Nutrients or pollutants? Nitrogen deposition to European forests

Werner Eugster and Matthias Haeni

Institute of Agricultural Sciences, Department of Environmental Systems Science, ETH Zürich, LFW C55.2, Universitätsstras 2, CH–8092 Zürich, Switzerland

Abstract

Forests take up gaseous, liquid, and particulate substances that are present in the air. Although considered nutrients on the one hand, nitrogen inputs exceeding the critical load that can be absorbed by an ecosystem act as pollutants. This chapter outlines the effects of N deposition to forest ecosystems and discusses recent progress that has been made to more accurately quantify dry deposition, which at many a forest location in Europe is larger than wet and occult deposition.

To quantify the effects of N deposition on tree growth, a good measure for net ecosystem production (NEP) is needed. Eddy covariance (EC) flux measurements are one established way to quantify NEP. While EC flux measurements are costly and remain restricted in their application to a few suitable locations, dendrometer measurements with high temporal resolution show a similar seasonal and annual signal to NEP. Such measurements are becoming increasingly important to quantify ecosystem biomass accumulation, which can be related to N deposition rates. The policy relevance of such activities emerges from the UNECE’s Gothenburg protocol to abate acidification, eutrophication and ground-level ozone, but also the quantification of natural sinks under the Kyoto protocol profits from such measurements.

1. Introduction

The effects of N deposition on forest ecosystems are described in the conceptual framework of Aber et al. (1989, 1998), which was further refined by Emmet (2007). N deposition affects all ecosystem compartments and alters many biogeochemical processes. Its effect is modified by site conditions, such as nutrient availability in the soil, species composition of the forest stand and management history, in combination with the changing climate conditions and pollutant loads.

In this chapter we will focus on N deposition loads to European forest ecosystems. The emphasis is on the most recent scientific developments aiming at quantifying N dry deposition, which tends to be the dominant pathway of N deposition in temperate climates where the majority of days do not see rainfall, and where wet deposition is hence unable to represent total ecosystem-scale N inputs. For conventional N deposition measurements it is
however indispensable to consult the standard protocols of ICP Forests (2010). It is not the intent of this chapter to duplicate the wealth of existing literature on non-dry deposition processes, but for the sake of completeness a very brief summary of all relevant pathways will be given to put dry deposition into the context of total N deposition. Effects of N deposition are always a combination of several input pathways and cannot be cleanly teased apart. But with respect to dry deposition, above-ground N uptake processes must be more strongly investigated than would be required in cases where the primary pathway of anthropogenic N inputs is wet deposition.

N input must be seen in a long-term perspective and on an integrative ecosystem scale. At short timescales and low N deposition rates the nutritional effect may be dominant, primarily increasing tree growth. At longer timescales and higher N deposition rates, this effect can become detrimental. This transition may be slow and cannot easily be generalized. Short-term studies with seedlings are best suited to study specific physiological processes, but the results obtained in this way cannot readily be transferred to mature trees and whole forest ecosystems.

N is the most important nutrient for plant growth and it is often the limiting factor for net primary production in temperate forest ecosystems (Vitousek and Field, 2001). Elaborated methods have been established to measure wet deposition within ICP Forests, and canopy balance models are used to estimate the total N deposition. At many European locations, however, dry deposition of reduced and oxidized N-containing gases is an even larger component of N deposition than wet deposition (see e.g. Flechard et al., 2011, Eugster et al., 1998). Newer developments now allow for a better quantification of dry deposition rates of gases and particles. After a short summary on effects of N deposition to forest ecosystems this chapter focuses on these recent developments that may in the future complement existing N deposition measurements in Europe, but also briefly addresses the other components of total N deposition measurements which must remain an essential part of long-term monitoring efforts even at sites where gaseous dry deposition is the primary source of N deposition.

2. Effects of nitrogen deposition to forest ecosystems

N uptake (Fig. 1) is typically via the roots of the trees, but it has been shown by many studies that canopy N uptake (CNU) in forests also play an important role (e.g. Schaefer and Reiners, 1990; Sievering et al., 2007). Earlier studies have mostly focused on foliar uptake by saplings and young plants in experimental treatments, and the results obtained in this way were then scaled up to whole canopies of more mature trees. It has been recognized that neglecting CNU in some of the widely used canopy budget models is probably a wrong assumption (ICP Forests, 2010, p. 39). This indicates where the most urgent present-day research needs are:
to obtain realistic and accurate estimates of ecosystem-scale total N deposition and to assess all pathways of N uptake that may have detrimental effects on ecosystems.

