Biogeochemistry (2010) 101:7–26
DOI 10.1007/s10533-010-9459-9

Ion concentrations and fluxes of seepage water before and after clear cutting of Norway spruce stands at Ballyhooly, Ireland, and Höglwald, Germany

Christian Huber · Julian Aherne · Wendelin Weis · Edward P. Farrell · Axel Göttlein · Thomas Cummins

Received: 29 October 2009 / Accepted: 22 April 2010 / Published online: 13 May 2010

Abstract Ion concentrations and fluxes in seepage water (below the main rooting zone) were compared before and after clear cutting at two similar long-term experimental Norway spruce forest plots. While Ballyhooly (Ireland) was influenced by sea salt deposition, Höglwald (Germany) received high nitrogen (N) deposition. These differences were reflected in seepage water concentrations with sodium (Na^+) and chloride (Cl^-) dominating at Ballyhooly and high nitrate (NO_3^-) and aluminium concentrations at Höglwald. Following clear cutting of the forest plots, NO_3^- concentrations peaked (Ballyhooly: 2018 $\mu mol_c\ L^{-1}$, Höglwald: 2595 $\mu mol_c\ L^{-1}$). Moreover, at Ballyhooly, NO_3^- concentrations and fluxes were continuously elevated for ~1.5 years. At Höglwald, the clear cut plot, which was replanted with spruce and beech saplings, exhibited periodically elevated NO_3^- concentrations with two distinct peaks. However, low concentrations, compared to the control (uncut) plot, were also observed. Further, at Höglwald a plot with a pre-existing dense natural regeneration of Norway spruce exhibited much lower NO_3^- concentrations before and after clear cutting. Nonetheless, NO_3^- concentrations following clear cut at both sites were elevated at least periodically above European drinking water standards (50 mg L^{-1}). An important prerequisite for NO_3^- leaching is that forests are N saturated or at least not N-limited; consequently chronic elevated N deposition may lead to increased deterioration of seepage water quality across Europe following forest disturbances (harvesting, windthrow, insect attacks). Clear cutting at Ballyhooly was responsible for significant element loss, especially of potassium, N and calcium, while magnesium loss was compensated by high sea salt inputs. At Höglwald the contamination of seepage water with NO_3^- has been the main problem for more than 20 years at the mature stand. A pre-existing

---

Collaborator under a German Academic Exchange Service (DAAD) Short-Term Lectureship.

C. Huber (✉) · W. Weis · A. Göttlein
Fachgebiet für Waldernährung und Wasserhaushalt,
Department für Ökologie und Ökosystemmanagement,
WZW, TU München, Am Hochanger 13, 85354 Freising,
Germany
e-mail: christian.huber@tum.de

W. Weis
e-mail: weisw@forst.tu-muenchen.de

A. Göttlein
e-mail: goettlein@boku.forst.uni-muenchen.de

C. Huber · J. Aherne
Environmental and Resource Studies, Trent University,
Peterborough, ON K9J 7B8, Canada
e-mail: jaherne@trentu.ca

E. P. Farrell · T. Cummins
UCD School of Agriculture, Food Science and Veterinary
Medicine, University College Dublin, Belfield,
Dublin 4, Ireland
e-mail: ted.farrell@ucd.ie

T. Cummins
e-mail: thomas.cummins@ucd.ie

regeneration can help to reduce NO_3^- and cation leaching after cutting.

Keywords Aluminium · Calcium · Clear cutting · Harvesting · Nitrogen saturation · Magnesium · Nitrate · Nutrient leaching · Potassium · Sea salt · Water quality

Introduction

The atmospheric deposition of sulphur (S) and nitrogen (N) compounds has greatly influenced the ion budgets of European forests. Despite significant reductions in anthropogenic emissions of S in Europe (Stern 2006), many sites are still affected owing to delayed recovery from acidification (Alewell et al. 2000). In concert, N emissions are still high; food and energy production have dramatically increased N availability since the 1960s (Galloway et al. 2003, 2004) causing elevated emissions of NOx and NHy. Moreover, the deposition of N is often found to be highest in agricultural areas owing to high livestock densities (Göttlein and Kreutzer 1991; Rothe et al. 2002; Gundersen et al. 2006). There is increasing concern that forests will loose their potential to store N and become ‘N saturated’; saturation is most widely assumed to occur when the availability of mineral N exceeds the nutritional demands of microbes and plants (Aber et al. 1989), which is ultimately reflected in elevated NO_3^- leaching (Gundersen et al. 2006). Nonetheless, seepage water from forests is generally of good quality and (forest) ground water resources are widely used for drinking water.

In addition to deposition processes, forest management operations, such as clear cutting, are known to have a negative effect on the quality of seepage waters by increasing nitrate (NO_3^-) concentrations (Vitousek et al. 1979). Elevated soil water concentrations of NO_3^- are primarily due to mineralization of litter and harvesting residues followed by nitrification, coupled with reduced nutrient uptake following harvest (Burns and Murdoch 2005). Nitrate not only impairs seepage water quality, but nitrification is a significant contributor to soil acidification, causing reduced acid-neutralising capacity, potential decrease in soil pH, increases in aluminium (Al^{3+}) mobilisation and losses of base cations (calcium (Ca^{2+}), magnesium (Mg^{2+}) and potassium (K^+)). Ultimately, forest ecosystem research and monitoring provide the basis for assessing the impacts of forest management operations on chemical quality of seepage water (Farrell et al. 2000).

The objective of this paper was to evaluate seepage water quality and fluxes before and after clear cutting at two long-term experimental Norway spruce forests at Ballyhooly, Ireland, and Höglwald, Bavaria, Germany. The study sites had comparable soil and stand properties, experimental methods (monitoring), and comparable stand history; however, the atmospheric deposition of inorganic ions differed between sites. Ballyhooly is located in an area of low industry, grass based agriculture, and a significant sea salt influence. The natural deposition of sea salts is significant in maritime regions (Farrell 1995); specifically sodium (Na^+) and chloride (Cl^-) are the dominant ions, but significant amounts of sulphate (SO_4^{2-}) and Mg^{2+} are also deposited (Kreutzer et al. 1998). Sea salts can induce short-term acidification in surface waters as a result of ion exchange reactions and the replacement of Al^{3+} on ion exchange complexes following episodic high inputs (Farrell 1995, Farrell et al. 1998; Kreutzer et al. 1998; Beier et al. 2003). In contrast, the marine influence is small at the Höglwald, but N deposition is high, and the site is considered to be N saturated (Rothe et al. 2002).

Materials and methods

Site descriptions

The Höglwald (HW) was established in 1982 as a long-term ecological monitoring and experimentation site (Kreutzer and Weiss 1998). The intensive forest monitoring plot at Ballyhooly (BH) was established in 1989 as the first Irish plot in the European Community’s programme on the ‘Protection of the Community’s Forests against Atmospheric Pollution’ (Council Regulation 3528/86: EEC 1986); notably ‘similarity to HW’ was the principal site selection criteria. Further, both study sites were part of the EXMAN project (Farrell et al. 1994; Cummins et al. 1995; Kreutzer et al. 1998).

Site characteristics for both plots have been previously described (Table 1: Farrell et al. 1993, 1994; Kreutzer and Weiss 1998; Huber et al. 2004a). The BH plot was located within Casleblagh Wood, a small forest of about 75 ha near Ballyhooly, County

Table 1 Geographical location and general site characteristics for intensive forest monitoring plots (*Picea abies* (L.) Karst.) at Ballyhooly, Ireland and Höglwald, Germany

Site characteristics	Ballyhooly	Höglwald
Latitude (decimal degrees)	52.136	48.288
Longitude (decimal degrees)	−8.405	11.075
Elevation (m asl)	69	540
Precipitation (mm y^{-1})	1167 (1989–1999)	947 (1985–2004)
Annual temperature (°C)	9.5	7.9
January temperature (°C)	6.0	−1.5
July temperature (°C)	16.0	17.0
Year of planting	1939	1910–1911
Number of trees (ha^{-1} in 1989)	632	603
Basal area (m^2 ha^{-1})	44	71
Mean increment (m^3 ha^{-1} y^{-1} in 1989)	21	23
Mean upper height (m)	24	37
Mean diameter (at 1.3 m)	29	36
Soil taxonomy		
WRB	Albic podzol and dystric cambisol	Dystric cambisol
USDA	Typic haploorthod	Typic hapludalf
Central European	Podsol to parabrownearth	Parabrownearth to brownearth
Parent material	Sandstone till, colluvium	Loess above tertiary silty sand deposits
Texture	Sandy loam	Silty loam to sandy loam

Cork (35 km north of the city of Cork) at the foot of the Nagle Mountains, 69 m above sea level in an area of relatively prosperous dairy farming mixed with less intensive hill farms. The relatively small forest district at the Höglwald (370 ha) is situated on a flat hilltop at 540 m above sea level about 70 km north of the Alps and 50 km west of Munich. Both forests belong to the zone of temperate climate, with higher annual precipitation and mean annual temperature at BH, and higher mean summer temperatures at HW (Table 1). The plots supported healthy, well growing mature pure Norway spruce plantations (*Picea abies* (L.) KARST.) exhibiting no symptoms of defoliation (Farrell et al. 1996; Huber et al. 2004a). The mean yearly increment of the trees was similar (Table 1), and foliar analysis indicated good nutrition of the major elements (Table 2; Huber et al. 2006a). Litterfall fluxes at HW were higher for most of the elements due to higher litterfall mass (Table 2), while the concentrations of P, Mg and Na were higher at BH (see Table 2). Soil properties were similar at the two plots (Tables 1 and 3); mineral soil textures were loamy, covered by surface humus (moder). The degree of podsolization was very weak at HW and (in contrast) rather strong at BH. The base saturation in the B horizon at HW increased with depth, while the base saturation at BH was relatively low in deeper soil horizons (Table 3). The shrub layer and the herb layer were more pronounced at BH, while at HW mosses dominated the ground vegetation (Table 4).

