deciduous and coniferous forests, higher NO_3^- seepage fluxes occurred in forest edges to up to approximately 60 m from the edge in comparison with the forest interiors. In the first 10 - 20 m from the forest edge, however, the NO_3^- seepage was no straightforward reflection of N input via throughfall deposition. The difference between N input and NO_3^- output via soil nutrient leaching of about 25 kg N per ha per year was partly attributed to the higher N uptake by the trees and mainly to the enhanced N retention in the soil. In contrast with what was expected based on the higher input of potentially acidifying pollutants at edges, the rate of acidification of the upper five centimeters of the mineral soil was lower at edges than in interior zones: at edges, higher pH values and higher levels of exchangeable base cations were found in the upper mineral soil in comparison with the forest interior. However, this trend of higher pH values at the edge was not sustained in the deeper layers of the mineral soil, and at 0.10 to 0.30 m depth, pH values of the mineral soil underneath the birch and Corsican pine stands of our study were lower at the edge than in the interior. We presumed that edge gradients in microclimate, input of base cations via throughfall deposition, and biomass production together with drift of agriculturally applied lime fertilizer were the main causes for the higher N retention in the soil and lower rates of soil acidification of the upper five centimeter of the mineral soil at edges.

Based on the results this thesis, we provided edge layout and management suggestions for the purpose of mitigating edge effects on the deposition of N and potentially acidifying pollutants. We suggest, on a scale of increasing impact on the edge patterns of deposition but decreasing ease of implementation: (i) early and frequent thinning of forests, (ii) layout of gradual edge vegetation at steep edges, and (iii) conversion of coniferous pine plantations to deciduous forest types. Particularly at edges exposed to the prevailing wind directions (in Flanders, the south to westerly oriented edges), efforts should be made to reduce the extent of edge effects. It should be kept in mind, however, that mitigating measures, such as adjusted edge layout and management, do not provide the key solution for the environmental problem of air pollution. Instead, the achievement of vast emission reductions is the most fundamental prerequisite for full and sustainable protection and recovery of forest ecosystems and ecosystems in general.

Samenvatting

Wereldwijd worden bosranden de dominante component van het landschap omwille van een doorgedreven bosfragmentatie, d.i. het proces van het opsplitsen van grote, aaneengesloten bosgebieden in kleinere, meer geïsoleerde fragmenten. Ook Vlaamse bossen hebben te lijden onder sterke fragmentatie: bijna 60 % van de totale beboste oppervlakte kan beschouwd worden als bosrandzone op basis van een randzone van 50 m. Bosranden als de grens tussen bebost en open gebied, zoals heide, grasland of akker, hebben een pertinente invloed op de stroom van nutriënten en polluenten vanuit de atmosfeer naar het bosecosysteem. Deze zogenaamde randeffecten veroorzaken een tot vier keer hoger niveau van doorvaldepositie van stikstof (N) en potentieel verzurende polluenten [sulfaat (SO_4^{2-}), nitraat (NO_3^-) en ammonium (NH_4^+)] in de bosrand en reiken tot meer dan 100 m diep in het bos. In de kern (de interne zone) van bossen kunnen hoge depositieniveaus van stikstof N (NH_x + NO_y) en zwavel (S, SO_x) aanleiding geven tot stikstofverzadiging, verhoogde nitraatuitspoeling naar het grondwater, bodemverzuring, en hiermee gepaarde gaande met afname van boomvitaliteit, verschuiving in floristische samenstelling en afname van biodiversiteit. Vlaanderen heeft te kampen met de één van de hoogste depositieniveaus van verzurende en vermestende polluenten (N en S) in Europa.

Deze confronterende gegevens pleiten voor het doorvoeren van verzachtende of mitigerende maatregelen om de input van N en potentieel verzurende polluenten in bosecosystemen te beperken. Met als doel aanbevelingen te formuleren betreffende de aanleg en het beheer van bosranden om randeffecten te verzachten, trachtten we met dit onderzoek inzicht te verwerven in (i) de impact van bostype en randstructuur op de grootte van randeffecten op doorvaldepositie en (ii) de respons van het ecosysteem betreffende nitraatuitspoeling en bodemverzuring in bosranden als gevolg van de verhoogde aanvoer van N en potentieel verzurende polluenten. Om deze doelstellingen te realiseren werden in contrasterende bossen doorvaldepositie, bodemoplossing en bodemstalen verzameld en structuurkarakteristieken van het bestand opgemeten langsheen transecten uitgelegd loodrecht op de bosrand en lopend vanaf de bosrand tot 64 of 128 m in het bos. Deze studie was specifiek gericht op bossen op arme zandbodems, die van nature gevoeliger zijn voor nitraatuitspoeling en bodemverzuring, en bovendien gelegen zijn in regio's met intensieve veehouderij met hoge emissies van ammoniak. Daarenboven werd een windtunnelstudie uitgevoerd waarbij randeffecten op depositie werden gesimuleerd in bossen met verschillende structuur en samengesteld uit modelbomen op schaal.