Figure 1 about here

It is agreed that N inputs below the critical loads are – in the long term – not detrimental to ecosystem structure and functions. Nitrogen deposition rates above a critical threshold will – again in the long term – lead to an impairment of the functions, depending on the site quality and other stressors present. Having said that, it should however also be recalled that the effect of N deposition on species composition in forests starts at rather low levels, and the apparently positive effect of added nutrients may have a strong negative effect on biodiversity (e.g. Vitousek et al. 1997, Bobbink et al. 1998, 2010, Kozovits and Bustamante, this volume). This may be a key issue to keep in mind in natural and semi-natural, extensively managed forest ecosystems in Europe. For Central Europe Ellenberg (1990) showed that the vast majority of endangered vascular plant species growing in forests and woodlands are associated with N-poor environments (up to 40% of the species that grow in the N-poor class 2 on the nominal scale by Ellenberg et al. 1991).

Foliar uptake leads to the fact that estimates of N input via throughfall and bulk deposition collection with subsequent calculation of total deposition with models may substantially underestimate actual N inputs to forest ecosystems (Eilers et al. 1992). Nave and Curtis (2011) estimated that more than 50% of atmospheric N deposition is actually intercepted and retained in forest canopies. Moreover, Eilers et al. (1992) found that above-ground uptake of N may cause nutritional disturbances by changes in the N:Ca and N:Mg ratios in the needles, which may lead to Mg-deficiency symptoms of Norway spruce (Eilers et al. 1992). Sparks (2009) clearly showed that direct foliar uptake of N is a direct addition of N to plant metabolism and could potentially more readily influence plant growth (Fig. 1, right) compared to soil-deposited N (Fig. 1, left). Present-day ecosystem process models however still do not explicitly simulate this more direct N pathway (see Sparks 2009), and hence cannot always adequately represent the effects of N deposition on forest growth.

Nihlgard (1985) drew the attention of the scientific community to N as a possible cause for forest dieback, and since then an ever increasing number of studies have addressed the question how N deposition – mostly in combination with other pollutants – affects tree growth and health status. Krupa (2003) provided a concise overview over the different pathways and effects on terrestrial ecosystems, and even more detailed information on specific aspects of pollutant fluxes to different forest tree species can be found in the book by Elling et al. (2007).

2.1 What are the typical detrimental effects of nitrogen deposition?

N deposition affects all compartments of forests. The narrowing of the carbon to nitrogen (C/N) ratio of the forest floor indicates accumulation of N and is a measure to assess the risk
of nitrate release in the seepage water. In general, a C/N below 25 can be seen as the threshold for enhanced nitrate leaching via the seepage water (e.g. Gundersen et al. 1998; Aber et al. 2003, Dise et al. 2009). At the same time the composition of the understory is altered in many regions leading to a reduced biodiversity (Bobbink et al. 2010; Nordin et al. 2006).

The N to nutrient ratios (N/Mg, N/K, N/P) in needles and leaves tend to become larger, indicating a shortage of other nutrients which will lead to an enhanced sensitivity to natural stress factors, like a lowered frost resistance and enhanced susceptibility to insects (e.g. Schulze 1989; Gundersen 1998; Flückiger and Braun 1999).

There is increasing evidence that in Central Europe the phosphorous (P) nutrition is also declining (Khanna et al. 2007; Prietzel et al. 2008), leading to – at least regional – reduced growth (Braun et al. 2010). The pathways are not fully understood, but a reduction of P uptake via roots due to N mediated inhibition of mycorrhiza is discussed (Aber et al. 1989; Braun et al. 2010; see Kraigher et al., this volume). The linkages between C, N and P must be taken into account for the modelling of carbon sequestration (Goll et al. 2012; see also Bytnerowicz et al., this volume).

To a certain degree N acts as a fertilizer. If the natural N limitation is removed, the trees react with enhanced growth. This is probably one reason for the observed increased growth of trees in Europe in the last decades (e.g. Spieker et al. 1996). However, this does not imply that the effect is also positive at the ecosystem level and over longer time periods; biodiversity of a forest, for example, is normally determined primarily by the understory vegetation where many N-sensitive species may disappear while the few tree species thrive (see review by Bobbink et al. 2010). De Shrijver et al. (2008) for example emphasize that the effects of N deposition on soil acidification, groundwater and surface water quality, biodiversity, and ecosystem services must be considered in addition to the simple nutritional effect that N deposition may exert on tree biomass increment alone.

