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# Governing processes for reactive nitrogen compounds in the atmosphere in relation to ecosystem, climatic and human health impacts

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Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

⏪

⏩

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion

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**BGD**

9, 9349–9423, 2012

---

**Governing  
processes for  
reactive nitrogen  
compounds**

O. Hertel et al.

---

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

⏪

⏩

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion

## Abstract

Reactive nitrogen ( $N_r$ ) compounds have different fates in the atmosphere due to differences in governing processes of physical transport, deposition and chemical transformation.  $N_r$  compounds addressed here include reduced nitrogen ( $NH_x$ : ammonia ( $NH_3$ ) and its reaction product ammonium ( $NH_4^+$ )), oxidized nitrogen ( $NO_y$ : nitrogen monoxide (NO) + nitrogen dioxide ( $NO_2$ ) and their reaction products) as well as organic nitrogen compounds (organic N). Pollution abatement strategies need to take into account these differences in the governing processes of these compounds when assessing their impact on ecosystem services, biodiversity, human health and climate.  $NO_x$  (NO +  $NO_2$ ) emitted from traffic affects human health in urban areas where the presence of buildings increases the residence time in streets. In urban areas this leads to enhanced exposure of the population to  $NO_x$  concentrations.  $NO_x$  emissions have little impact on nearby ecosystems because of the small dry deposition rates of  $NO_x$ . These compounds need to be converted into nitric acid ( $HNO_3$ ) before removal through deposition is efficient.  $HNO_3$  sticks quickly to any surface and is thereby either dry deposited or incorporated into aerosols as nitrate ( $NO_3^-$ ). In contrast to  $NO_x$  compounds,  $NH_3$  has potentially high impacts on ecosystems near the main agricultural sources of  $NH_3$  because of its large ground-level concentrations along with large dry deposition rates. Aerosol phase  $NH_4^+$  and  $NO_3^-$  contribute significantly to background  $PM_{2.5}$  and  $PM_{10}$  (mass of aerosols with a diameter of less than 2.5 and 10  $\mu m$ , respectively) with an impact on radiation balance as well as potentially on human health. Little is known quantitatively and qualitatively about organic N in the atmosphere, other than that it contributes a significant fraction of wet-deposited N, and is present in both gaseous and particulate forms in the atmosphere. Further studies are needed to characterize the sources, air chemistry and removal rates of organic N emissions.

**BGD**

9, 9349–9423, 2012

### Governing processes for reactive nitrogen compounds

O. Hertel et al.

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures



Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion

## 1 Introduction

Reactive nitrogen compounds ( $N_r$ ) affect ecosystem health (Sutton et al., 2011), human health (Aneja et al., 2009) and contribute to climate change (Butterbach-Bah et al., 2011; Erisman et al., 2011; Xu and Penner, 2012). However, the actual impact of specific emissions depends on a cascade of competing processes taking place simultaneously, which transforms pollutants along multiple paths, with multiple impacts (Galloway et al., 2003). One example is the release of  $NH_3$ . Strong  $NH_3$  sources contribute to high N loads to nearby ecosystems through a fast dry deposition rate to nearby surfaces (Hertel et al., 2006). This process competes with reactions that lead to the formation of aerosol phase  $NH_4^+$ . The latter contributes to  $PM_{2.5}$  and  $PM_{10}$  (mass of aerosols with an aerodynamic diameter less than 2.5 and 10  $\mu m$ , respectively) with much longer transport distance and with potential health effects and impact on the radiation balance. The  $NH_4^+$ -containing aerosols have low dry deposition velocities, can be transported up to 1000 km away and are mainly wet scavenged. Figure 1 provides an overview of the  $N_r$  pathways, with an emphasis on atmospheric N deposition.  $NH_3$  is emitted mainly from agricultural sources (Bouwman et al., 1997), and has a typical atmospheric residence time of 24 h. Both  $NH_3$  and gaseous  $NO_x$  lead to formation of aerosol phase compounds ( $NH_4^+$  and  $NO_3^-$ , respectively) which have atmospheric residence times of several days. Furthermore  $NH_3$  can take part in bi-directional surface fluxes (Sutton et al., 1995); in areas with large nitrogen (N) inputs,  $NH_3$  may be released from plants and soil, and in areas with high gas-phase concentrations, deposition rates may be limited by surface saturation, and re-emission of absorbed  $NH_3$  may occur (Flechard et al., 1999; Massad et al., 2010b; Nemitz et al., 2004a; Sutton et al., 2009a, b). By comparison, the primary form in which oxidized nitrogen ( $NO_y$ ) is emitted is nitrogen monoxide (NO), which reacts to form nitrogen dioxide ( $NO_2$ ) over minutes and hours depending on local oxidant chemistry (interactions are described well in various text books see e.g. Seinfeld and Pandis, 2006). Both of these compounds (especially NO) are removed slowly by dry deposition to vegetation. The main