Twee veldstudies toonden aan dat randeffecten op doorvaldepositie van stikstof en potentieel verzurende polluenten in dennenbossen dieper doordringen in het bos en aanleiding geven tot grotere depositietoenames aan de bosrand dan in eiken- en berkenbossen. In de eiken- en berkenbossen was de extra input van stikstof en potentieel verzurende polluenten als gevolg van randeffecten 10 tot 15 keer lager dan in de dennenbossen. Bovendien werd aangetoond dat het verschil tussen een dennenbestand en een nabijgelegen eikenbestand in extra input van potentieel verzurende polluenten groter was dan het verschil in input van zogenaamd basische kationen. Dit geeft aan dat de hogere extra input in dennenbosranden niet gecompenseerd wordt door een gelijkaardig hogere input van ‘potentieel neutraliserende’ basische kationen, en dat een sterkere bodemverzuring kan worden verwacht in dennenbosranden dan in eiken- en berkenbosranden.

De impact van bosrandstructuur op patronen van doorvaldepositie van N en potentieel verzurende polluenten in bosranden werd onderzocht door middel van een studie in een windtunnel en twee veldstudies. De windtunnelstudie gaf aan dat een graduele bosrand (d.i. een bosrand met een geleidelijke toename in hoogte van open ruimte naar bos) in staat is de windstroming af te leiden over de bosrand en de wind sterk te vertragen vlak voor het de bosrand bereikt. De graduele overgang zorgde zodoende voor een afname in de mate van depositieverhoging aan de bosrand in vergelijking met een scherpe overgang. Deze resultaten werden bevestigd door een veldstudie waarbij randeffecten op doorvaldepositie werden vergeleken tussen naast elkaar gelegen scherpe en graduele transities. Deze studie toonde echter aan dat een graduele transitie ook aanleiding geeft tot een afname in indringingsdiepte van de randeffecten. De extra depositie van N en S in het bos als gevolg van randeffecten aan een scherpe bosrand was drie keer groter dan de extra depositie in het bos én de gradueel opgaande vegetatie vlak voor het bos in het geval van een graduele bosrand. Bovendien bleek uit deze veldstudie dat de vorm en de grootte van de gradueel opgaande vegetatie uitermate belangrijk zijn bij het verzachten van randeffecten op depositie van stikstof en potentieel verzurende depositie.

Testen in de windtunnel toonden aan dat randeffecten dieper doordringen in ijler bos dan in dichter bos, wat bleek gekoppeld te zijn aan een sterkere afname van windsnelheid en turbulentie in dichtere bossen. Dit effect van kroondensiteit, uitgedrukt als relatieve bladoppervlakte (*leaf area index*, LAI) werd geverifieerd in een veldstudie op homogene bestanden van Corsicaanse of grove den met verschillende LAI. Blijkbaar speelt LAI een sleutelrol in de processen die randeffecten veroorzaken omdat het zowel de mate van depositietoename aan de bosrand als de indringingsdiepte van randeffecten stuurt. De algemene toename van depositie in de ganse randzone van bossen vertoonde een optimum bij gematigde LAI-waarden. In bossen met lage LAI bleek de mate van depositietoename klein te zijn, terwijl in bossen met hoge LAI de randeffecten zeer beperkt doordringen in het bos.

Zowel in loof- als in naaldbossen werden hogere niveaus van nitraatuitspoeling waargenomen vanaf de bosrand tot op ongeveer 60 m van de rand. Echter, in de eerste 10 tot 20 m van de rand was de nitraatuitspoeling geen directe weerspiegeling van de aanvoer van stikstof via doorvaldepositie. Het verschil tussen stikstofinput via doorvaldepositie en de stikstofoutput via uitspoeling bedroeg 25 kg N per ha per jaar en kan worden toegeschreven aan de hogere opname van stikstof door de bomen en, hoofdzakelijk, de hogere retentie van stikstof in de bodem. De hogere aanvoer van potentieel verzurende polluenten aan de bosrand werd niet gereflecteerd in een snellere verzuring van de bovenste 5 cm van de minerale bodem: aan de bosranden werden hogere pH-waarden en hogere concentraties aan uitwisselbare kationen vastgesteld in de bovenste laag van de minerale bodem dan in de boskernen. Echter, in de diepere bodemlagen zette deze trend zich niet verder, en op een diepte van 0.10-0.30 m vertoonde de minerale bodem onder de onderzochte bestanden van berk en Coriscaanse den lagere pH-waarden aan de bosrand dan in de boskern. We veronderstellen dat randgeïnduceerde gradiënten van microklimaat, aanvoer van basische kationen via doorvaldepositie en biomassaproductie samen met de inwaai van landbouwkundige kalkstoffen de belangrijkste oorzaken zijn van de hogere retentie van stikstof in de bodem en de tragere bodemverzuring in de bovenste vijf cm van de minerale bodem aan de bosranden.