The input of N above the demand for tree growth is termed N saturation, which means that no further increase of growth results from additional N input (De Vries et al. 2006, 2008, Laubhann et al. 2009). There are indications, that the enhanced growth is today halted on many sites (Nellemann and Thomsen 2001). The excess N cannot be taken up or stored in soil or vegetation and percolates through the soil, leading to acidification. This in turn means a depletion of base cations (Mg, K, Ca) or an increase of the amount of acidic cations (Al, Mn, H) in the soil. If the acidification potential of nitrogen is higher than the natural weathering rate of the soil forming bedrock, the soils acidify.

To integrate all these aspects, an ecosystem-specific critical threshold is defined above which ecosystem damage is likely to occur. This is taken into account in the calculation of the critical loads, which are the basis for the air pollution abatement policy in the UNECE region...
The site-specific calculated critical load for nutritional N is for most forests in the range of 10 to 20 kg N ha\(^{-1}\) yr\(^{-1}\) (Spranger et al. 2004, p. 125); the empirical critical load for e.g. coniferous woodland is about 5 to 15 kg N ha\(^{-1}\) yr\(^{-1}\) (Bobbink and Hettelingh 2010). On the European scale 74% of the total terrestrial area (forests and other ecosystems) of the EU27 countries (52% for all European countries) is at risk of nutrient N eutrophication (reference year 2000, Slootweg et al. 2010, p. 17).

Knowledge of the effects of N on the ecosystem level is essential. To find and verify cause-effect relations and to establish critical thresholds, the input of N in forests and the interrelations with other pollutants should be known as accurately as possible.

### 3. The components of nitrogen deposition

The IPC Forests (2010) manual lists all relevant components of N deposition:

1. Wet-only deposition
2. Bulk deposition
3. Dry deposition
4. Occult deposition
5. Throughfall deposition
6. Stemflow deposition
7. Total atmospheric deposition

These components are non-additive and basically reflect the difficulty of separating one component clearly from another. However, this would not be that critical, if a good measurement method existed to accurately quantify total atmospheric deposition. Its definition according to IPC Forests (2010) is the sum of wet-only deposition and dry deposition, but not including occult deposition. A more integrative definition would define total deposition as wet (rainfall and snow, Section 3.1), occult (fog, Section 3.2), and dry deposition (Section 3.3) as we will do in this text. In areas with substantial snowfall one could argue that snow deposition is a separate pathway for N input, but for the sake of simplicity we will consider snow deposition a component of wet deposition, since for the ecosystem this input becomes typically only relevant when the snow melts.

Despite the difficulties in terminology and how to measure the components it should be clear that total N input by all relevant pathways must be quantified accurately to relate measurable effects to the N deposition rates in order to specify and quantify detrimental effects (see Section 2) on ecosystems.
3.1 Wet deposition by rainfall and snow

3.1.1 Wet-only deposition

Wet deposition is the most important pathway of N deposition in tropical rainforests without a dry period. In temperate forests the contribution of wet deposition to total N deposition however depends (1) on the probability of rain at a given site, and (2) on the concentrations of N compounds in rainfall. At localities where rain is rare it is clear that wet deposition may be substantially less than 50% of total deposition. For example, in western Switzerland Eugster et al. (1998) found wet deposition rates on the order of 7–8 kg N ha\(^{-1}\) yr\(^{-1}\), which contributed roughly 14% to 33% to total N deposition. Wet deposition is often a larger-scale phenomenon since very local rains are restricted to short periods in summer, and even then the ions in the rainwater are often transported over larger distances and hence small-scale variability in wet deposition is small compared to the variability of the other components.

3.1.2 Bulk deposition

If bulk deposition is measured in place of wet-only deposition, then the deposition estimates tend to be higher. Even higher than bulk deposition are N deposition values determined from throughfall data in forest canopies. Throughfall is believed to reflect the composition of both incident precipitation (wet deposition) and dry deposition from N containing components that are deposited on branches and foliage during dry periods, which are then washed off by precipitation and can be collected as throughfall (e.g. Thimonier et al. 2005). If CNU takes place, or leaching from the canopy, then some assumptions must be made to be able to derive dry deposition from the difference between throughfall deposition and bulk deposition measurements (Thimonier et al. 2005). Namely, gaseous components can be taken up via stomata (see Cieslik, this volume) and hence will not get deposited on leaf surfaces and thus also will not be washed off. It may however hold for aerosol deposition of solid dry fine dust particles. The N input from aerosols is however a much smaller component than gaseous dry deposition (see Section 3.3).