Table 2 Mean elemental concentration in current year needles for Ballyhooly (1988–1995) and Höglwald (1982–2004), and litterfall concentrations and fluxes for Ballyhooly (1989–1994) and Höglwald (1984–2004)

	Mass	N	P	K	Ca	Mg	Mn	Na
Needle concentrations (mg g^{-1})								
Ballyhooly		16.0	1.7	6.1	3.6	1.4	1.2	1.0
Höglwald		15.1	1.6	4.0	3.5	1.1	2.1	1.0
Litterfall concentrations (mg g^{-1})								
Ballyhooly		10.5	1.1	1.5	4.1	1.1	1.5	1.5
Höglwald		11.6	0.8	1.8	5.8	0.6	2.1	0.0
Litterfall fluxes (kg ha^{-1} y^{-1})								
Ballyhooly	4660	49	5	7	19	5	7	7
Höglwald	6224	72	5	11	36	4	13	0

Table 3 Chemical analysis (organic carbon, nitrogen, exchangeable cations, cation exchange capacity and base saturation) of soils by horizon at Ballyhooly and Höglwald

				Exchangeable cations									
Layer	Depth (cm)	C (%)	N (%)	H^+ (mmol$_c$ g^{-1})	Al^{3+} (mmol$_c$ g^{-1})	Mn^{2+} (mmol$_c$ g^{-1})	Fe^{3+} (mmol$_c$ g^{-1})	Ca^{2+} (mmol$_c$ g^{-1})	Mg^{2+} (mmol$_c$ g^{-1})	K^+ (mmol$_c$ g^{-1})	Na^+ (mmol$_c$ g^{-1})	CEC (mmol$_c$ g^{-1})	BS (%)
Ballyhooly													
L	−10 to −8	38.5	1.74	n.d.	4.0	33.1	0.3	137.4	121.1	114.0	17.1	427.0	91
Of	−8 to −2	38.5	1.20	n.d.	15.6	10.2	3.3	65.9	41.6	19.3	7.7	163.6	82
Oh	−2 to 0	37.4	1.21	n.d.	14.3	3.1	1.7	68.4	59.3	8.3	7.7	162.8	88
Ahe	0–5	4.4	0.11	n.d.	16.7	0.1	1.1	2.7	3.3	1.2	1.3	26.4	32
E1	5–10	3.5	0.13	n.d.	61.3	0.3	4.9	1.9	1.9	1.0	1.5	72.7	9
E2	10–20	1.6	0.06	n.d.	42.5	0.3	0.6	1.0	0.6	0.8	1.1	46.9	7
Bs	20–40	0.7	0.05	n.d.	28.0	0.2	0.1	0.9	0.4	0.7	0.9	31.2	9
B/C	40–60	1.2	0.05	n.d.	28.2	0.1	0.2	1.0	0.4	0.8	0.9	31.6	10
Höglwald													
LOf1	−5 to −2	47.4	1.74	9.9	2.4	44.1	0.4	137.4	24.5	14.4	1.2	233.4	76
Of2	−2 to −1	46.3	1.82	40.7	15.2	20.4	2.7	103.3	16.3	6.2	1.0	204.9	62
Oh	−1 to 0	39.4	1.65	37.8	59.5	9.5	6.1	62.2	11.9	4.2	0.9	191.2	41
Aeh	0–5	5.5	0.28	10.0	67.1	1.3	5.1	5.1	2.5	1.1	0.4	92.3	10
Ahl	5–10	2.4	0.13	4.4	58.3	1.6	1.7	1.7	1.2	0.7	0.3	69.6	6
	10–20	1.2	0.08	2.0	42.1	1.3	0.3	1.0	0.7	0.6	0.3	48.0	5
Al	20–30	0.6	0.05	1.3	39.5	1.2	0.2	0.9	0.6	0.8	0.3	44.5	6
	30–40	0.3	0.04	1.6	53.9	1.2	0.1	1.1	1.0	1.3	0.3	60.1	6
Bt	40–60	0.2	0.02	2.5	74.5	2.2	0.0	9.1	7.8	2.1	0.7	98.5	20
Bv	60–80	0.1	0.02	2.7	57.6	1.9	0.0	26.8	22.5	2.4	0.8	114.4	46

n.d. not determined

Table 4 Coverage of plant species in the tree layer, shrub layer and moss layer of the stands at Ballyhooly and Höglwald before felling

Stratum and species	Vegetation coverage			
	Ballyhooly	Höglwald		
	CCR	UCC	CCR	CCN
Tree layer (>5 m)				
Picea abies (L.) KARST.	5	5	5	5
Shrub layer (0.5–5 m)				
Sorbus aucuparia L.	+			
Castanea sativa MILL.	+			
Dryopteris dilatata (VILL.) FUCHS	3			
Ilex aquifolium L.	2			
Pteridium aquilinum (L.) KUHN	+			
Quercus spp.	+			+
Tsuga heterophylla RAFIN.	+			
Picea abies (L.) KARST.				2
Fagus sylvatica (L.)				1
Herb layer (0.05–0.5 m)				
Blechnum spicant (L.) ROTH	+			
Calluna vulgaris (L.) HULL.	+			
Hedera helix L.	3			
Lonicera periclymenum L.	2			
Luzula sylvatica (HUDSON) GAUDIN.	4			
Oxalis acetosella L.	3	2	2	2
Rubus fruticosus agg. L.	2	2		+
Vaccinium myrtillus L.	+			
Viola selkirkii (PURSH ex GOLDIE)	1			
Dryopteris carthusiana (VILL.) FUCHS		+	+	+
Deschampsia cespitosa (L.)				1
Deschampsia flexuosa (L.)				+
Picea abies (L.) KARST.		+	+	+
Fagus sylvatica (L.)		+	+	+
Moss layer (<0.05 m)				
Thuidium tamariscinum HEDW.	+	5	5	4
Eurhynchium striatum HEDW. SCHIMP.		+		
Hypnum cupressiforme HEDW.		+		
Mnium affine BLAND.		+		
Polytrichum attenuatum MENZ.		+		2
Lophocolea heterophylla SCHRAD.		+		

Braun–Blanquet rating symbols indicate the coverage (5: >75% coverage: 4: 50–75% coverage: 3: 25–50% coverage, 2: 5–25% coverage, 1: 5% coverage, numerous individuals, +: <5% coverage, few individuals). Plots: clear cut replant (*CCR*), clear cut and natural regeneration (*CCN*) and uncut control (*UCC*)

Ballyhooly experienced lower deposition of NH_4^+ and NO_3^- (Table 5); furthermore, the marine influence was clearly visible, with higher amounts of Na^+, Cl^- and Mg^{2+} in throughfall at BH (Table 5). At HW, deposition of NO_3^- and especially NH_4^+ was (continuously) high, whereas SO_4^{2-} deposition decreased owing to emissions reductions.

Table 5 Ion deposition in bulk precipitation and throughfall at Ballyhooly (BH) and Höglwald (HW)

Site	Period	Rainfall (mm y^{-1})	Ca^{2+} (mmol$_c$ m^{-2} y^{-1})	Mg^{2+} (mmol$_c$ m^{-2} y^{-1})	K$^+$ (mmol$_c$ m^{-2} y^{-1})	Na$^+$ (mmol$_c$ m^{-2} y^{-1})	H$^+$ (mmol$_c$ m^{-2} y^{-1})	NH$_4^+$ (mmol$_c$ m^{-2} y^{-1})	NO$_3^-$ (mmol$_c$ m^{-2} y^{-1})	SO$_4^{2-}$ (mmol$_c$ m^{-2} y^{-1})	Cl$^-$ (mmol$_c$ m^{-2} y^{-1})
Bulk precipitation											
BH	1989–1994	1141	17	32	6	191	13	31	13	44	195
BH	1995–1999	1206	16	28	5	141	11	37	14	45	194
HW	1989–1994	854	19	6	6	6	10	46	34	38	11
HW	1995–1999	980	21	5	6	9	7	46	36	33	17
HW	2000–2003	1074	20	4	8	12	11	51	40	31	14
Throughfall											
BH	1989–1994	632	37	69	57	296	3	54	21	87	372
HW	1989–1994	470	37	15	40	9	3	130	63	98	24
HW	1995–1999	529	36	15	44	13	1	110	66	70	26
HW	2000–2003	605	34	16	50	13	5	109	71	44	24

Experimental setup, sampling and chemical analyses: Ballyhooly

Intensive monitoring at BH commenced in 1989 and continued until 1994. The plot was clear cut during October 1995 (manually by chainsaw); stems including bark were removed and slash left on site. Monitoring was re-established between 1996 and 1999. In September 1996 the plot was disc trenched and replanted with Oak *(Quercus petraea* (MATTUSCHKA) LIEBL.) and Douglas fir (*Pseudotsuga menziesii* (MIRBEL) FRANCO). Herbicide (glyphosphate) was sprayed in September 1998 to reduce ground vegetation. Bulk precipitation was sampled with three polypropylene collectors outside the forest stand, while throughfall was sampled weekly with nine collectors. Seepage water was collected with eight suction cups installed vertically in hand augered holes at 70 cm depth (from the top of the mineral soil). The samples were taken at intervals ranging from 2 to 4 weeks. A portable 12 V vacuum pump was used to partially evacuate glass bottles attached to the samplers (ca. 400 hPa 1 week prior to sample collection). Samples from individual cups were pooled, in proportion to their volume, filtered (Whatman No. 41) and stored in the dark at 4°C prior to analysis.

The pH was measured on the bulked unfiltered sample using a low-conductivity combination electrode. Ammonium (NH_4^+) was measured colorimetrically using the phenol-hypoclorite method with UV detection at 635 nm (Solórzano 1967). Calcium, Mg^{2+}, K^+, Na^+, manganese (Mn^{2+}), iron (Fe^{3+}) and Al^{3+} were measured using an inductively coupled plasma optical emission spectrophotometer (ICP-OES: Varian). Chloride, NO_3^- and SO_4^{2-} were analysed by ion chromatography (IC: Shimadzu). Meteorological data were taken from the nearby station at Moorepark, Fermoy.

Soil water fluxes were estimated using the computer simulation package FORHYD (FORest HYDrology: Tiktak and Bouten 1992) as described by Aherne et al. (1999). FORHYD contains several modules simulating the hydrological processes in forested ecosystems, SWIF (Soil Water In Forests) describes root-water uptake and vertical water flow in the unsaturated soil zone as well as lateral drainage in the saturated zone using the Richards equation (Richards 1931). The energy available for forest floor evaporation and root-water uptake is calculated as the Makkink reference evapotranspiration (de Bruin 1987) based simply on air temperature and global solar radiation. The global solar radiation was estimated using an Ångström-type relationship solved for Ireland (Connaughton 1967). The available energy is partitioned between potential transpiration and soil evaporation, using a canopy gap fraction. A crop factor is used to calculate the root water uptake from the potential transpiration. Evaluation of the model output was carried out by comparing simulated and measured weekly soil water tensions (1989–1994) using Thies Clima type tubes with an unglazed ceramic cup. Further details are given by Aherne et al. (1999).

Experimental setup, sampling and chemical analyses: Höglwald

The uncut control plot at HW has been intensively monitored since 1984. In 1999 a separate experimental plot, with uniform stand and site characteristics, was established for clear cut treatments. The experimental plot was situated approximately 400 m southeast of the uncut control plot, separated by game fences and small utility roads. The pre-treatment monitoring period lasted from June 1999 to February 2000 and felling was performed with a harvester at the end of February 2000. Stems including bark were removed and slash remained on site. The small scale clear-cut (1.0 ha) was divided into two treatments: (a) replant with five year old Norway spruce and European beech saplings in April 2000, and (b) natural spruce regeneration. Vegetation cover (prior to harvest) was similar on the uncut control and the clear cut replant plots, with mosses dominating the herb layer and the absence of a shrub layer; however, the natural spruce regeneration had a shrub layer dominated by spruce and a higher coverage of herbs (Table 4). Bulk deposition was sampled with three open polyethylene collectors on a clear cut about 1 km away from the study sites and throughfall was sampled with ten open collectors at each plot. Seepage water was collected with five to ten ceramic cup lysimeters on each plot. Suction cups were installed vertically, in hand augered holes at 40 cm depth (from the top of the mineral soil). The samples were taken at intervals ranging from 10 to 30 days. A portable 12 V vacuum pump was used to partially evacuate glass bottles attached to the samplers (ca. 600 hPa 1 week prior to sample collection).