**BGD**

9, 9349–9423, 2012

### Governing processes for reactive nitrogen compounds

O. Hertel et al.

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

⏪

⏩

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion

scavenging process is therefore by chemical conversion. In the case of  $\text{NO}_2$ , chemical reaction with hydroxyl (OH) radical forms nitric acid ( $\text{HNO}_3$ ) with a typical transformation rate of 5% per hour (Hertel, 1995).  $\text{HNO}_3$  has a short atmospheric lifetime (on the order of seconds to hours), since it is quickly scavenged by uptake in aerosols, reaction with  $\text{NH}_3$  or by dry deposition (it typically sticks to any surface – aerosol as well as vegetation and the ground) (Hertel et al., 2006). A US study (Day et al., 2008) found a lifetime of  $\text{HNO}_3$  of 2.5 h for a mountain area in a sunny and warm climate, but the lifetime depends strongly on the environment and another study has shown lifetimes of 7–20 min in air masses processed by clouds containing ice-crystals (von Kuhlmann and Lawrence, 2006). Uptake of  $\text{HNO}_3$  into existing aerosols or formation of new aerosols by reaction with  $\text{NH}_3$  leads to aerosol bound nitrate ( $\text{NO}_3^-$ ) as does the reaction with sea-salt (Pryor and Sorensen, 2000). Such ammonium ( $\text{NH}_4^+$ ) and  $\text{NO}_3^-$  containing aerosols are mainly scavenged by wet deposition, due to the relatively small dry deposition rate of submicron aerosols (Gallagher et al., 2002). Aerosol phase  $\text{NH}_4^+$  and  $\text{NO}_3^-$  may under certain circumstances – depending on humidity and temperature – be released back to gas phase  $\text{NH}_3$  and  $\text{HNO}_3$ . The greenhouse gas nitrous oxide ( $\text{N}_2\text{O}$ ) is excluded in the figure, as it has little impact on atmospheric N deposition. To maintain simplicity of the diagram, reservoir compounds (e.g. HONO,  $\text{HO}_2\text{NO}_2$ , PAN and PAN-like species) are also disregarded in Fig. 1, although these are discussed in the subsequent sections. The actual split between various pathways depends on the precise location where local climate conditions, local emission densities and distance from high emission areas all determine the overall fate of  $\text{N}_r$ . Another example of atmospheric processes affecting the impact of  $\text{N}_r$  is the complex flow conditions in urban street canyons. The generally short residence time of air in urban streets means that only fast reactions of  $\text{NO}_x$  are relevant. However, the presence of buildings, especially in streets with close building facades on both sides of the street, increases the local residence time of traffic pollutants emitted in the street sufficiently to significantly increase also the exposure of the population (Hertel and Goodsite, 2009). These processes – the flow conditions and the chemical conversion – affect the ratio between NO

---

## Governing processes for reactive nitrogen compounds

O. Hertel et al.

---

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

⏪

⏩

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion



which has little direct impact on human health and the airway irritant NO<sub>2</sub> that e.g. may enhance impact on airborne allergens (Tunncliffe et al., 1994). As can be seen from this short introduction, the atmospheric chemistry and governing processes removing N<sub>r</sub> from the atmosphere is both complex and diverse, reflecting the many compounds involved and their different characteristics. To be able to relate these processes more effectively, the present paper provides an overview of the governing atmospheric processes of N<sub>r</sub> in the atmosphere, with particular attention to the nature of the emissions and removal processes, including bi-directional exchange where this occurs. This review represents further develops and an extensive analysis of this topic (Hertel et al., 2011), as originally conceived in the European Nitrogen Assessment (2011). While the paper focuses on Europe, the general principles are applicable globally. For the purpose of this review, we have disregarded N<sub>2</sub>O as it has no health effects at ambient concentration levels and an insignificant role in atmospheric deposition of N. It should be noted though, that it does play a role in the chemistry of the stratosphere and as a climate forcer (Pinder et al., 2012), as well as in leading to depletion of stratospheric ozone (Brink et al., 2011; Butterbach-Bah et al., 2011).

## 2 Emissions

N<sub>r</sub> is mainly released to the atmosphere as NH<sub>3</sub>, NO<sub>x</sub>, and in poorly quantified amounts of organic N such as amines. By comparison, emission of N<sub>2</sub>O accounts for a much smaller fraction, representing about 15% of the total European N<sub>r</sub> emission to the atmosphere (Leip et al., 2011).

### 2.1 Ammonia emissions

This section is devoted to the ammonia emissions and contains sub-sections regarding: Animal houses and manure storages, Manure application and mineral fertilizer, Grazing animals, Non-agricultural sources, Spatial distribution, and Long term trends.

**BGD**

9, 9349–9423, 2012

## Governing processes for reactive nitrogen compounds

O. Hertel et al.

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

⏪

⏩

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion



In western countries, agricultural activities contribute 85 %–98 % of the atmospheric NH<sub>3</sub> emissions (Anderson et al., 2003; Gyldenkærne et al., 2005; Sutton et al., 2000a). These emissions arise in some cases from the direct breakdown of fertilizer, but more importantly from the breakdown of organic matter (like urea) within animal wastes.

5 The emission process is a chemical and physical process which is highly temperature dependent and varies significantly over day and season (Gyldenkærne et al., 2005; Skjøth et al., 2004). The agricultural sources may be grouped as:

- point sources i.e. animal houses, manure storage and “slurry lakes” (Sommer et al., 2006);
- 10 – application of manure and mineral fertilizer to fields (Smith et al., 2009);
- grazing animals (Petersen et al., 1998);
- other sources including releases from plants (Larsson et al., 1998) such as legumes (Gyldenkærne et al., 2005) and vegetation during management and senescence (Sutton et al., 2000b, 2009a, b).

15 And the non-agricultural sources include:

- wild animals (Anderson et al., 2003; Riddick et al., 2012; Simpson et al., 1999; Sutton et al., 2000a; Theobald et al., 2006);
- catalyst processes, mainly related to road traffic (Kean et al., 2009; van Vuuren et al., 2011);
- 20 – manufacturing processes such as production of fertilizer, glass wool, catalysts and cement;
- humans, pets and sewage systems (Reche et al., 2012; Sutton et al., 2000a);
- other sources such as landfill and non-anthropogenic sources including natural fires from ecosystems (Andreae and Merlet, 2001; Yokelson et al., 2007);

---

**Governing  
processes for  
reactive nitrogen  
compounds**

O. Hertel et al.

---

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures



Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion



- emission from plants due to compensation points (Farquhar et al., 1980; Husted et al., 2000; Massad et al., 2010a; Mattsson et al., 2009; Sutton et al., 1995);
- emission from sea surfaces (Barrett, 1998; Sørensen et al., 2003).