3.1.3 Deposition by snow

Very little attention has been given to N deposition by snow so far. In northern countries and in mountain ranges this however may be a relevant additional pathway of N input. The scavenging of aerosol particles by falling snow flakes was found to be somewhat greater than that for liquid precipitation (Kyrö et al. 2009) and corresponded to rain intensities around 3 mm h\(^{-1}\) (rather intense downpour) even when snowfall remained below 0.2 mm h\(^{-1}\). This means that snow deposition of N should be carefully considered in polluted areas where the increased efficiency for scavenging aerosols may constitute a significant pathway for N deposition. Because snow deposition occurs during the dormant season of the vegetation, it may not immediately affect the ecosystem, but during spring snow melt a peak influx of
deposited ions may occur. For lakes and streams this was termed “acid shock” (Jeffries et al. 1979) which in a similar way can also be expected for N inputs deposited by snow.

### 3.2 Occult deposition by fog

In temperate forests, water inputs by fog are rather small, but N concentrations in fog water are one to two orders of magnitude larger than in rain water, and hence deposition amounts can be considerable at localities with frequent fog. At the Lägeren mixed forest in Switzerland, for example, the N input by fog was quantified at 4.3 kg N ha\(^{-1}\) yr\(^{-1}\) which is 16.4% of total N deposition (Burkard et al. 2003). The tree species composition is dominated by beech, followed by Norway spruce, ash, sycamore, silver fir and 8 less abundant species. This is 62% of wet deposition (6.9 kg N ha\(^{-1}\) yr\(^{-1}\)) or 28% of dry deposition (15 kg N ha\(^{-1}\) yr\(^{-1}\)). Also at the Waldstein site in the Fichtelgebirge, Germany, fog deposition contributed a similar amount of N to total deposition as was measured as wet-only deposition (Thalmann et al. 2002).

Although these are important N additions, an experimental treatment of selected trees at the Lägeren site with a mist that reflects N concentration in fog water primarily showed an increased growth effect without clear signs of induced stress symptoms (Wortman et al. 2012). Elemental concentrations in Lägeren beech leaves showed that both N/P and N/K ratios were 6% and 19% higher, respectively, for N-amended leaves in comparison with control leaves. This supports the expectation of CNU (Fig. 1) which is enhanced by occult deposition compared to the shorter contact time and lower N concentrations brought about by rainwater. On the other hand, occult deposition to temperate forests should not be overestimated: often the high frequencies of fog occurrence coincide with the cold period of the year when trees are dormant, and hence it remains to be investigated in more detail how important the timing of fog and its N input is to forest ecosystems.

### 3.3 Dry deposition of gases and aerosol particles

#### 3.3.1 Methods for field measurement

Gaseous dry deposition is most directly measured by the eddy covariance (EC) method (Aubinet et al. 2012). This method uses a three-dimensional anemometer – mostly one working with ultrasonic impulses to determine wind speed and air temperature – in combination with a fast gas analyzer that allows measuring every tenth of a second or so to systematically probe turbulent transport of the gas of interest. It is then assumed that the deposition rate equals the flux rate measured with EC at a certain height above the forest canopy. Such measurements are feasible for most gases such as NO (nitric oxide), NO\(_2\) (nitrogen dioxide), NO\(_x\) (the sum of NO and NO\(_2\)), HNO\(_3\) (nitric acid) and PAN (peroxyacetile nitrate).
The dry deposition of aerosol particles can also be measured with EC, but it is still a challenge: while instruments exist to measure the number flux of particles, it is not yet possible to measure the exact chemical composition at a time interval suitable for EC. Also gaseous fluxes of NH$_3$ are not easily measured directly with EC, as Ferrara et al. (2012) have shown. Hence many scientists chose a variant of the EC method that allows the deployment of relatively slow sensors via the relaxed eddy accumulation (REA) technique: the gas of interest or aerosol components are sampled in two reservoirs, one which collects air whenever the vertical wind speed is positive and above a given threshold value, while the other collects air from down-moving wind gusts with a wind speed above that threshold. The concentration difference between the two reservoirs can then be analysed with an analyser that requires considerably longer analysis time than that required by EC (which is on the order of fractions of a second in most cases). The deposition flux is then derived from that concentration gradient times a turbulent mixing coefficient (which relates the gradient to the corresponding flux), and a correction factor ($\beta$) to correct for the effect of the REA wind speed thresholds on the flux estimates. In the case of aerosols the REA method was used in several combinations to obtain defensible flux estimates.