The pH was measured with an Ingold glass-electrode on unfiltered samples. Subsequently, water samples were filtered using membrane filters with a pore size of 0.45 µm (Schleicher and Schuell, NC45) and stored at 4°C until they were analysed. Aluminium, Mn^{2+}, Fe^{3+}, Ca^{2+}, Mg^{2+}, Na^{+}, and K^{+} were analysed with an ICP-OES (Perkin Elmer, Optima 3000). Chloride, NO_3^- and SO_4^{2-} were determined by IC (Dionex, IC2020I). Meteorological data were obtained from a nearby station in the German weather service. Supplementary climatic measurements were taken on each plot at hourly intervals for precipitation, air temperature, relative humidity and photoactive radiation.

The water fluxes were either measured directly (precipitation, throughfall) or calculated using a mechanistic model as described in Huber et al. (2004b) and Weis et al. (2006). The model can include up to three different vegetation layers (mature stand, regeneration, ground vegetation). Hourly values of bulk precipitation, air temperature, relative humidity and wind speed were needed as input data. Soil water fluxes for the organic layer and the soil depths 20, 40, 60, 80, 100 and 120 cm were calculated using the Darcy Richards equation (Richards 1931). For every soil layer, and each of the three ground covers, the relationships between soil matric potential, soil water content and soil hydraulic conductivity were determined for undisturbed soil cores (diameter 15 cm, height 20 cm) with one TDR probe (Trime P2, IMKO-GmbH, Ettlingen, Germany) and two tensiometers (miniature pressure-transducer tensiometer T5, UMS GmbH, Munich, Germany) installed horizontally in the core. Modelled matric potential and water content were compared to measured values. Ion fluxes were calculated by multiplying the mean concentrations of every sampling date with the daily water fluxes for the period prior to sampling.

Results

Ion concentrations before and after clear cutting

Seepage water NO_3^- concentrations at BH (pre-clear cut 1989–1994) were relatively low with a median of

23 μmol$_c$ L^{-1}, a 75th percentile of 101 μmol$_c$ L^{-1} and a maximum of 385 μmol$_c$ L^{-1} (Fig. 1). None of the observations were above the European Community level for drinking water (50 mg NO_3^- L^{-1} [806 μmol$_c$ NO_3^- L^{-1}]; EC 1998). In contrast, NO_3^- concentrations at HW (uncut control) were predominantly above the level for drinking water (1989–1999). The lowest concentration at HW was 390 μmol$_c$ L^{-1}, the median was 1279 (1989–1994) and 934 (1995–1999) μmol$_c$ L^{-1} and the maximum was 2226 μmol$_c$ L^{-1}. Following clear cut, elevated concentrations were observed at BH until the end of 1997 (2 years after clear cutting) with a maximum of 2018 μmol$_c$ L^{-1}. Further, a smaller peak was observed during spring 1999 (following herbicide application). At HW (clear cut), the highest concentrations (2595 μmol$_c$ L^{-1}) were observed in the year of felling (September to October 2000). Further, in 2001 a second lower peak was observed (Figs. 1 and 2). The clear cut plot with natural spruce regeneration showed lower concentrations pre-clear cut and a smaller but pronounced peak in 2000 following clear cut. However, from early 2002 onwards NO_3^- at HW was lower than pre-harvest concentrations at both clear cut plots. Nitrate concentration at the uncut control plot did not show similar peaks and was within the range observed during previous periods (1989–1999). Nitrate concentration for the individual suction cups at HW was highly variable; the maximum for a single suction cup was 2743 μmol$_c$ L^{-1} at the uncut control, 3675 μmol$_c$ L^{-1} at the clear cut replant and 1846 μmol$_c$ L^{-1} at the clear cut with natural spruce regeneration variable (Fig. 2).

Median SO_4^{2-} concentrations were higher at HW (991 μmol$_c$ L^{-1}) than at BH (284 μmol$_c$ L^{-1}) during the period 1989–1994 (Fig. 3). Maximum concentrations at HW decreased in the following years (1995–1999), while the median remained similar. Following clear cut, SO_4^{2-} concentrations at BH remained

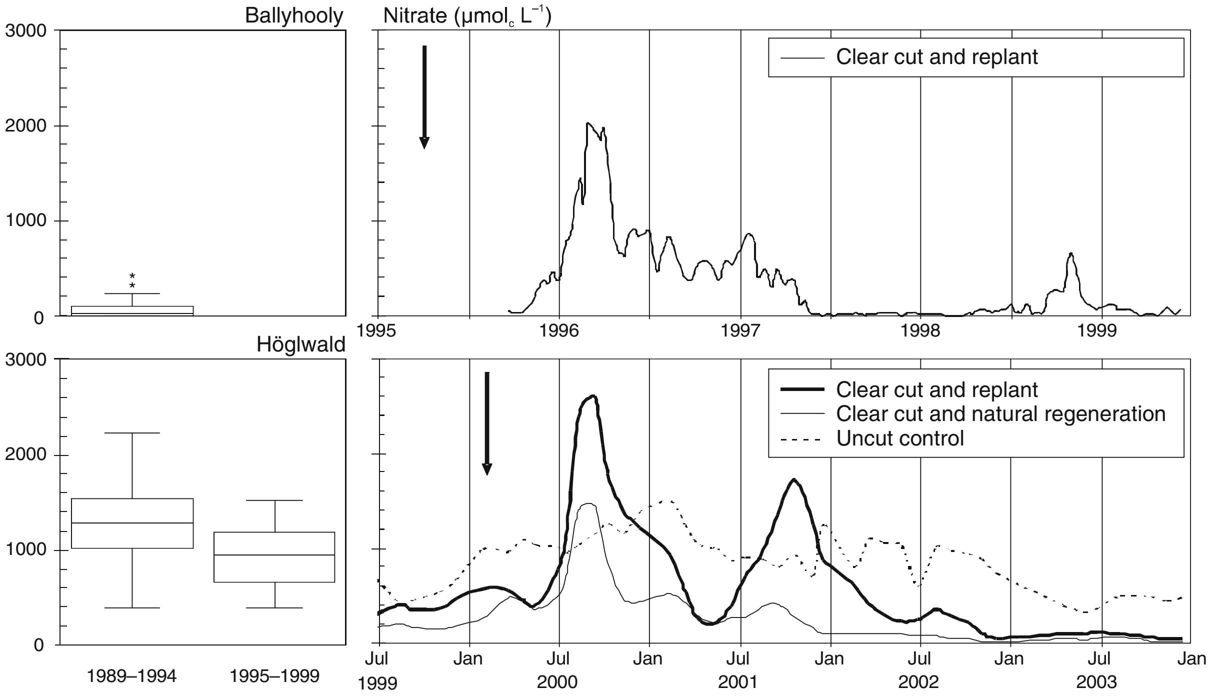

Fig. 1 Nitrate (NO_3^-) concentrations in soil solution below the main rooting zone at Ballyhooly (BH: 75 cm depth) and Höglwald (HW: 40 cm depth) before (box plot summaries: *left*) and after clear cutting (time-series of mean concentration: *right*). The box plot summaries represent the minimum, lower quartile, median, upper quartile and maximum observations for the periods 1989–1994 (BH and HW) and 1995–1999 (HW). Outliers are indicated by an *asterisk*. Clear cutting was carried out during October 1995 at BH and during February 2000 at HW (as indicated by *vertical arrows*). Soil water monitoring at BH was stopped prior to clear cut and re-commenced shortly after (February 1996). In addition to the clear cut and replant, uncut control and clear cut and natural regeneration plots were also monitored at HW

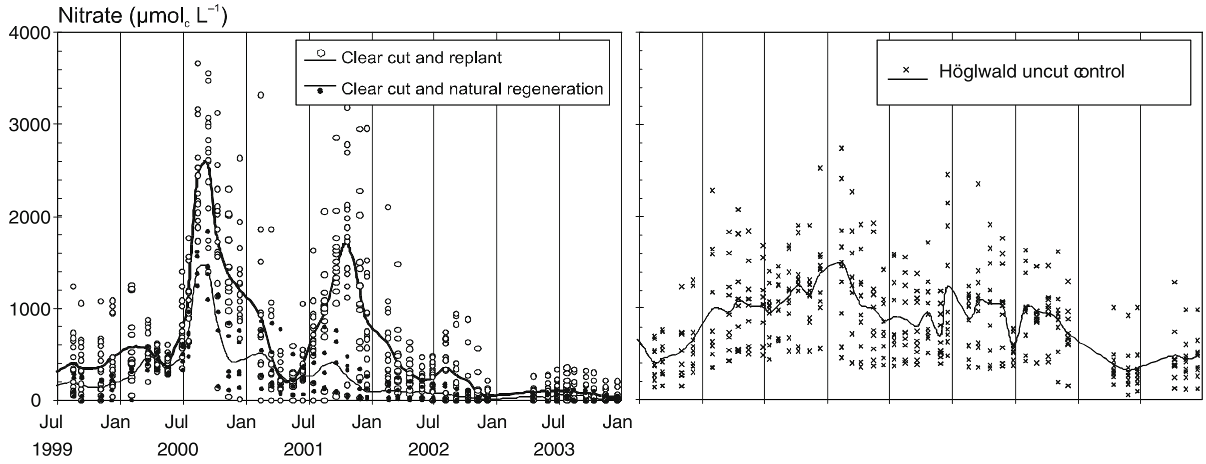

Fig. 2 Nitrate (NO_3^-) soil water concentrations at 40 cm depth for the Höglwald on the clear cut plots (replant and natural regeneration: *left*) and uncut control plot (*right*). Each point represents an observation from a single suction cup. The *lines* indicate the mean concentrations