The above split of the agricultural sources has been used in a parameterization currently considered as the most advanced description of temporal and spatial variations in NH<sub>3</sub> emissions (Pinder et al., 2007). This parameterization was originally developed for Denmark (Gyldenkerne et al., 2005; Skjøth et al., 2004), later extended to Northern Europe (Skjøth et al., 2011), and is currently being further developed in an open source version within the FP7 project ECLAIRE (<http://www.eclair-fp7.eu>). This parameterization and other more simple approaches have been successfully implemented in the Atmospheric Chemistry and Deposition (ACDEP) model (Skjøth et al., 2004), the DEHM model (Brandt et al, 2012, Skjøth et al, 2011), the EMEP MSC-W model (Fagerli et al., 2007; Simpson et al., 2012), the local scale models OML-DEP (Geels et al., 2012; Sommer et al., 2009), the SCAIL (Theobald et al., 2009) screening method developed for the UK, as well as most recently the local scale models AIRMOD and ADMS (Theobald, 2012). We here use the mechanistic description in this model as the basis for describing NH<sub>3</sub> emissions.

### 2.1.1 Animal houses and manure storages

Highly complex surface models have been developed for describing NH<sub>3</sub> emissions from agricultural buildings (Muck and Steenhuis, 1982; Olesen and Sommer, 1993; Oudendag and Luesink, 1998; Zhang et al., 1994) in order to help better manage emissions. However, in chemistry-transport modelling (CTM) a simplified parameterization based on wind speed and temperature has proven more practical (Skjøth et al., 2004):

$$E(t) = C \times T(t)^{0.89} \times V(t)^{0.26} \quad (1)$$

**Governing processes for reactive nitrogen compounds**

O. Hertel et al.

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures



Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion





where  $C$  is a constant related to the amount of N or TAN (Total Ammoniacal nitrogen = sum of  $\text{NH}_3$  and  $\text{NH}_4^+$ ) in the manure at a given time and location,  $T(t)$  is the temperature as function of time, and  $V$  is the wind speed or the rate of ventilation. Formula (1) is used to distribute a known annual emission into shorter time steps. In Northern Europe, pig and poultry sheds are heated in winter and have reduced ventilation (Fig. 2). The annual emissions from manure storage vary with type of storage, and temporal variation follows ambient temperature (Gyldenkærne et al., 2005; Skjøth et al., 2004). In warm areas and during warm periods, emissions from buildings reflect outdoor temperature (Seedorf et al., 1998). Buildings with pigs and poultry have significant emissions also in cold periods, due to their warmer indoor temperatures, when outdoor storages and naturally ventilated cattle barns have low emission rates. Based on these principles and ambient temperature, it is possible to simulate temporal variations in emissions from manure storage, pig, poultry and cattle sheds (Fig. 3).

### 2.1.2 Manure application and mineral fertilizer

$\text{NH}_3$  emissions from field application of manure and mineral fertilizer take place at distinct times of year and with relatively short duration compared with animal related point sources. Soil type (Loubet et al., 2010) and the application method are crucial for establishing the magnitude of the emission. Broad-spread application over the whole of the ground surface leads to very high emissions, whereas direct soil injection or band-spreading methods strongly reduce emissions (Skjøth et al., 2008). National regulations can significantly affect seasonality when these affect application methods (Skjøth et al., 2008) although these emissions have substantial uncertainties (Sintermann et al., 2012). In many countries manure application is constrained by almost no regulation, whereas in most Northern European countries, such application is banned during winter to limit nutrient wash out, especially under the EU Nitrate Directive. To overcome shortage in storage capacity, farmers in Northern Europe often empty their tanks in autumn to allow them to handle emissions from animals kept indoors throughout the winter. This practice leads to late autumn emission peaks in  $\text{NH}_3$  emission. Also for

**BGD**

9, 9349–9423, 2012

## Governing processes for reactive nitrogen compounds

O. Hertel et al.

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

⏪

⏩

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion



mineral fertilizer there are regional differences, as farmers in Southern Europe start fertilizer application earlier than Northern European farmers.

### 2.1.3 Grazing animals

The time that animals are in the field, together with the N content in the grass, govern the emission from grazing animals. Animals that are feeding on grass with a high N content excrete large amounts of N (as TAN) compared with animals on grassland with less nitrogen rich grass (Petersen et al., 1998). However, urine from grazing animals quickly enters the soil (Webb et al., 2005) which reduces the emission in comparison e.g. to surface applied slurry. The larger dry matter content of slurries limits infiltration into the soil, which is the main reason for the larger emissions compared with urine from grazing animals. In Southern Europe, animals in general are in the field most of the year. In Northern Europe, only sheep stay in the field most of the time. In most countries cattle are inside sheds approximately half of the year (Skjøth et al., 2011). In principle temperature should increase  $\text{NH}_3$  emission related to grazing animals or outdoor yards used by livestock, however, clear temperature responses have not always been seen in practise (Misselbrook et al., 2001). However studies suggest that emissions from the urine fraction are larger during summer than during autumn (Ryden et al., 1987).