All these methods are expensive, require high scientific and technical skills to operate, and can only be carried out at very few forest sites that meet all these requirements. Hence, probably the most important advance in the methodology of dry deposition flux measurements in the past decade was made during the NitroEurope project (http://www.nitroeurope.eu) with its development of a low-cost Delta denuder system for sampling gaseous NH$_3$, NO$_2$, HNO$_3$, HONO, SO$_2$, HCl and aerosol NH$_4^+$, NO$_3^-$, SO$_4^{2-}$, Cl$^-$, Na$^+$, Ca$^{2+}$, and Mg$^{2+}$ (Fig. 2). This allows obtaining monthly mean concentrations of the most important N gases and aerosol components, from which deposition can be estimated via transfer models. In this way an independent estimate of dry deposition can be made that does not require any assumptions on how the difference between throughfall and bulk precipitation may actually be related to dry deposition.

**3.3.2 Contribution to total N deposition**

Aerosol dry deposition was only estimated to be on the order of 2.0 to 2.3 kg N ha$^{-1}$ yr$^{-1}$ in parts of Switzerland (Eugster et al. 1998). Fluxes derived from Delta denuders as presented by Flechard et al. (2011) in their Table 3 were used for additional analyses presented here. These numbers suggest that aerosol dry deposition (NH$_4^+$ and NO$_3^-$) to European forests ranges between 0.3 and 6.6 kg N ha$^{-1}$ yr$^{-1}$ with a median of 3.0 kg N ha$^{-1}$ yr$^{-1}$ and the gaseous dry deposition (NH$_3$, NO$_2$ and HNO$_3$) ranges between 1.2 and 25.8 kg N ha$^{-1}$ yr$^{-1}$. On average, gaseous dry deposition was a factor 1.5 to 4.5 larger than aerosol deposition (median 2.6). The most challenging finding by Flechard et al. (2011) however might be that these estimates appear rather sensitive to deposition model assumptions, as can be seen in Fig. 3.
The four deposition models used to quantify dry deposition based on mean concentration measurements were: CBED used in the United Kingdom; CDRY used by Environment Canada; EMEP used in Europe under the Convention on Long Range Transboundary Air Pollution (see also Tuovinen et al., this volume); and IDEM used in The Netherlands. All four models use a surface-vegetation-atmosphere transfer scheme that quantifies a deposition velocity $v_d$ for each gaseous and particulate component of dry deposition. The deposition rate is then computed as $v_d$ times the mean concentration. Differences in the models to quantify $v_d$ hence reflect differences in degree of complexity in how the canopy is modelled, and differences in how atmospheric conditions (turbulence, stability) are treated by the model and how stomatal uptake is computed (see also Cieslik, this volume).

A principal component analysis of total N dry deposition and all individual components (NH$_3$, HNO$_3$, NO$_2$, NH$_4^+$ and NO$_3^-$) obtained from the four deposition models is shown in Fig. 4. Despite the large differences between models in their relative contributions to total N dry deposition seen in Fig. 3 the PCA in Fig. 4 clearly reveals that in absolute numbers the model differences are not that important: 96.7% of total variance in the dry deposition estimates for forest ecosystems in Europe can be explained by the first principal component. Total N deposition from the CBED model from the UK (Smith et al. 2000) almost perfectly represents the first principal component (Fig. 4). This means that the CBED total is the best representation for the whole set of deposition estimates. It is more strongly influenced by NH$_3$ deposition than the other three models and assumes much lower NO$_2$ deposition rates than the Dutch IDEM model (Bleeker et al., 2004; Erisman et al., 1994) and – even more pronounced – the Canadian CDRY model (Zhang et al., 2003). Despite these functional differences, the overall statistical average N dry deposition as a function of altitude (Fig. 5) is surprisingly similar for the CBED and IDEM models. Deposition rates are however substantially larger than what EMEP and CDRY predict from exactly the same measured concentration data obtained from low-cost Delta denuder systems.