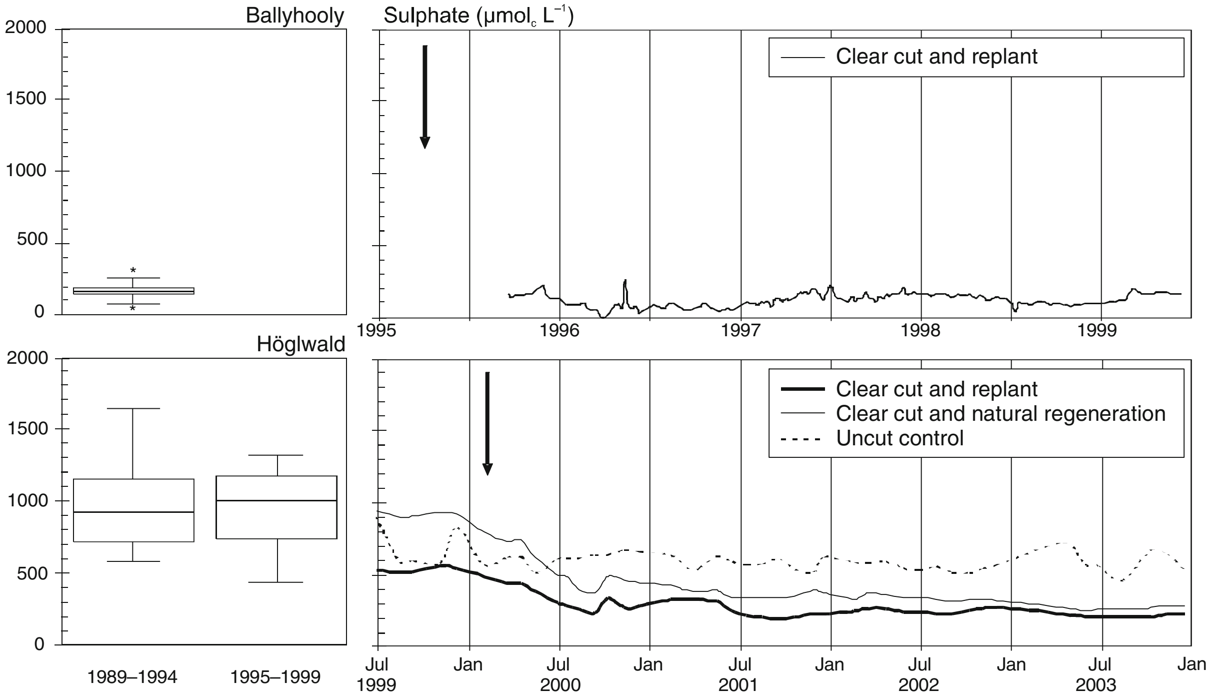

Fig. 3 Sulphate (SO_4^{2-}) concentrations below the main rooting zone at Ballyhooly (75 cm depth) and Höglwald (40 cm depth) before (box plot summaries: *left*) and after clear cutting (time-series of mean concentrations: *right*). See Fig. 1 for further details

relatively unchanged (Fig. 3). Lower concentrations of SO_4^{2-} were observed at the HW uncut control during the clear cut period (compared to previous periods) owing to reductions in S emissions. The decrease at the clear cut plots was more pronounced owing to loss of canopy scavenging and increased water fluxes. No distinct peak in SO_4^{2-} concentrations were observed at either study site.

Chloride concentrations in seepage water were much higher at BH; moreover, Cl^- was the dominant anion in seepage water pre-clear cut albeit with high variability (Fig. 4). Following clear cut,

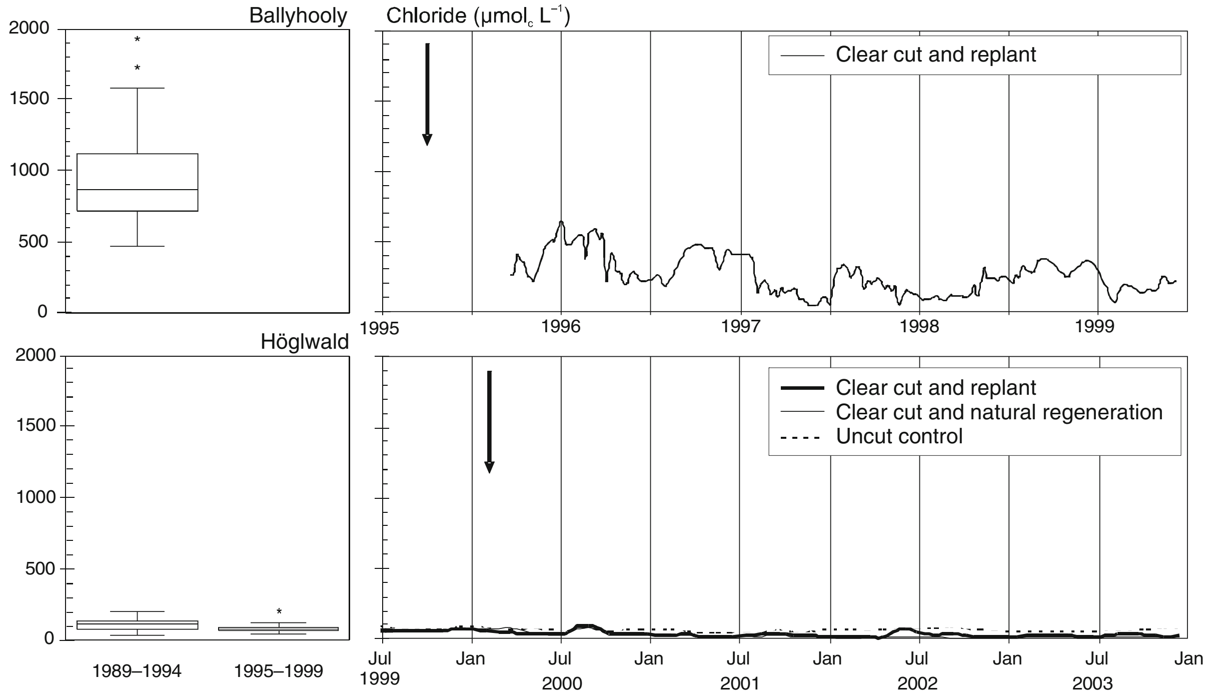

Fig. 4 Chloride (Cl^-) concentrations below the main rooting zone at Ballyhooly (75 cm depth) and Höglwald (40 cm depth) before (box plot summaries: *left*) and after clear cutting (time-series of mean concentrations: *right*). See Fig. 1 for further details

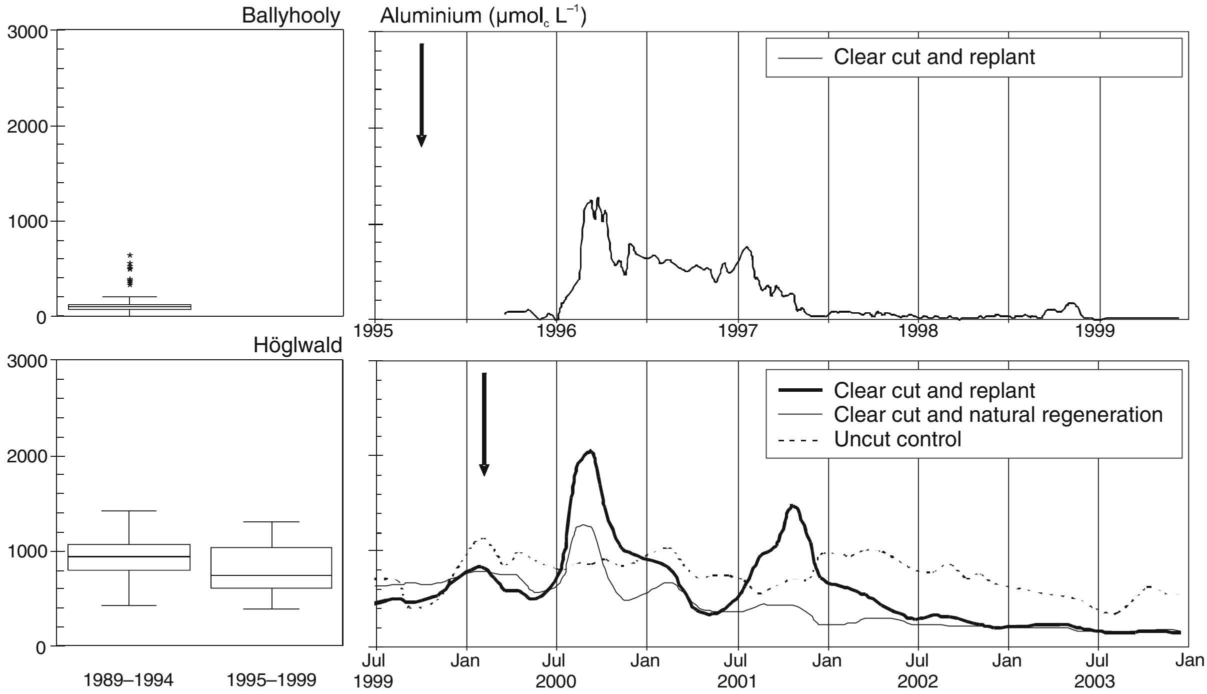

Fig. 5 Aluminium (Al^{3+}) concentrations below the main rooting zone at Ballyhooly (75 cm depth) and Höglwald (40 cm depth) before (box plot summaries: *left*) and after clear cutting (time-series of mean concentrations: *right*). See Fig. 1 for further details

concentrations were lower owing to reduce scavenging and increased water fluxes without distinct peaks; moreover, only during the first year post-clear cut were concentration within the range of the intact stand. Pre-clear cut, Al in seepage water at BH was low (Fig. 5), occurring mostly with elevated NO_3^- concentrations (data not shown). At HW, Al concentrations (during 1989–1999) were comparable to NO_3^- concentrations (albeit lower). After the clear cut, Al at BH and HW shows same trend pattern as NO_3^- (compare with Fig. 1). Similarly, from (early) 2002 onwards, Al concentration at HW was lower than pre-harvest concentrations at both clear cut plots.

Sodium was the dominant cation at BH; in contrast, Na^+ concentrations were low at HW (Fig. 6). Following clear cut, Na^+ concentrations decreased below the pre-cut median at both study sites (Fig. 6). However, there was a noticeable peak in Na^+ concentrations at BH immediately post-clear cut, which clearly deviated from the Cl^- pattern (see Fig. 4). Pre-clear cut Mg^{2+} concentrations were higher at HW than BW (Fig. 7). Clear cutting resulted in distinct peak concentrations at both study sites within the first year of cutting and lesser pronounced peak concentrations during the next year; further, there were small peaks in 1999 at BH. However, Mg^{2+} concentrations post-clear cut were predominantly lower than pre-clear cut concentrations. Calcium concentrations were also higher at HW than at BH (Fig. 8) during the pre-clear cut period. At both study sites, clear cut had lesser impacts on Ca^{2+} in seepage waters than on Al (see Fig. 5), i.e., Ca^{2+} peaks were less pronounced post-clear cut and were generally lower than pre-clear cut concentrations. However, at BH peak Ca^{2+} concentrations following clear cut were higher than pre-clear cut. Potassium concentrations were low at both study-sites pre-clear cut (Fig. 9). At HW, K^+ was not altered to any extent following cutting, while at BH the K^+ concentrations were elevated for nearly the entire investigation period (mid-1996–1999). Moreover, there were distinct peak concentrations following clear cut and concentrations were elevated above HW (in contrast to pre-clear cut).