### 2.1.4 Non-agricultural sources

Non-agricultural sources of  $\text{NH}_3$  are generally poorly described, which is largely a consequence of the limited number of data on many different processes. These sources include: human sweat, excreta from pets and wild animals, exhaust from gasoline vehicles with catalytic converters, stationary combustion sources and industry, as well as evaporation from waste deposits (Sutton et al., 2000a). Elevated  $\text{NH}_3$  levels have been reported near roads and in urban areas as a consequence of  $\text{NH}_3$  emissions from vehicles fitted with catalytic converters (Cape et al., 2004). Evaporation from  $\text{NH}_4^+$  containing salts is a possible source of  $\text{NH}_3$  in Southern and Central European cities

**BGD**

9, 9349–9423, 2012

## Governing processes for reactive nitrogen compounds

O. Hertel et al.

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

⏪

⏩

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion



where strongly elevated levels have been found: Barcelona (Reche et al., 2012), Rome (Perrino et al., 2002) and Rijeka (Croatia) (Alebic-Juretic, 2008). Perhaps the best described non-agricultural sources of NH<sub>3</sub> are from colonies of wild seabirds (Blackall et al., 2007; Riddick et al., 2012; Wilson et al., 2004) and seals (Theobald et al., 2006). Although these contribute to a small fraction of global emissions, they occur in otherwise clean remote environments, representing major point sources under purely climatic control for the given animal densities.

### 2.1.5 Spatial distribution

EMEP (the Co-operative Programme for Monitoring and Evaluation of the Long-range Transmission of Air Pollutants in Europe: <http://ww.emep.int>) and CORINAIR (CORe Inventory AIR emissions) have compiled inventories at the European level on annual mean emissions with a spatial resolution of 50 km × 50 km (Tørseth et al., 2012). EDGAR (Emissions Database for Global Atmospheric Research, <http://edgar.jrc.ec.europa.eu>) and GEIA (Global Emissions Inventory Activity, <http://geiacenter.org>) databases are available on 0.1 × 0.1° resolutions, and EUROTRAC (EUREKA project on the transport and chemical transformation of trace constituents in the troposphere over Europe) GENEMIS (GENeration and Evaluation of eMISSion data; <http://www.gsf.de/eurotrac>) compiled emission inventories for the year 1994 with a grid resolution of 16.67 km × 16.67 km (Schwarz et al., 2000; Wickert et al., 2001). The GENEMIS data have in some cases been used to redistribute EMEP emission to higher resolution also for subsequent years assuming unchanged relative distributions (Hertel et al., 2002). The need for high resolution inventories has been recognized by the model groups within MACC and MACC-II research programmes (<http://www.gmes-atmosphere.eu/>). They have therefore adapted a high resolution inventory with 7 km resolution of annual emissions at SNAP level-1 (Pouliot et al., 2012). The TNO MACC-II inventory is a combination of the officially reported inventories, information on geographic location of point sources and a correction procedure for inconsistencies. Figure 4 shows the spatial distribution of the EMEP and EDGAR emission data sets. The EDGAR data set has

**BGD**

9, 9349–9423, 2012

## Governing processes for reactive nitrogen compounds

O. Hertel et al.

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

⏪

⏩

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion



---

## Governing processes for reactive nitrogen compounds

O. Hertel et al.

---

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

⏪

⏩

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion

a considerably higher spatial resolution compared to the EMEP data set. The EDGAR data set is constructed centrally using a uniform methodology and uniform data set, while the EMEP data set is a compilation of national inventories that originates from national experts. One result is that the two inventories do not agree with respect to amount of emissions within national boundaries. Such differences can be substantial (de Vries et al., 2011a; de Vries et al., 2011b). Here these differences are clearly seen in Fig. 4, with considerably higher emissions in central Europe in the EDGAR data set compared to the EMEP data set. Emissions are the most important input to all CTMs (Simpson et al., 2011), but for more than a decade been considered among the largest uncertainties in these models (Reis et al., 2009). The differences between the different emission data sets can therefore be expected to have a direct impact on model results for both process descriptions and impact assessments. Whereas emissions are usually reported as annual totals, there is considerable variation on hourly, daily and seasonal time-scales which is represented poorly (if at all) in the data used for CTMs (Menut and Bessagnet, 2010; Pouliot et al., 2012).

### 2.1.6 Long term trends

NH<sub>3</sub> emissions have been reduced in countries like Denmark, Germany and the Netherlands, whereas for France, Sweden and Norway only minor changes have occurred over the past 15 yr, and in countries like China and the US emissions have increased (Fig. 5). The uncertainties in these trends, including the small response of NH<sub>3</sub> concentrations to reduced animal numbers in central Europe (Horvath and Sutton, 1998) and to emission abatement policies in the Netherlands and Denmark (Bjerrgaard, 2011; Bleeker and Sutton, 2006; Erisman et al., 1998; Skjøth et al., 2008; Sutton et al., 2003) have been a major topic of debate. The recent evidence (Bleeker et al., 2009; Horvath et al., 2009) indicates that increasing NH<sub>3</sub> concentrations in Eastern Europe were due to a reduction in SO<sub>2</sub> emissions (resulting from longer atmospheric lifetime of NH<sub>3</sub>) a factor which also influenced the NH<sub>3</sub> trends in the Netherlands,

Denmark and elsewhere. The observed trends in  $\text{NH}_3$  concentrations are therefore a result of changes in  $\text{NH}_3$  emissions as well as changes in atmospheric chemistry.

## 2.2 Nitrogen oxide emissions

This section is devoted to the nitrogen oxides emissions and contains sub-sections regarding: spatial distribution, temporal variation, trends and projections.

$\text{NO}_x$  is generated at high temperatures in combustion processes mainly from oxidation of free atmospheric nitrogen ( $\text{N}_2$ ). Road transport and public power generation are by far the largest contributors to the emissions – for Europe these sources each contribute 20 % to 25 % of total  $\text{NO}_x$  emissions (Vestreng et al., 2008). Industrial production, other mobile sources and shipping each contribute 10 % to 15 %, whereas smaller contributions are related to aviation, and biomass and agricultural burning (2 % to 3 %). Emissions from forest and agricultural soils are discussed in Sect. 2.3. Emissions from waste incineration and lightning can probably be neglected (< 1 %).