Figures 4 and 5 about here

4. Recent developments to assess effect on tree growth

At sites with long-term EC flux measurements the net CO$_2$ flux can be directly related to N deposition. For other sites where EC flux measurements are not feasible, some progress has recently been made to use point-dendrometer data with high temporal resolution to monitor growth rates over the season, which in the long term should provide similarly important data to link N deposition to tree growth.

Stem radius variations of trees are a measure for biomass accumulation at the monthly, seasonal and annual time scale (Zweifel et al. 2010). Figure 6 shows the concept of how tree growth can be quantified from these measurements: on the annual time scale the time traces...
of continuous high-resolution stem diameter measurements (Fig. 6a) and independent cumulative net CO$_2$ flux measurements performed by EC (Fig. 6b) are strikingly similar.

**4.1 Dendrometric measurements**

The extra noise seen in the dendrometer data reflects the diurnal water cycle in the tree (see Zweifel et al. 2010) which is unrelated to tree growth at short (daily, weekly) timescales, whereas the signal at longer timescales is related to environmental factors, mostly temperature trend and water availability, including bark and the degradation of dead phloem cells, in turn affecting plant water balance and plant growth (see also Zweifel and Häsl er, 2000, 2001; Cocozza et al., 2012). An important additional insight into tree growth is shown during the winter where the natural shrinking of the bark is similar to the respiration losses measured by EC outside the growing season. This signal adds value on top of conventional simple tree-ring width measurements which only reflect growing-season conditions.

**4.2 Net ecosystem productivity measurements**

The CO$_2$ sources (respiration) and CO$_2$ sinks (assimilation) are large budget components of opposite directions that sum to a relatively small NEP over a year (Barford et al., 2001; Körner et al., 2005; Zweifel et al., 2010). As the net budget of these components is directly measured with EC, it is generally assumed that the accuracy of the annual estimates of NEP should be better than the accuracy of the difference between the estimates of the two large components gross primary production (GPP) and total ecosystem respiration (TER) determined separately (Buchmann and Schulze, 1999).

By installing dendrometers at EC research sites within Europe, we envisage gaining dendrometric growth data of various tree species under different climates. These will then be related to NEP measured by EC as a proof of concept that NEP at annual and possibly even monthly resolution (during the growing season) could be obtained from representative dendrometer measurements.

**5. Policy Relevance of the Knowledge on Nitrogen Deposition**

The reduction of N deposition is a matter of the UNECE Convention on Long Range Transboundary Air Pollution (CLRTAP), which came into force in 1979. The protocol on reduction of nitrous oxides (1988, Sofia) was the first based on the effect based approach, in that it used the critical loads to calculate the required deposition reduction to reach ecosystem protection. The Gothenburg protocol (UNECE 1999), the “Protocol to Abate Acidification, Eutrophication and Ground-level Ozone”, is also effect based and regards also the interactions between acidification, eutrophication and ozone. Since 2008 the Task Force on reactive N under the Working Group on Strategies and Review of the CLRTAP is mandated
to develop a better understanding of the integrated, multi-pollutant nature of reactive N. The main aim is the development of integrated solutions for N reductions. This aim requires adequate research networks, and the development of supersites for integrated forest ecosystem research (Mikkelsen et al., this volume) is an important step in this direction.

In Switzerland the effects of N are monitored at 9 Level II sites of ICP Forests and on cantonal monitoring sites (IAP). At more than 15 of these sites, selected to form an N and water deficit gradient (TreeNet-CH1), the long-term net ecosystem productivity (NEP) is measured with point dendrometers. At two of these sites NEP is measured also with the eddy covariance method (EC). An initial study by Zweifel et al. (2010) has shown excellent agreement between EC-derived NEP and tree diameter increments (Section 4). This unique network started in 2011 and has since then delivered high-resolution dendrometer data and ancillary meteorological data. It is envisaged to combine the TreeNet-CH network with the Swiss Fluxnet network (Lägeren and Davos) and the regular forest inventories (National Forest Inventory, NFI) to be able to compile quarterly if not even monthly interpolated maps of actual tree growth throughout a growing season. This would allow us to see effects of N inputs in combination with climate anomalies in due time. At the annual time scale such information is expected to provide a more robust basis to estimate the forest CO2 budget, as is expected in the national inventory reporting under the Kyoto Protocol. This will allow estimating the mitigation potential of forests and its relationship with N inputs and other environmental factors.