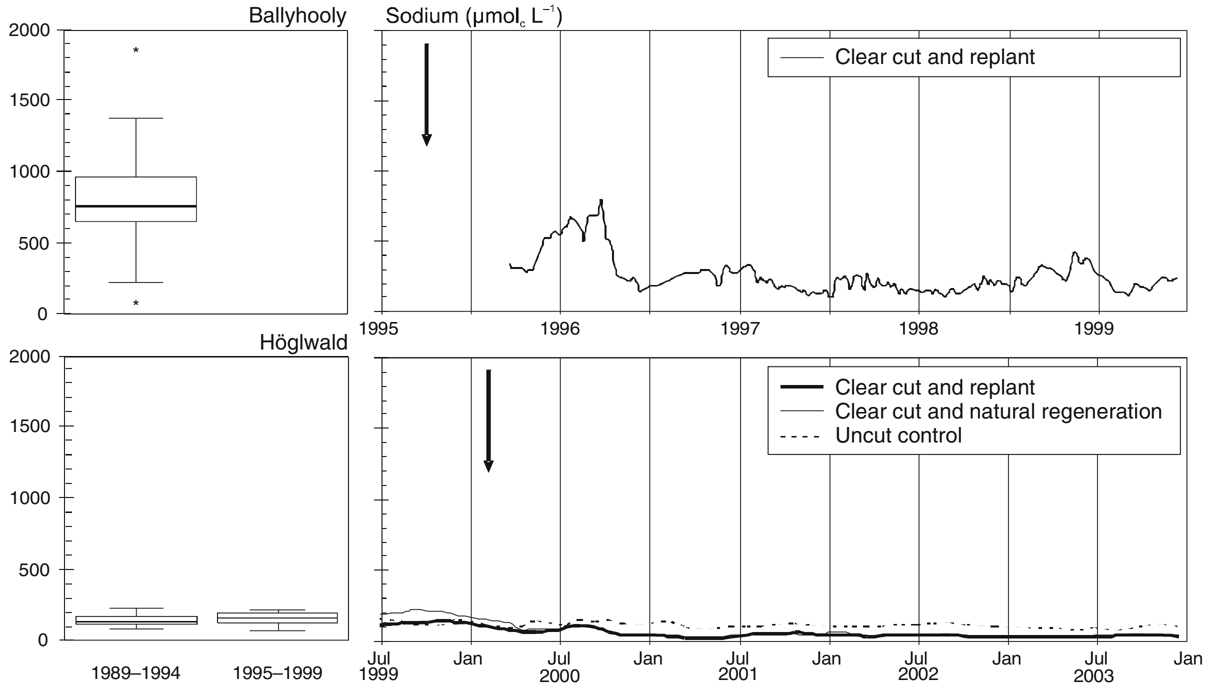

Fig. 6 Sodium (Na^+) concentrations below the main rooting zone at Ballyhooly (75 cm depth) and Höglwald (40 cm depth) before (box plot summaries: *left*) and after clear cutting (time-series of mean concentrations: *right*). See Fig. 1 for further details

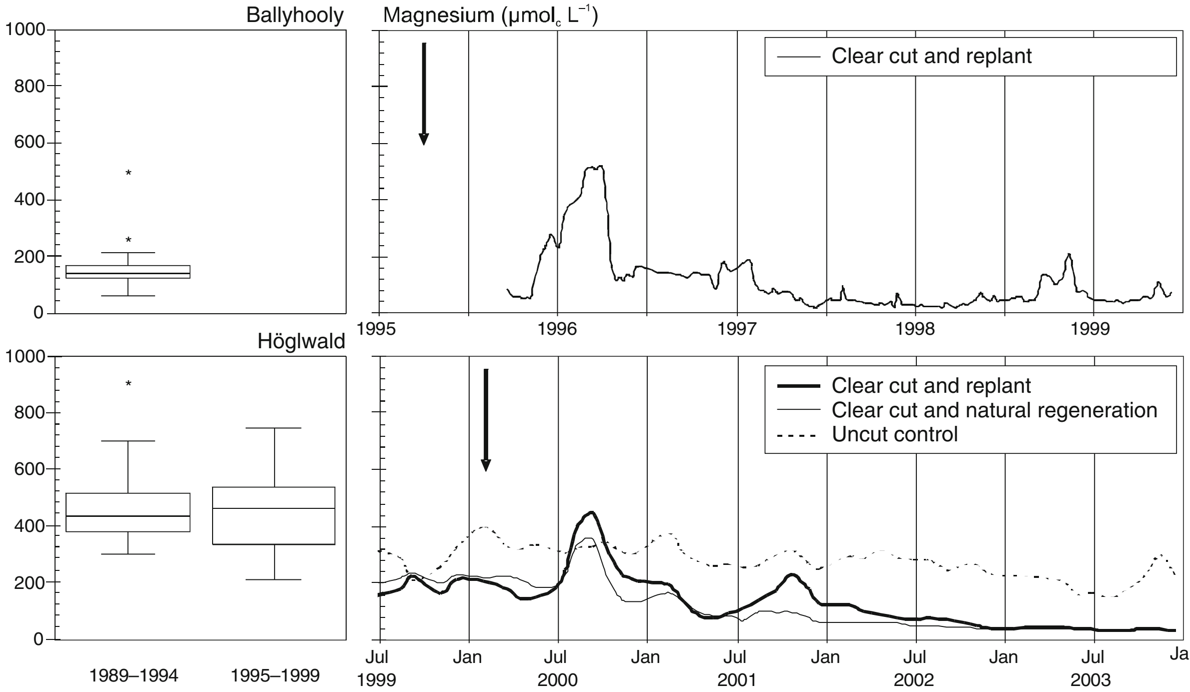

Fig. 7 Magnesium (Mg^{2+}) concentrations below the main rooting zone at Ballyhooly (75 cm depth) and Höglwald (40 cm depth) before (box plot summaries: *left*) and after clear cutting (time-series of mean concentrations: *right*). See Fig. 1 for further details

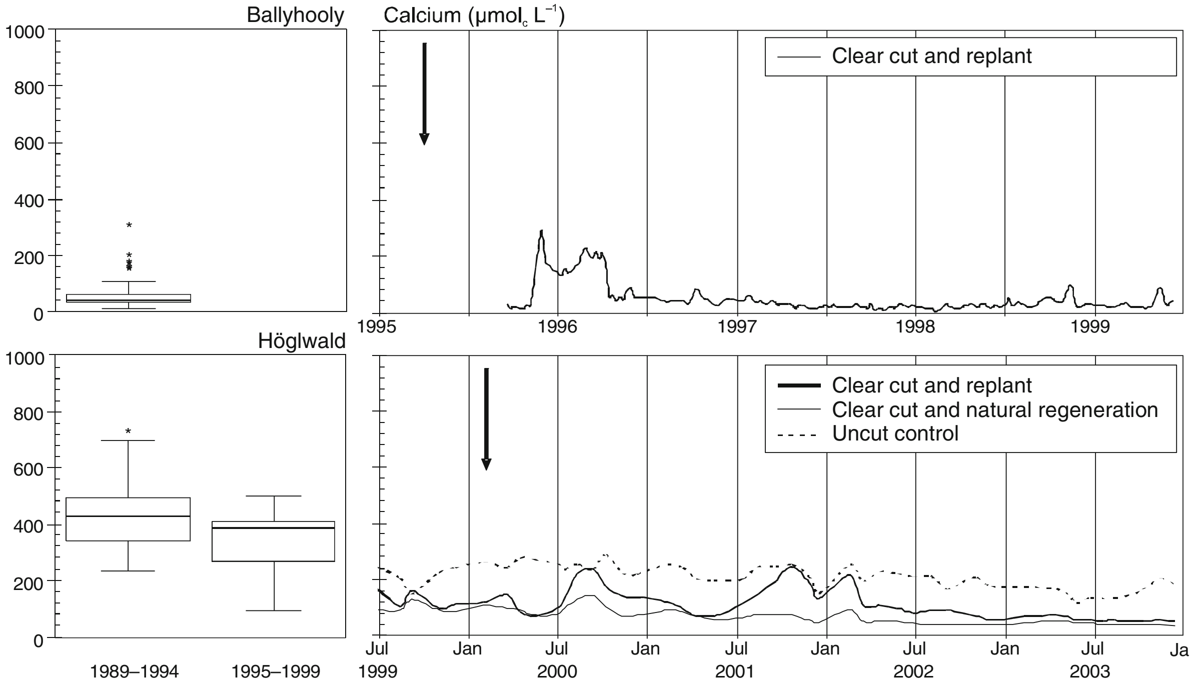

Fig. 8 Calcium (Ca^{2+}) concentrations below the main rooting zone at Ballyhooly (75 cm depth) and Höglwald (40 cm depth) before (box plot summaries: *left*) and after clear cutting (time-series of mean concentrations: *right*). See Fig. 1 for further details

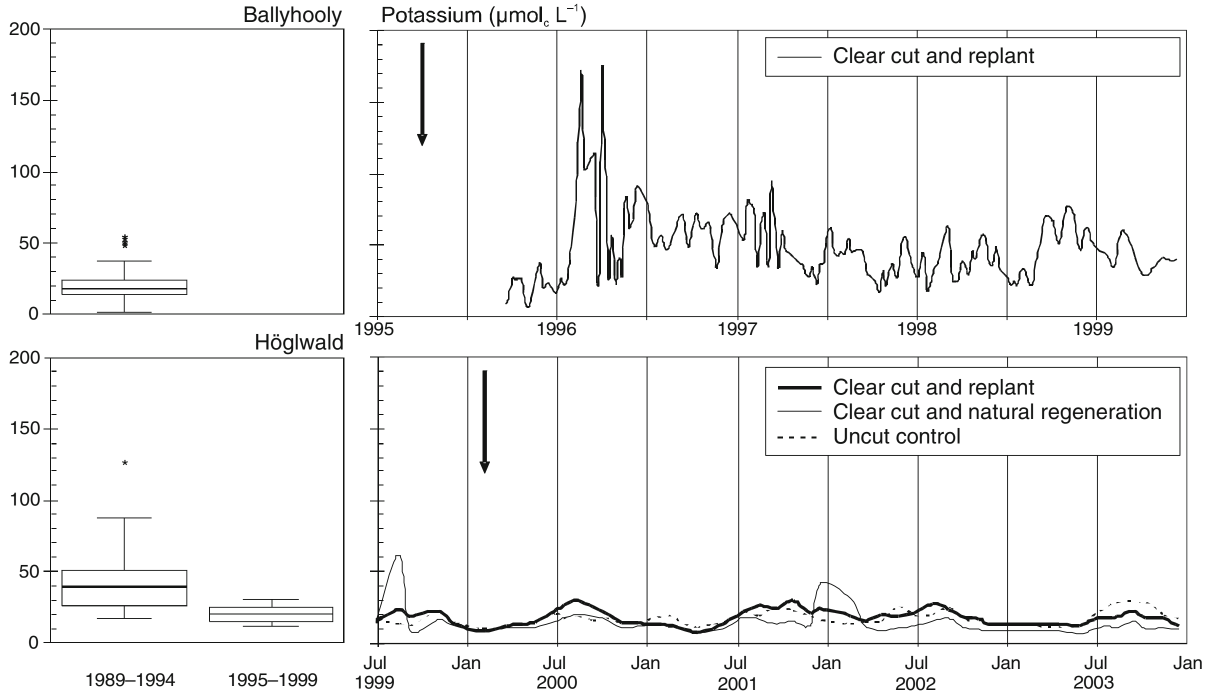

Fig. 9 Potassium (K^+) concentrations below the main rooting zone at Ballyhooly (75 cm depth) and Höglwald (40 cm depth) before (box plot summaries: *left*) and after clear cutting (time-series of mean concentrations: *right*). See Fig. 1 for further details

Water and ion fluxes before and after clear cutting

Following clear cut, canopy interception was strongly reduced; as such, water fluxes at both sites were more than threefold higher post-clear cut than pre-clear cut and similarly higher than uncut controls (Table 6). However 2003 at HW was an exceptional dry year with much smaller water and ion fluxes. In comparison, water fluxes were slightly higher at BH during pre- and post-clear cut. Higher ion fluxes of Ca^{2+}, Mg^{2+}, NO_3^-, SO_4^{2-} and Al^{3+} were observed at HW, while higher fluxes of Na^+ and Cl^- were observed at BH, and higher K^+ fluxes at BH post-clear cut. Following clear cut at HW, the highest NO_3^- flux was observed at the clear cut replanted plot attributable to elevated fluxes during the first 2 years after cutting (Table 6). In contrast, the lowest NO_3^- flux in seepage water (export) was observed at the clear cut plot with natural spruce regeneration. Similar results were also observed for Al^{3+}, but with lower Al^{3+} fluxes on the uncut control compared to the clear cut with natural regeneration. Higher fluxes for nearly all ions were observed at both clear cut replant study plots, compared to the pre-clear cut (and uncut control); the highest relative differences were observed for NO_3^-, Al^{3+} and K^+. At HW, the clear cut with natural regeneration had lower NO_3^-, Ca^{2+} and Mg^{2+} fluxes but slightly elevated fluxes of K^+, Al^{3+} and SO_4^{2-}. At BH, the fluxes of all ions increased after clear cutting, except Na^+ and Cl^-. The highest absolute increases were found for NO_3^- and Al^{3+} at BH, while the highest relative increase were for NO_3^- and K^+.