### 2.2.1 Spatial distribution

Annual inventories of  $\text{NO}_x$  emissions are available from similar sources as for  $\text{NH}_3$  (Sect. 2), which includes EMEP, EDGAR, EUROTRAC, GENEMIS and TNO-MACC. In the same way as for  $\text{NH}_3$ , the differences between the EDGAR and EMEP  $\text{NO}_x$  inventories can be seen in Fig. 6. The major road networks and urban areas are obvious hotspots and therefore easily visible. The same is true for shipping, where main transport routes are easily detectable and emissions may overall be higher than in urban areas. As a result of European regulation, power plants and industry release pollutants at a height that leaves local urban areas almost unaffected by these emissions, although they naturally contribute to pollutant levels further downwind.

**BGD**

9, 9349–9423, 2012

## Governing processes for reactive nitrogen compounds

O. Hertel et al.

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

⏪

⏩

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion

## 2.2.2 Temporal variation

Road traffic follows very regular temporal patterns. In Northern Europe the typical urban road traffic pattern include two rush hour periods – in the morning and afternoon. Often the morning rush hour is more pronounced than the afternoon rush hour, and this is reflected in emissions and resulting concentrations of NO<sub>x</sub>, as e.g. reported for Denmark (Hertel et al., 2008). In some parts of Southern Europe there may even be four rush hours as people are off work in the middle of the day. Furthermore, different patterns are observed for weekday vs. weekend, and the pattern also differs for passenger cars and commercial vehicles. Furthermore, highways are used for transport over longer distance and these have thus different diurnal traffic pattern compared with urban streets. Seasonal variations are less pronounced compared with diurnal variations, although increased power generation for heating, and increased direct fuel use lead to increased NO<sub>x</sub> emissions in winter in Northern Europe, whereas increased demand for air conditioning in summer is expected to lead to increased NO<sub>x</sub> emissions in summer in Southern Europe.

## 2.2.3 Trends and projections

European NO<sub>x</sub> emissions have followed a steady downward trend over the time period 1990 to 2005. In total, a reduction of 34% has been seen for the 27 Member States of the European Union (EU27) as a result of regulation of emissions – like the EC Large Combustion Plant Directive and the EURO regulations for road traffic vehicles (Fig. 5). Over the next decades further reductions will result from stricter regulation of road traffic despite increasing vehicle numbers and increasing distance travelled (Vestreng et al., 2008). Phasing out of nuclear energy may, however, lead to additional coal burning with the result of increasing NO<sub>x</sub> emissions from electricity generation. International shipping has been steadily increasing, as have the associated emissions, but reductions will result from implementation of the HELCOM convention that includes NO<sub>x</sub> Emission Control Areas (NECA) ([http://www.helcom.fi/Recommendations/en\\_GB/](http://www.helcom.fi/Recommendations/en_GB/))

**BGD**

9, 9349–9423, 2012

### Governing processes for reactive nitrogen compounds

O. Hertel et al.

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

⏪

⏩

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion



rec28E\_13/) for the Baltic Sea and the North Sea. In urban streets, NO<sub>2</sub> has not followed the decreasing trend observed for NO<sub>x</sub>, (Carslaw et al., 2007; Lambrecht, 2007). The explanation is that the NO<sub>2</sub> to NO<sub>x</sub> ratio in road traffic emissions has changed substantially in recent years as a result of late diesel technology vehicles and catalytic converters in gasoline driven vehicles that reduce the overall NO<sub>x</sub> emission but with a higher NO<sub>2</sub> fraction in the remaining emission (Carslaw and Beevers, 2005). EU limit values for NO<sub>2</sub> are therefore currently exceeded in many European cities (Hertel and Goodsite, 2009), and this will persist until stricter EURO 6 norms are implemented from 2015 ([http://eur-lex.europa.eu/LexUriServ/LexUriServ.do?uri=\\_CELEX:32007R0715:EN:NOT](http://eur-lex.europa.eu/LexUriServ/LexUriServ.do?uri=_CELEX:32007R0715:EN:NOT)).

### 2.3 Emissions of NO from ecosystems and soil

In a global perspective, NO<sub>x</sub> emissions from soil contribute > 40 % (Davidson and Kinglerlee, 1997) (IGAC Newsletter Dec 2000), and > 10 % for some European countries (Butterbach-Bahl et al., 2004; Skiba et al., 1997; Stohl et al., 1996). Estimates of emissions of NO<sub>x</sub> from agricultural soil varies by more than a factor of two and are mainly related to grazing animals and fertilizer (de Vries et al., 2011c). Emissions that are related to application of fertilizer are estimated to account for 40 % of the soil emissions (IGAC Newsletter, 2000) and up to 65 % for the USA (Hall et al., 1996). Rural agricultural areas receiving N fertilizers in countries with long dry periods are likely the largest sources of soil NO where they can be of the same order of magnitude as N<sub>2</sub>O fluxes (Loubet et al., 2011). The NitroEurope Integrated Project (Sutton et al., 2007; <http://www.nitroeuropa.eu>) and the NOFRETETE (Nitrogen oxides emissions from European Forest Ecosystems) project point to European forests being large sources of NO from soils (Pilegaard et al., 2006) presumably affected by enrichment with N from atmospheric deposition (Butterbach-Bahl et al., 2011a, b).

**BGD**

9, 9349–9423, 2012

## Governing processes for reactive nitrogen compounds

O. Hertel et al.

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

⏪

⏩

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion



### 3 Transformations

In the following a description of the most important chemical transformation processes for  $N_r$  is provided with highlights concerning where these processes play a significant role.