Op basis van de resultaten uit deze studie verstrekten we suggesties voor randaanleg en randbeheer in functie van de bescherming van bosranden tegen randeffecten op depositie van stikstof en potentieel verzurende polluenten. Met toenemende impact op randeffecten maar afnemende graad van snelle realiseerbaarheid, stellen we de volgende maatregelen voor: (i) het vroegtijdig en frequent dunnen van bossen, (ii) de aanleg van graduele vegetatie (mantel- en zoomvegetatie) aan scherpe randen en (iii) de omvorming van naaldboomaanplantingen naar loofbos. Voornamelijk in bosranden blootgesteld aan de overheersende windrichting - in Vlaanderen zijn dit de zuid- tot westgeoriënteerde randen - kunnen mitigerende maatregelen een grote rol kunnen spelen om de druk van atmosferische pollutie op bosecosystemen enigermate te verlichten. Er dient echter op gewezen dat mitigerende maatregelen, zoals het hier voorgestelde aangepaste design en beheer van bosranden, niet de sleutel bieden voor de oplossing van de milieuproblematiek rond luchtvervuiling. De meest fundamentele voorwaarde om een volledige en duurzame bescherming en herstel van bosecosystemen, en ecosystemen in het algemeen, te kunnen bewerkstelligen, is het realiseren van aanzienlijke emissiereducties.

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Curriculum vitae

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Scientific activities

Participation in congresses, symposia, and workshops

Participation with presentation

11-13 March 2008. COST Strategic Workshop 'Forest ecosystems in a changing environment: identifying future monitoring and research needs', Istanbul, Turkey. Poster presentation: Wuyts, K., De Schrijver, A., Verheyen, K., Incorporating forest edge deposition in evaluating excess of critical loads.

20-21 February 2008. NitroEurope IP Open Science Conference 'Reactive nitrogen and the European greenhouse gas balance', Ghent, Belgium. Poster presentation: Wuyts, K., Verheyen, K., Conversion of forest type influences edge patterns of throughfall deposition.

17 October 2007. 13th PhD symposium on Applied Biological Sciences, Leuven, Belgium. Poster presentation: Wuyts, K., Cornelis, W.M., Gabriels, D., Verheyen, K., Impact of a gradual forest edge on patterns of wind speed, turbulence and deposition: a wind tunnel study.

22 March 2007. Inverde symposium 'Starters in het bosonderzoek', Brussels, Belgium. Oral presentation: Wuyts, K., Impact van bestandsoort op ruimtelijke patronen van atmosferische depositie, uitspoeling en bodemverzuring in de bosrand.

23 November 2006. ANB topical day 'Van dennenplantages naar een beloofd land?! Theoretische en praktische aspecten van bosomvorming', Hasselt, Belgium. Oral presentation: Wuyts, K., Gielis, L., De Schrijver, A., Geudens, G., Staelens, J., Vanhellemont, M., Bosomvorming als mitigerende maatregel tegen verzuring en vermesting.

Participation without presentation

7 - 8 February 2008. NecoV Winter symposium 2008 'Timeless Ecology: From seconds to centuries', Antwerp, Belgium.

17 October 2006. SCK-CEN topical day 'Biogeochemical response of forest vegetation to chronic pollution: processes, dynamics and modelling', Mol, Belgium.

21 - 23 May 2006. ConForest-IEFC conference 'Plantation or conversion - The debate!', Freiburg-im-Breisgau, Germany.

2 - 4 May 2006. 1st Meeting for Bilateral Project Universidad Austral de Chile (Valdivia) - Ghent University (Flanders) 'Ecohydrological monitoring and modelling in managed and unmanaged native forest ecosystems in Southern Chile', Universidad Austral de Chile, Valdivia, Chile.

Supervision of M.Sc. thesis students

2006-2007	Frederic Vermeiren. De impact van graduele randen op doorvaldepositie in bosranden. Promotor: Prof. dr. ir. K. Verheyen