Discussion

Nitrate leaching

The highest NO_3^- concentrations at BH and HW were observed following clear cut. Average peak concentrations were up to 2018 μmol$_c$ L^{-1} (BH) and 2595 μmol$_c$ L^{-1} (HW), with up to 3675 μmol$_c$ L^{-1} at single suction cups (HW). Similar results were observed following a later clear cut at HW (see Kohlpaintner et al. 2009); however, higher NO_3^- concentrations were found after liming of the spruce stand (see Huber et al. 2006b), where large amounts

Table 6 Ion fluxes at Höglwald (*HW*) and Ballyhooly (*BH*) below the main rooting zone (BH 75 cm depth, HW 40 cm depth) for different periods

Plot and period	Flux (mm)	Ca^{2+} (mmol$_c$ m^{-2})	Mg^{2+} (mmol$_c$ m^{-2})	K^+ (mmol$_c$ m^{-2})	Na^+ (mmol$_c$ m^{-2})	H^+ (mmol$_c$ m^{-2})	Al^{3+} (mmol$_c$ m^{-2})	NO_3^- (mmol$_c$ m^{-2})	SO_4^{2-} (mmol$_c$ m^{-2})	Cl^- (mmol$_c$ m^{-2})
Höglwald UCC										
2000	329	87	111	5	41	26	302	370	205	20
2001	317	68	94	5	34	222	266	326	183	17
2002	472	90	122	8	52	34	361	399	267	32
2003	112	19	26	2	12	6	63	59	72	7
Σ 2000–2003	1230	263	354	20	140	88	992	1154	728	76
Höglwald CCR										
2000	1012	145	260	19	77	79	1085	1266	341	48
2001	1046	134	144	18	37	61	751	808	275	24
2002	1088	95	77	21	37	55	342	253	272	33
2003	448	26	18	7	14	21	80	36	97	11
Σ 2000–2003	3594	400	499	65	165	215	2258	2363	985	115
Höglwald CCN										
2000	766	73	174	11	65	56	598	530	413	35
2001	823	56	82	12	31	45	344	236	307	15
2002	900	39	45	11	34	39	205	46	295	8
2003	318	13	12	3	12	12	55	11	86	4
Σ 2000–2003	2807	181	313	37	142	151	1202	823	1102	62
Ballyhooly CCR										
1996	1030	59	144	33	353	63	345	471	126	308
1997	1037	33	88	61	203	68	332	348	124	197
1998	980	22	43	43	180	27	51	31	143	192
1999	926	20	60	35	196	23	39	92	119	203
Σ 1996–1999	3973	134	335	172	932	181	767	942	512	900
Annual average fluxes										
HW: 1989–1994	177	85	87	7	29	10	169	218	188	21
HW UCC: 2000–2003	308	66	89	5	35	22	248	289	182	14
HW CCR: 2000–2003	899	100	125	16	41	54	565	591	246	29
HW CCN: 2000–2003	702	45	78	9	36	38	301	206	276	16
BH: 1989–1994	334	16	54	5	285	12	41	18	58	342
BH CCR: 1996–1999	993	34	84	43	233	45	192	236	128	255

Clear cutting was carried out during October 1995 at BH and during February 2000 at HW. Plots: clear cut replant (*CCR*), clear cut and natural regeneration (*CCN*) and uncut control (*UCC*) of the organic N in the surface humus was mineralized. Elevated NO_3^- concentrations after disturbances due to clear cutting or other harvesting methods (Vitousek et al. 1979; Dahlgren and Driscoll 1994; Weis et al. 2006; Fukuzawa et al. 2006), windthrows (Mellert et al. 1996), or insect attacks (Huber et al. 2004c, Huber 2005) have been frequently observed. Nitrate concentrations generally increase within the first two or three years (Dahlgren and Driscoll 1994; Gundersen et al. 2006; Weis et al. 2006) owing to mineralisation and nitrification of litterfall initially and harvesting residues. The decrease in NO_3^- after some years can be explained by a decrease of easily degradable N compounds, the lower atmospheric input due to the absence of the canopy, and the increased uptake of N by new vegetation (Weis et al. 2006). However, on sites with short vegetation periods and (or) ongoing disruptions (such as herbicide treatment), longer lasting NO_3^-concentrations have been observed (Huber 2005). In some studies, NO_3^- concentrations following clear cut were at least episodically elevated above European drinking water standards, notably in southern Germany (Mellert et al. 1996; von Wilpert et al. 2000; Rothe and Mellert 2004; Huber 2005; Weis et al. 2006). Ballyhooly is one of only a few studies outside of Germany (e.g., Ravels forest in Belgium) where NO_3^- exceeded the European standards for drinking water following clear cut (see data given in Gundersen et al. 2006). This may be due to the relatively high atmospheric N input to these forests, where a high proportion of the stands has already reached N saturation (Mellert et al. 2005), but may also be due to a lack of newer clear-cut investigations in other areas, where N saturation has occurred or is highly likely (e.g., The Netherlands). In most clear cut studies, NO_3^- concentrations were significant but remained below the drinking water standard, while in many studies very low concentrations ($\leq$50 μmol$_c$ L^{-1}) were observed (e.g. Johnson and Todd 1987; Reynolds and Edwards 1995; Neal et al. 1998; Piirainen et al. 2002; Cummins and Farrell 2003). An important prerequisite for significant NO_3^- leaching is that sites are N saturated or at least not N limited. At the study plots, NO_3^- leaching was observed over >20 years at HW and periodically at BH prior to clear cut; the soil C:N ratio at both plots was close to the threshold of 30, below which there is potential for NO_3^- leaching. Moreover, both mature stands exhibited optimal N nutrition in current needles (see Table 2 and Huber et al. 2006a). According to Aber et al. (1998), N_2O emission is a sign of N saturation; very high N_2O emission was observed at HW, and both emission and deposition of N_2O were also observed at BH (Butterbach-Bahl et al. 1998), suggesting that BH was close to or at the beginning of N saturation.

Spatial and temporal variability in nitrate leaching

The spatial variability of NO_3^- leaching at HW is extremely high (Fig. 2); higher concentrations of NO_3^- are usually found close to the stem (Huber et al. 2004c), however, heterogeneity in the interstem area is quite significant before and after the clear cut. The reasons for this observation are not entirely understood; the results of Kohlpaintner et al. (2009) at a clear-cut plot nearby HW indicate that sites with dryer conditions, thinner surface humus layers and higher coverage of vascular plants exhibited lower NO_3^- concentrations. While a higher coverage of plants may indicate a higher N-uptake, patterns with drier conditions and thinner surface humus layers may have lower net and gross nitrification, which were found to correlate with NO_3^- in seepage water at the HW clear cut plot (Matejek et al. 2008).

While clear cutting caused a ~1.5 year lasting increase in of seepage water NO_3^- concentrations at BH, the effects at HW could be discussed in greater detail (Fig. 1). At the clear cut plot, periodically elevated NO_3^- concentrations were observed; however, elevated water fluxes and reduced deposition resulted in lower concentrations between peaks. The establishment of a dense regeneration before clear cutting may be a strategy to reduce NO_3^- fluxes; NO_3^- concentrations and fluxes pre- and post-clear cut at the natural spruce regeneration plot were much lower than at the clear cut plot or even compared to the uncut control. Mellert et al. (1996) found lower NO_3^- concentrations in wind thrown stands with dense birch regeneration. At BH, the presence of ground vegetation may also have reduced the NO_3^- peak in the second year to some extent (owing to increased uptake), as there was a significant shrub and herb layer already present before the clearance. Ground vegetation is considered to reduce NO_3^- concentrations after clear cutting (Emmett et al. 1991; Mellert et al. 1996; De Keersmaeker et al. 2000;

Parfitt et al. 2002; Stevens and Hornung 1990; Weis et al. 2001; Gundersen et al. 2006). At HW, the forest floor was covered nearly exclusively by mosses (Table 4), and so herbs could not occupy the clear cut rapidly enough to reduce NO_3^- concentrations more effectively. Similar results were found at a further clear-cut experiment at Höglwald, which was performed in close vicinity some years later (Kohlpaintner et al. 2009). However, even when measuring places were significantly covered with ground vegetation (>60% coverage) the average reduction in nitrate concentration was small. At measuring spots with 0% coverage with ground vegetation the average NO_3^- concentration was 135.9 mg L^{-1}, while at spots with >60% coverage the mean NO_3^- concentration was 117.4 mg L^{-1} (see Kohlpaintner et al. 2009).

BH and HW were influenced by dry years; at BH exceptional dry periods were observed in 1997 (2 years after clear cutting), and at HW an extreme summer drought occurred in 2003 (4 years after clear cutting). While fluxes of all ions at HW are lower at the uncut control in this year compared to the other years, only NO_3^- and Al concentrations decreased. Lowered NO_3^- concentrations in dry years are typically found at HW, concentrations similar to 2003 were found for dry periods during 1994 and 1997 (see Fig. 6 in Huber et al. 2006b). Drought experiments at BH did not increase NO_3^- leaching (Lamersdorf et al. 1998; Ryan et al. 1998); therefore the peaks found in 1997 are probably attributable to clear cutting and not to the dry period. Further, it is unlikely that sea salt deposition had any noticeable impact at BH following clear cut. It has been suggested that high NaCl concentrations can inhibit nitrification (Quanzhong and Guanhua 2009); however, sea salt deposition was reduced following clear cut and NO_3^- leaching at BH was elevated for ~1.5 years.