#### 3.1 $NH_x$ chemistry in the atmosphere

New aerosol particles are formed in gas phase reactions between  $NH_3$  and gas phase acids. However,  $NH_3$  may also deposit onto existing atmospheric particles. Whenever sulphuric acid ( $H_2SO_4$ ) is present,  $NH_3$  practically always reacts with this compound in an irreversible process forming aerosol phase ammonium bisulphate ( $NH_4HSO_4$ ) and ammonium sulphate ( $(NH_4)_2SO_4$ ). Naturally this reaction depends on the availability of  $H_2SO_4$  which is currently decreasing in concentrations in Europe. Early experiments showed that 50 % of the available  $NH_3$  was converted in about 35 min at concentration levels in those days ( $20 \mu g m^{-3} SO_2$  and  $2.7 \mu g m^{-3} NH_3$ ) (Mckay, 1971).  $H_2SO_4$  is formed from gas phase oxidation of  $SO_2$  by hydroxyl (OH) radical or from aerosol phase conversion by hydrogen peroxide ( $H_2O_2$ ) and ozone ( $O_3$ ). The latter process is pH dependent, and may be catalysed by  $NH_3$ , since uptake of  $NH_3$  increases the pH of aerosols (ApSimon et al., 1994; Junge and Ryan, 1958).

The rate of conversion of gas phase  $SO_2$  and  $NH_3$  into aerosol phase ammonium sulphate has been studied in detail in laboratory experiments (Baldwin and Golden, 1979; Gupta et al., 1995; Huntzicker et al., 1980; McMurry et al., 1983). At high RH, the limiting factor for the transformation is the molecular diffusion of  $NH_3$  to the acid particles, whereas at low RH only between 10 % and 40 % of the collisions between  $NH_3$  gas molecules and  $H_2SO_4$ -containing particles lead to reaction (Huntzicker et al., 1980; McMurry et al., 1983). For small particles, their relatively large surface area makes the diffusion process more efficient. It has, however, been shown that organic material on the surface of the particles may limit the uptake of  $NH_3$  (Daumer et al., 1992). Whereas the  $NH_3$  reaction with  $H_2SO_4$  may generally be considered irreversible,

**BGD**

9, 9349–9423, 2012

## Governing processes for reactive nitrogen compounds

O. Hertel et al.

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

⏪

⏩

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion





---

**Governing  
processes for  
reactive nitrogen  
compounds**O. Hertel et al.

---

[Title Page](#)[Abstract](#)[Introduction](#)[Conclusions](#)[References](#)[Tables](#)[Figures](#)[⏪](#)[⏩](#)[◀](#)[▶](#)[Back](#)[Close](#)[Full Screen / Esc](#)[Printer-friendly Version](#)[Interactive Discussion](#)

this is not always the case, and not the case for the reactions with other acid gases. While reaction to form ammonium sulphates is effectively not reversible when these exist in particulate form, the  $\text{NH}_3$  can be returned back to the gas phase as a result of changes in atmospheric composition e.g. during cloud processing (Bower et al., 1995, 1997; Wells et al., 1997).

The presence of  $\text{HNO}_3$  and/or hydrochloric acid (HCl) together with  $\text{NH}_3$  leads to equilibrium between these gases and their aerosol phase reaction products – ammonium nitrate ( $\text{NH}_4\text{NO}_3$ ) and ammonium chloride ( $\text{NH}_4\text{Cl}$ ). For this reaction an equilibrium product,  $k_{\text{eq}} = [\text{NH}_3][\text{HNO}_3]$  of the gas phase concentrations of  $\text{NH}_3$  and  $\text{HNO}_3$  at saturation of the air is applied. Experimental studies show that this may be expressed as a function solely of temperature and humidity (Stelson et al., 1979; Stelson and Seinfeld, 1982). Besides the reactions with  $\text{H}_2\text{SO}_4$  and  $\text{HNO}_3$ ,  $\text{NH}_3$  may also react with HCl and form  $\text{NH}_4\text{Cl}$  (Pio and Harrison, 1987; Raes et al., 2000). Whereas HCl is emitted as a primary pollutant from coal burning and waste incineration,  $\text{HNO}_3$  is the main secondary pollutant from oxidation of  $\text{NO}_x$ .

Ice core data have indicated that the HCl input from sea salt de-chlorination has been enhanced by a factor of 2–3 during the second half of the 20th century similarly to the increase of the atmospheric acidity in response to growing  $\text{NO}_x$  and  $\text{SO}_2$  anthropogenic emissions (Legrand et al., 2002). New measurement data indicate that in NW Europe, HCl concentrations are similar (in terms of mixing ratio) to those of  $\text{HNO}_3$  in summertime. However,  $\text{NH}_4\text{Cl}$  concentrations are generally much lower than  $\text{NH}_4\text{NO}_3$  concentrations. HCl is emitted from anthropogenic sources including domestic coal burning in Europe which was historically a significant source of  $\text{NH}_x$  in the atmosphere (Sutton et al., 2008), but also released in displacement reaction in sea spray particles when these take up  $\text{HNO}_3$  (Wall et al., 1988). This displacement is most likely the explanation for HCl concentrations of up to 250 pptv observed in the marine boundary layer (Harris et al., 1992). The displacement plays an important role in halogen and  $\text{O}_3$  cycling (van Glasow and Crutzen, 2007). The resultant decline in  $\text{HNO}_3$  due to the reactions with sea salt can promote the dissociation of  $\text{NH}_4\text{NO}_3$ , and this reaction has

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## Governing processes for reactive nitrogen compounds

O. Hertel et al.

---

therefore been given considerable scientific attention as e.g. in the EU projects ANICE (de Leeuw et al., 2001) and MEAD (Spokes et al., 2006). Model estimates suggest that equilibrium between  $\text{HNO}_3$  and sea salt can take hours to days to reach steady state (Erickson et al., 1999; Keene et al., 2002), similar to time scales for transport across the coastal zone. Thus reactions may not be complete within the coastal zone and  $\text{HNO}_3$  concentrations relative to  $\text{NO}_3^-$  can be highly variable depending on the history of the air mass and the history of the mixing between sea salt aerosols and polluted continental air (Fischer et al., 2006).