Acknowledgments

We thank Dr. Sabine Augustin from the Swiss Federal Office for the Environment for her valuable contribution to this chapter, in particular to the policy relevance aspects. The authors were supported by funds received from the Swiss Staatssekretariat für Bildung und Forschung (contract SBF Nr. C10.0101) to participate in COST Action FP0903 “Climate Change and Forest Mitigation and Adaptation in a Polluted Environment”, and by the Swiss National Science Foundation (grant PDFMP3_132562). We also thank the Swiss Federal Office for the Environment for funding the Swiss TreeNet network, and Dr. Roman Zweifel for scientific collaboration and technical support with dendrometer measurements.

1 http://www.treenet.info
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Figures

Fig. 1: Schematic showing the similarities and differences between nutrient and pollutant inputs to ecosystems by rainfall (left) and by fog deposition (right). Reprinted from Eugster (2008).
Fig. 2: Delta denuder system for sampling gaseous NH$_3$, NO$_2$, HNO$_3$, HONO, SO$_2$, HCl and aerosol NH$_4^+$, NO$_3^-$, SO$_4^{2-}$, Cl$^-$, Na$^+$, Ca$^{2+}$, Mg$^{2+}$ (a) on site operational at Lägeren mixed forest in Switzerland; (b) denuder sample train which is exchanged every month; (c) interior of housing with door open. Numbers in (c) are: 1, durable casing; 2, sampling inlet (funnel); 3,4, sampling train; 5,7, tube connection; 6, gas volume flow meter; 8, in-line large particle filter; 9, air pump with heater; 10–13, electrical installation. Modified after http://www.nitroeurope.eu/sites/nitroeurope.eu/files/neu_data/Dissemination-deliverables/NEU_delta_ws_tfmm_Helsinki_may2006_presentation.pdf
Fig. 3: Relative contributions of reactive N species to total inorganic N dry deposition to 29 European forest ecosystems (F) in comparison with 9 semi-natural grasslands (SN), 8 cut or grazed grasslands (G), and 9 croplands (C). Deposition was calculated with measured monthly mean concentrations of NH$_3$, HNO$_3$, NO$_2$, NH$_4^+$ and NO$_3^-$ using four different deposition models (DBED, CDRY, EMEP, IDEM). Location names start with an abbreviation for the country followed by a 3-character location acronym. Reprinted from Flechard et al. (2011).
Fig. 4: Principal component analysis (covariance-based PCA) of dry deposition estimates for 29 European forest ecosystems based on data published by Flechard et al. (2011). Deposition was calculated with measured monthly mean concentrations of NH$_3$, HNO$_3$, NO$_2$, NH$_4^+$ and NO$_3^-$ using four different deposition models (DBED, CDRY, EMEP, IDEM). The graph shows the first two principal components explaining 98.4% of total variance (96.7% and 1.7% for the first and second component, respectively). This indicates that total deposition (bold vectors) strongly depends on assumptions made by the deposition model. With the British CBED gaseous deposition via NH$_3$ is by far the most relevant dry deposition pathway, whereas the Canadian CDRY model more strongly emphasizes NO$_2$ dry deposition. Unnamed grey arrows represent all other deposition pathways that do not lead to substantial contributions to total dry deposition by any of the models used by Flechard et al. (2011).
**Fig. 5:** Altitude dependence of dry deposition (log scale) of total inorganic N to 29 European forest ecosystems determined from four deposition models that use measured monthly mean concentrations. All models show a local maximum for sites with an altitude around 800 m a.s.l., indicating the specific relevance of N deposition to montane forest ecosystems. Data were taken from Flechard et al. (2011).
Fig. 6: The time traces of dendrometer measurements DR on single trees (a) and of ecosystem-scale eddy covariance measurements of net ecosystem productivity NEP (b) are very similar in a subalpine Norway spruce forest at Davos, Switzerland. Both panels show the full year 2003 with the last months of 2002 and the first months of 2004. Using such relationships it should become possible to see effects of pollutants and nitrogen deposition at the whole-tree scale and at time resolutions well below the annual resolution of tree ring widths that have been used so far. Reprinted from Zweifel et al. (2010). Current research focuses on how the relationship between DR (a) and NEP (b) holds for other forest types in different climatic zones of Europe.