Aluminium leaching

Aluminium concentrations in seepage water showed the same trend as NO_3^- at both study sites. At BH, significant Al concentrations were observed in association with NO_3^-, while at HW Al was also present when low NO_3^- concentrations were observed. Kreutzer et al. (1998) found a better correlation between exchangeable Al and Al in soil solution for the HW and Solling (Germany) forest plots, while the relationship at coastal sites like BH and Klosterhede (Denmark) was rather weak. The low Al in soil solutions at coastal sites occurs owing to the high portions of Na^+, which has low adsorption strength (Kreutzer et al. 1998). This suggests that high Al saturation of the cation exchange complex is necessary but not sufficient for elevated Al in soil solution even when the sum of acid ions is rather high (as in BH). At BH, Al is released in seepage water mainly after acidification pulses due to nitrification, especially after clear cutting. The high correlation between Al and NO_3^- was also observed in numerous others N saturated acidified sites, especially after disturbances of the forest cover (e.g. Huber et al. 2004c; Weis et al. 2006). Despite the low level of air pollution and atmospheric deposition of anthropogenic acidifying substances (Aherne and Farrell 2002), BH showed deep reaching soil acidification, which is rather common in Ireland on soils derived from base poor parent material vegetated by conifer and oak forests or heath (Kreutzer et al. 1998), Despite significant N inputs, the leaching of NO_3^- at BH (the main acidifying component at HW (Rothe et al. 2002)) was low. Some acidification may be attributed to the export of base cations after tree harvesting, but most of the acidification at BH is probably due to the formation of acid mor humus layers as a result of the wet climate. The transfer of dissolved organic compounds from the humus layer to the mineral soil are primarily responsible for increased silicate weathering in the mineral soil up to the Al buffer range and losses of other cations (Kreutzer et al. 1998). Nonetheless, higher input of sea salts (e.g., storm events) will increase the Al concentration in seepage water, as Al will be exchanged by the deposited seasalt-derived cations (Beier et al. 2003).

At HW, Al contamination of groundwater is unlikely as it will be exchanged with other cations in the Ca^{2+} and Mg^{2+} rich deeper soil horizons; in contrast groundwater contamination at BH is more likely (see Table 3 for soil data). High pulses of Al and low Ca:Al and Mg:Al ratios during the growing season are indicative of risk to sapling tree growth according to Cronan and Grigal (1995). However, it is well known that European beech and Norway spruce can avoid Al stress when they are well supplied with nutrients and other growth factors

(Marschner 1991; Kreutzer 1995, Kreutzer et al. 1998; Huber et al. 2004a).

Following clear cut, peaks in Na^+, Cl^- and $SO_4{}^{2-}$ (comparable to $NO_3{}^-$ and Al^{3+}) were not observed. Pre-clear cut Na^+ and Cl^- concentrations were much higher at BH, because of the significance of sea salt deposition; similarly $SO_4{}^{2-}$ deposition decreased following clear cut at both plots. The decrease is not surprising as mineralization of these ions is typically low. Moreover, soil adsorption of Na^+ and Cl^- is generally small. In contrast, adsorption of $SO_4{}^{2-}$can be significant; recent $SO_4{}^{2-}$ concentrations at HW were still influenced from periods of higher $SO_4{}^{2-}$ input during the 1960s and 1970s (Weis et al. 2007). While a decrease in $SO_4{}^{2-}$ was observed at HW, the effects at BH were small despite reduced $SO_4{}^{2-}$ (due to the absence of the canopy). The delayed response at BH also suggests potential $SO_4{}^{2-}$ desorption processes, which is in agreement with previous modelling studies (Aherne et al. 1998). Despite the potential of acidity generation by sea salt deposition and $SO_4{}^{2-}$ desorption, nitrification and $NO_3{}^-$ leaching were the dominant acidification process driving Al leaching.

Base cation leaching

Potassium concentration at HW was nearly unaffected by clear cutting, while at BH a sharp increase in K^+ was observed. Further, due to the higher water fluxes, K^+ fluxes were higher on both clear cut plots, but the increase at BH was more pronounced. The increase in K^+ concentrations and fluxes reflect the reduced uptake (post harvest) and the fast release of K^+ from slash (especially needles), while later K^+ leaching may be due to decay of roots and woody material (Katzensteiner 2003). Potassium leaching differs widely from site to site; at Hubbard Brook clear-cutting resulted in large and persistent losses of K^+ in stream water (Likens et al. 1994), which were highly correlated with $NO_3{}^-$ (Dahlgren and Driscoll 1994). A rapid increase in K^+ concentrations was also found in a study in Wales (Reynolds et al. 1995), in Austria (Northern Limestone Alps; Katzensteiner 2003) and after a bark beetle attack in the Bavarian Forest National Park (Huber et al. 2004c). However, Weis et al. (2006) found no increased K^+ leaching after clear cutting of Norway spruce at Ebersberg, Southern Germany, mostly because the slash, which is an important K^+ source, was removed. Often K^+ concentrations tend to decrease with depth in the soil profile (Dahlgren and Driscoll 1994; Huber et al. 2004c). Potassium losses comprise a substantial depletion for ecosystems with low K^+ supply (see Katzensteiner 2003; Huber et al. 2004c). This may be relevant for BH, and should be taken into account when calculating nutrient losses after harvesting. In comparison, K^+ losses at HW were low, potentially due to K^+-fixation in the soil. Although no detailed investigations have been carried out, K^+-fixation is responsible for K^+ depleted agricultural soils in the region (Vogel and Niederbudde 1984). Larger peaks were observed for Mg^{2+} than for Ca^{2+} at both sites. As with K^+, the response at HW was smaller than at BH. Magnesium mineralised from slash material may be transported much faster than Ca^{2+} because of the lower binding capacity of Mg^{2+} compared to Ca^{2+}, clearly seen after dolomitic limestone application at the HW site (Huber et al. 2006b).

Conclusion

The study sites were N saturated (HW) or close to N saturation (BH), which is an important precondition for elevated $NO_3{}^-$ concentrations after clear cutting (according to Gundersen et al. 2006). Consequently, following clear cut both sites showed elevated $NO_3{}^-$ concentrations ($\sim$1.5 years duration) at levels not previously observed. At BH, the clear cut was responsible for long lasting increases in $NO_3{}^-$ and Al^{3+} concentrations, notably Al^{3+} peaks were most pronounced during nitrification episodes. Clear cutting was also responsible for significant leaching of K^+, $Mg^{2+}$and Ca^{2+} at BH. While Mg^{2+} was not problematic due to significant sea salt inputs at BH, K^+ and Ca^{2+} losses may be critical for the next-generation trees. At the HW the primarily concern is the contamination of seepage water with $NO_3{}^-$ and ongoing soil acidification (see Weis et al. 2007). The critical point for forest management at sites like HW is the reduction of $NO_3{}^-$ leaching and subsequent base cations losses during the whole rotation cycle. One strategy would be to initiate forest regeneration much earlier; ideally before $NO_3{}^-$ leaching takes place (Huber et al. 2004b, Weis et al. 2006).

Sustainable nutrient management should be considered in the context of forest sustainability;

Sverdrup et al. (2006) provide some guidance on assessing nutrient sustainability by integrating biogeochemical models (Warfinge and Sverdrup 1992; Aherne et al. 1998; Cosby 2001) with mass balance calculations. Mass balances can be calculated for each nutrient incorporating deposition and weathering (sources) and uptake and leaching (sinks). However, all management phases including clear cutting must be incorporated into nutrient sustainability. While forest operations have moved towards sustainable management including the protection of water quality, these practices are under pressure from the increased demand worldwide for wood, including harvesting residues as a bioenergy resource, which is seen as an important mitigation strategy against the effects of climate change. This issue may be all the more pressing given the dramatic increases in global N emissions and concerns that chronic elevated N deposition may lead to ecosystems saturation.

Acknowledgments The data were compiled during a Short-Term Lectureship at Trent University awarded to Christian Huber by the German Academic Exchange Service (DAAD). Funding was provided in part by the Irish Department of Agriculture, Food and Fisheries under the STIMULUS fund programme 2007. This research was also undertaken, in part, thanks to funding from the Canada Research Chairs Program and an NSERC Discovery grant. Most of the work at Höglwald was funded by the Bundesministerium für Bildung, Forschung und Technologie (BMBF), under contract number 0339733A and other contacts before. The Bayerische Staatsministerium für Landwirtschaft und Forsten helped with funding the projects B60 and St240.

References

Aber JD, Nadelhoffer KJ, Streudler P, Mellio JM (1989) Nitrogen saturation in northern forest ecosystems. BioScience 39:378–386

Aber J, McDowell W, Nadelhoffer K, Magill A, Berntson G, Kamakea M, McNulty S, Currie W, Rustad L, Fernandez I (1998) Nitrogen saturation in temperate forest ecosystems. Hypotheses revisited. BioScience 48:921–934

Aherne J, Farrell EP (2002) Deposition of sulphur, nitrogen and acidity in precipitation over Ireland: chemistry, spatial distribution and long-term trends. Atmos Environ 36:1379–1389

Aherne J, Sverdrup H, Farrell EP, Cummins T (1998) Application of the SAFE model to a Norway spruce stand at Ballyhooly, Ireland. For Ecol Manage 101:331–338

Aherne J, Farrell EP, Cummins T (1999) Modelling soil water fluxes in a Norway spruce (*Picea abies* (L.) Karst.) stand at Ballyhooly, Co Cork. Irish For 56(2):22–27

Alewell C, Manderscheid B, Meesenburg H, Bittersohl J (2000) Environmental chemistry–is acidification still an ecological threat? Nature 407:856–857

Beier C, Moldan F, Wright RF (2003) Terrestrial ecosystem recovery–modelling the effects of reduced acidic inputs and increased inputs of sea-salts induced by global change. Ambio 32:275–282

Burns DA, Murdoch PS (2005) Effects of a clearcut on the net rates of nitrification and N mineralization in a northern hardwood forest, Catskill Mountains, New York, USA. Biogeochemistry 72:123–146

Butterbach-Bahl K, Gasche R, Huber C, Kreutzer K, Papen H (1998) Impact of N-input by wet deposition on N-trace gas fluxes and CH_4-oxidation in spruce forest ecosystems of the temperate zone in Europe. Atmos Environ 32:59–564

Connaughton MJ (1967) Global solar radiation, potential evaporation and potential water deficit in Ireland. Department of Transport and Power Meteorological Service. Technical note 31, 26 pp

Cosby BJ (2001) Modelling the effects of acid deposition: refinements, adjustments and inclusion of nitrogen dynamics in the MAGIC model. Hydrol Earth Syst Sci 5:499–517

Cronan CS, Grigal DF (1995) Use of calcium aluminium ratios as indicators of stress in forest ecosystems. J Environ Qual 24:209–226

Cummins T, Farrell EP (2003) Biogeochemical impacts of clearfelling and reforestation on blanket peatland streams. II. Major ions and dissolved organic carbon. For Ecol Manage 180:557–570

Cummins T, Beier C, Blanck K, de Visser PHB, Farrell EP, Rasmussen L, Kreutzer K, Weis W, Bredemeier M, Lamersdorf N (1995) The Exman project—biogeochemical fluxes in plantation forests on acid soils. Water Air Soil Pollut 85:1653–1658

Dahlgren RA, Driscoll CT (1994) The effects of whole-tree clear-cutting on soil processes at the Hubbard Brook Experimental Forest, New Hampshire, USA. Plant Soil 158:239–262

De Bruin HAR (1987) From Penman to Makinik, TNO committee on hydrological research. The Hague. Environment and weather. Proceedings and information 39:5–32

De Keersmaeker L, Neirynck J, Maddelein D, De Schrijver A, Lust N (2000) Soil water chemistry and revegetation of a limed clearcut in a N saturated forest. Water Air Soil Pollut 122:49–62