$\text{NO}_3^-$  aerosol is found in the coarse as well as the fine mode as a result of the displacement reaction whereas  $\text{NH}_4^+$  is present in the fine mode and only in very small amounts in the coarse mode (Yeatman et al., 2001) which affects the subsequent efficiency of deposition.

### 3.2 $\text{NO}_y$ chemistry in the atmosphere

The chemistry of  $\text{NO}_y$  and  $\text{O}_3$  in the atmosphere has been discussed in numerous reviews (see e.g. Crutzen et al., 1999; Jenkin and Clemitshaw, 2000), so only an outline is given here. A sketch showing the main chemical paths of  $\text{NO}_y$  is given in Fig. 7, and a short description including main research findings in this area is given in the following. The fraction of directly emitted  $\text{NO}_2$  from road traffic in western countries has increased in recent years. However, in the tropospheric boundary layer the distribution between  $\text{NO}$  and  $\text{NO}_2$  is governed to a large degree by  $\text{O}_3$  that reacts quickly with  $\text{NO}$  to form  $\text{NO}_2$ . In sunlight  $\text{NO}_2$  photo-dissociates (wavelengths 290 to 420 nm) to form  $\text{NO}$  and the very short-lived oxygen ( $\text{O}(^3\text{P})$ ) radical. The latter will in most cases again form  $\text{O}_3$  in reaction with free oxygen ( $\text{O}_2$ ), involving a third body (either an  $\text{N}_2$  or  $\text{O}_2$  molecule) that absorbs excess vibrational energy and thereby stabilizes the formed  $\text{O}_3$  molecule. These reactions all have time-scales of seconds to minutes. The rate of reaction between  $\text{NO}$  and  $\text{O}_3$  is temperature dependent, but has a typical value about  $4 \times 10^{-4} \text{ ppbv}^{-1} \text{ s}^{-1}$ . Under typical atmospheric boundary layer conditions, this reaction

[Title Page](#)[Abstract](#)[Introduction](#)[Conclusions](#)[References](#)[Tables](#)[Figures](#)[⏪](#)[⏩](#)[◀](#)[▶](#)[Back](#)[Close](#)[Full Screen / Esc](#)[Printer-friendly Version](#)[Interactive Discussion](#)

will either lead to the complete conversion of all the  $O_3$  to  $NO_2$ , or to the conversion of all  $NO$  to  $NO_2$  (Clapp and Jenkin, 2001). In a highly polluted atmosphere (e.g. an urban area) or close to pollution sources, the former behaviour is usually observed because, although  $O_3$  is widely distributed in the lower atmosphere, its concentration is not usually high compared with  $NO$  in the highly polluted atmosphere, and hence  $O_3$  concentrations become rapidly depleted. During daylight, the main fate of  $NO_2$  is to undergo photolysis, which is reforming  $O_3$  and  $NO$  (Dickerson et al., 1982). This photolysis reaction has a typical rate coefficient under summer conditions in the mid-afternoon at mid-altitudes of about  $7 \times 10^{-3} \text{ s}^{-1}$ . The  $NO$ - $NO_2$ - $O_3$  chemistry in urban streets may be simplified (Palmgren et al., 1996), assuming that  $O_3$  is reformed instantaneously, and thereby disregarding the formation of  $O(^3P)$ . This is done in one of the most commonly applied street pollution models – OSPM (Berkowicz, 2000; Kakosimos et al., 2010).

The OH radical initiates the oxidation of a wide range of compounds in the atmospheric boundary layer. OH interacts with peroxy radicals that are responsible for the formation of excess concentrations of photo-oxidants like  $O_3$ . In the background troposphere, carbon monoxide (CO) plays a role in this system. In the reaction with OH radical CO is oxidized to  $CO_2$ , and at the same time a hydroperoxy radical ( $HO_2$ ) is formed. Whenever NO is present, the most important atmospheric reaction of the hydroperoxy radical ( $HO_2$ ) radical is the conversion of  $NO$  to  $NO_2$ . The hydroperoxy radical is one of many peroxy radicals that take part in the conversion of  $NO$  to  $NO_2$ . Organic peroxy radicals ( $RO_2$ ) are likewise important and are mainly formed by the attack of the OH radical on the organic compounds ubiquitously present in the polluted atmosphere. These reactions follow a similar path as the CO oxidation, leading to formation of alkyl peroxy radicals (Jenkin and Clemitshaw, 2000) that subsequently convert  $NO$  to  $NO_2$  in the same way as the  $HO_2$  radical. In combustion processes at high temperatures e.g. inside the engine of a petrol or diesel-driven vehicle,  $NO$  is formed from ambient  $N_2$ . However, in very  $NO$  rich air e.g. inside the exhaust pipe of vehicles and inside emitting chimneys, a 3rd order reaction can take place between two  $NO$  molecules