EC (1998) Council regulation 98/83/EC of 3 November 1998 on the quality of water intended for human consumption. Official Journal of the EU, No. 330, Brussles, 32 pp

EEC (1986) EEC Council Regulation 3528/86 of 17 November 1986 on the protection of the Community's forests against atmospheric pollution. Official Journal of the EEC, No. L362/2, Brussels, 24 pp

Emmett BA, Anderson JM, Hornung M (1991) The control on N-losses following two intensities of harvesting in a Sitka spruce forest (N. Wales). For Ecol Manage 41:81–93

Farrell EP (1995) Atmospheric deposition in maritime environments and its impacts on terrestrial ecosystems. Water Air Soil Pollut 85:123–130

Farrell EP, Cummins T, Boyle GM, Smillie GW, Collins JF (1993) Intensive monitoring of forest ecosystems. Irish For 50(1):53–69

Farrell EP, Cummins T, Collins JF, Beier C, Blanck K, Bredemeier M, de Visser PHB, Kreutzer K, Rasmussen L, Rothe A, Steinberg N (1994) A comparison of sites in the EXMAN project, with respect to atmospheric deposition and the chemical composition of the soil solution and foliage. For Ecol Manage 68:3–14

Farrell EP, Boyle GM, Cummins T, Aherne J, van der Beuken R (1996) Continued monitoring of a forest ecosystem in Ireland, BAL3 project, Final Report. Forest Ecosystem Research Group Report 17, Ballyhooly. Department of Environmental Resource Management, University College, Dublin, 122 pp

Farrell EP, Van Den Beuken R, Boyle GM, Cummins T, Aherne J (1998) Interception of seasalt by coniferous and broadleaved woodland in a maritime environment in western Ireland. Chemosphere 36:985–987

Farrell EP, Führer E, Ryan D, Andersson F, Hüttl R, Piussi P (2000) European forest ecosystems: building the future on the legacy of the past. For Ecol Manage 132:5–20

Fukuzawa K, Shibatab H, Takagic K, Nomurac M, Kurimad N, Fukazawad T, Satohb F, Sasae K (2006) Effects of clear-cutting on nitrogen leaching and fine root dynamics in a cool-temperate forested watershed in northern Japan. For Ecol Manage 225:257–261

Galloway JN, Aber JD, Erisman JW, Seitzinger SP, Howarth RW, Cowling EB, Cosby BJ (2003) The nitrogen cascade. BioScience 53:341–356

Galloway JN, Dentener FJ, Capone DG, Boyer EW, Howarth RW, Seitzinger SP, Asner GP, Cleveland CC, Green PA, Holland EA, Karl DM, Michaels AF, Porter JH, Townsend AR, Vorosmarty CJ (2004) Nitrogen cycles: past, present and future. Biogeochemistry 70:153–226

Göttlein A, Kreutzer K (1991) The Höglwald experimental site as compared to other ecological field studies. In: Kreutzer K, Göttlein A (ed) Ökosystemforschung Höglwald, pp 22–29, Forstwissenschaftliche Forschungen 39, Verlag Paul Parey. Hamburg und Berlin

Gundersen P, Schmidt IK, Raulund-Rasmussen K (2006) Leaching of nitrate from temperate forests–effects of air pollution and forest management. Environ Rev 14:1–57

Huber C (2005) Long lasting nitrate leaching after bark beetle attack in the highlands of the Bavarian Forest National Park. J Environ Qual 34:1772–1779

Huber C, Kreutzer K, Röhle H, Rothe A (2004a) Response of artificial acid irrigation, liming, and N-fertilisation on elemental concentrations in needles, litter fluxes, volume increment, and crown transparency of an N saturated Norway spruce stand. For Ecol Manage 200:3–21

Huber C, Weis W, Baumgarten M, Göttlein A (2004b) Spatial and temporal variation of seepage water chemistry after femel and small scale clearcutting in a N-saturated Norway spruce stand. Plant Soil 267:23–40

Huber C, Baumgarten M, Göttlein A, Rotter V (2004c) Nitrogen turnover and nitrate leaching after bark beetle attack in mountainous spruce stands of the Bavarian forest national park. Water Air Soil Pollut Focus 4:391–414

Huber C, Weis W, Göttlein A (2006a) Tree nutrition of Norway spruce as modified by liming and experimental acidification at the Höglwald site, Germany, from 1982 to 2004. Ann For Sci 63:861–869

Huber C, Baier R, Göttlein A, Weis W (2006b) Changes in soil, seepage water and needle chemistry between 1984 and 2004 after liming an N-saturated Norway spruce stand at the Höglwald, Germany. For Ecol Manage 233: 11–20

Johnson DW, Todd DE Jr (1987) Nutrient export by leaching and whole tree harvesting in a loblolly pine and mixed oak forest. Plant Soil 102:99–109

Katzensteiner K (2003) Effects of harvesting on nutrient leaching in a Norway spruce (*Picea abies* KARST.) ecosystem on a Lithic Leptosol in the Northern Limestone Alps. Plant Soil 250:59–73

Kohlpaintner M, Huber C, Weis W, Göttlein A (2009) Spatial and temporal variability of nitrate concentration in seepage water under a mature Norway spruce [*Picea abies* (L.) KARST] stand before and after clear cut. Plant Soil 314:285–301

Kreutzer K (1995) Effects of forest liming on soil processes. Plant Soil 168–169:447–470

Kreutzer K, Weiss T (1998) The Höglwald field experiment–aims, concept and basic data. Plant Soil 199:1–10

Kreutzer K, Beier C, Bredemeier M, Blanck K, Cummins T, Farrell EP, Lamersdorf N, Rasmussen L, Rothe A, de Visser PHB, Weis W, Weiß T, Xu Y-J (1998) Atmospheric deposition and soil acidification in five coniferous forest ecosystems: a comparison of the control plots of the Exman sites. For Ecol Manage 101:125–142

Lamersdorf NP, Beier C, Blanck K, Bredemeier M, Cummins T, Farrell EP, Kreutzer K, Rasmussen L, Ryan M, Weis W, Xu Y-J (1998) Effect of drought experiments using roof installations on acidification/nitrification of soils. For Ecol Manage 101:95–109

Likens GE, Driscoll CT, Buso DC, Siccama TG, Johnson CE, Ryan DF, Lovett M, Fahey T, Reiners WA (1994) The biogeochemistry of potassium at Hubbard Brook. Biogeochemistry 25:12–61

Marschner H (1991) Mechanisms of adaption of plants to acid soils. Plant Soil 134:1–20

Matejek B, Kohlpaintner M, Dannenmann M, Huber C, Gasche R, Papen H (2008) The small-scale pattern of seepage water nitrate concentration in an N saturated spruce forest is regulated by net N mineralization in the organic layer. Plant Soil 310:167–179

Mellert K-H, Kölling C, Rehfuess KE (1996) Bioelement leaching from Norway spruce ecosystems in Bavaria after windthrow. Forstw Cbl 115:363–377

Mellert KH, Gensior A, Kölling C (2005) Stickstoffsättigung in den Wäldern Bayerns–Ergebnisse der Nitratinventur. Forstarchiv 76:35–43

Neal C, Reynolds B, Wilkinson J, Hill T, Neal M, Hill S, Harrow M (1998) The impact of conifer harvesting on runoff water quality: a regional survey for Wales. Hydrol Earth Syst Sci 2:323–344

Parfitt RL, Salt GJ, Hill LF (2002) Clear-cutting reduces nitrate leaching in a pine plantation of high natural status. For Ecol Manage 170:43–53

Piirainen S, Finer L, Mannerkosk H, Starr M (2002) Effects of forest clear-cutting on the carbon and nitrogen fluxes through podzolic soil horizons. Plant Soil 239:301–311

Quanzhong H, Guanhua H (2009) Effects of NaCl salt on mineralization and nitrification of a silt loam soil in the North China Plain. Int J Agric Biol Eng 2(2):14–22

Reynolds B, Edwards A (1995) Factors influencing dissolved nitrogen concentrations and loadings in upland streams of the UK. Agric Wat Manage 27:191–202

Reynolds B, Stevens PA, Hughes S, Parkinson JA, Weatherly NS (1995) Stream impacts of conifer harvesting in Welsh catchment. Water Air Soil Pollut 79:147–170

Richards LA (1931) Capillary conduction of liquids in porous mediums. Physics 1:318–333

Rothe A, Mellert KH (2004) Effects of forest management on nitrate concentrations in seepage water of forests in southern Bavaria, Germany. Water Air Soil Pollut 156:337–355

Rothe A, Huber C, Kreutzer K, Weis W (2002) Deposition and soil leaching in stands of Norway spruce and European beech: results from the Höglwald research in comparison with other European case studies. Plant Soil 240:33–45

Ryan MG, O'Toole P, Farrell EP (1998) The influence of drought and natural rewetting on nitrogen dynamics in a coniferous ecosystem in Ireland. Environ Pollut 102(S1):445–451

Solórzano L (1967) Determination of ammonia in natural waters by the phenolhypochlorite method. Limnol Oceanogr 14:799–801

Stern DI (2006) Reversal of the trend in global anthropogenic sulfur emissions. Global Environ Change 16:207–220

Stevens PA, Hornung M (1990) Effect of harvest intensity and ground flora establishment on inorganic N-leaching from a Sitka spruce plantation in north Wales. Biogeochemistry 10:53–65

Sverdrup H, Thelin G, Robles M, Stjernquist I, Sorensen J (2006) Assesing nutrient sustainability of forest production for different tree species considering Ca, Mg, K, N and P at Bjornstorp Estate, Sweden. Biogeochemistry 81:219–238

Tiktak A, Bouten W (1992) Modelling soil water dynamics in a forest ecosystem. III: model description and evaluation of discretization. Hydrol Process 6:455–465

Vitousek PM, Gosz JR, Grier CC, Melillo JM, Reiners WA, Todd RL (1979) Nitrate losses from disturbed ecosystems. Science 204:469–474

Vogel W, Niederbudde EA (1984) The K-potential of soils in Lower Bavaria as influenced by long-term K fertilization. Z. Pflanzenernähr Bodenkunde 147:1–12

Von Wilpert K, Zirlewagen D, Kohler M (2000) To what extent can silviculture enhance sustainability of forest sites under the immission regime in central Europe? Water Air Soil Pollut 122:105–120

Warfinge P, Sverdrup H (1992) Calculating critical loads of acid deposition with PROFILE—a steady-state soil chemistry model. Water Air Soil Pollut 63:119–143

Weis W, Huber C, Göttlein A (2001) Regeneration of mature Norway spruce stands: early effects of selective cutting on seepage water quality and soil fertility. In: Optimizing nitrogen management in food and energy production and environmental protection. Proceedings of the 2nd international nitrogen conference on science and policy. The Scientific World 1(S2):493–499

Weis W, Rotter V, Göttlein A (2006) Water and element fluxes during the regeneration of Norway spruce with European beech: effects of shelterwood-cut and clear-cut. For Ecol Manage 224:304–317

Weis W, Beier R, Huber C, Göttlein A (2007) Long term effects of acid irrigation at the Höglwald on seepage water chemistry and nutrient cycling. Water Air Soil Pollut Focus 7:211–223