## Governing processes for reactive nitrogen compounds

O. Hertel et al.

[Title Page](#)[Abstract](#)[Introduction](#)[Conclusions](#)[References](#)[Tables](#)[Figures](#)[⏪](#)[⏩](#)[◀](#)[▶](#)[Back](#)[Close](#)[Full Screen / Esc](#)[Printer-friendly Version](#)[Interactive Discussion](#)

and O<sub>2</sub> with a reaction rate coefficient of  $2.3 \times 10^{-38} \text{ cm}^6 \text{ molecules}^{-2} \text{ s}^{-1}$  (Hampson and Gavin, 1978) to give NO<sub>2</sub>. The further transformation of NO<sub>2</sub> to HNO<sub>3</sub> takes place with a typical rate of about 5 % per hour in the troposphere. The hydroxyl radical (OH) is formed in daytime in the presence of sunlight (Jenkin and Clemitshaw, 2000) The photo-dissociation of O<sub>3</sub> leads to the formation of both O(<sup>3</sup>P) and O(<sup>1</sup>D) radicals; a fraction of the latter reacts with water vapour to form two OH radicals. This reaction is in competition with O(<sup>1</sup>D)'s reaction with 3rd body O<sub>2</sub> or N<sub>2</sub> molecules to form O(<sup>3</sup>P), that in turn reforms O<sub>3</sub>. The OH radicals initiate most of the degradation of hydrocarbons in the atmosphere, a chain of reactions that e.g. lead to the formation of high O<sub>3</sub> concentrations during summer.

In the night-time the nitrate (NO<sub>3</sub>) radical has a less important but still somewhat similar role for the degradation of hydrocarbons in the atmosphere as the OH radical in daytime (for an extensive review of the chemistry, see Wayne et al., 1991). Despite the considerably lower reactivity compared with OH, its higher peak concentrations in the night time troposphere allow the NO<sub>3</sub> radical to play a major role in the transformation of organic compounds. The NO<sub>3</sub> radical is formed during night-time in reaction with NO<sub>2</sub>. Dinitrogen pentoxide (N<sub>2</sub>O<sub>5</sub>) is a reservoir compound for the NO<sub>3</sub> radical at low temperatures, but it is broken down to its precursors NO<sub>2</sub> and NO<sub>3</sub> at higher temperatures in the dark. The typical night time NO<sub>3</sub> radical concentrations in the atmospheric boundary layer are of the order 10<sup>7</sup> to 10<sup>8</sup> molecules cm<sup>-3</sup> (which is the ppt range), although both measurements and modelling suggest that values away from the surface may reach 100s of ppt (e.g. Brown et al., 2006; Riemer et al., 2009). During the daytime both NO<sub>3</sub> and N<sub>2</sub>O<sub>5</sub> photo-dissociate so fast that the concentrations of these compounds are insignificant. In the tropospheric boundary layer the photolysis of NO<sub>3</sub> radical (with a typical noon lifetime of about 5 s) follows two different wavelength dependent paths forming NO and O<sub>2</sub> (wavelength > 700 nm) or NO<sub>2</sub> and O(<sup>3</sup>P) (wavelength > 580 nm). Close to pollution sources from combustion processes e.g. road traffic or power plants, the NO<sub>3</sub> radical is quickly removed by reaction with NO leading to formation of two NO<sub>2</sub> molecules. In urban areas, emissions of NO rapidly destroy

## Governing processes for reactive nitrogen compounds

O. Hertel et al.

[Title Page](#)[Abstract](#)[Introduction](#)[Conclusions](#)[References](#)[Tables](#)[Figures](#)[⏪](#)[⏩](#)[◀](#)[▶](#)[Back](#)[Close](#)[Full Screen / Esc](#)[Printer-friendly Version](#)[Interactive Discussion](#)

---

## Governing processes for reactive nitrogen compounds

O. Hertel et al.

---

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

⏪

⏩

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion



NO<sub>3</sub> close to ground-level. However, above the surface at night NO<sub>2</sub> and O<sub>3</sub> can exist together in the absence of NO (at it has all been converted to NO<sub>2</sub>), leading to NO<sub>3</sub> formation, as observed on a tower in London (Benton et al., 2010). During night time the heterogeneous conversion of N<sub>2</sub>O<sub>5</sub> to HNO<sub>3</sub> is an important process. The lifetime of N<sub>2</sub>O<sub>5</sub> with respect to this removal is of the order of minutes in the tropospheric boundary layer. This production of HNO<sub>3</sub> may in winter be equally important as daytime conversion of NO<sub>2</sub> by OH radical. As already described, particulate nitrate (NO<sub>3</sub><sup>-</sup>) is formed when HNO<sub>3</sub> reacts with NH<sub>3</sub> and forms new aerosols, and when it sticks to existing particles in the atmosphere. In addition organic NO<sub>3</sub><sup>-</sup> may be formed from gaseous NO<sub>2</sub> on the surfaces of aerosols in other heterogeneous reactions. The NO<sub>3</sub> radical attacks alkanes by hydrogen abstraction in a similar way as the reactions of the OH radical, and again leading to formation of a peroxy radical (RO<sub>2</sub>) that may oxidise an NO molecule to NO<sub>2</sub>. Also for alkenes, the attack of the NO<sub>3</sub> radical is similar to the reactions of the OH radical; the NO<sub>3</sub> radical adds to the double bond. This reaction is followed by rapid O<sub>2</sub> addition which leads to the production of a peroxy radical. Reaction of soil emissions of NO with atmospheric OH has been suggested to provide an in-canopy source of HNO<sub>3</sub> (Farmer et al., 2006). HNO<sub>3</sub> concentrations usually peak during the day, regulated by the emissions of NO<sub>x</sub>, photochemical activity and the gas/aerosol equilibrium of NH<sub>4</sub>NO<sub>3</sub> shifting towards the gas phase with increasing temperature and decreasing relative humidity (see e.g. Andersen and Hovmand, 1994).

A reaction similar to reaction between NO<sub>2</sub> and OH forming HNO<sub>3</sub>, but less important, is the reaction between NO and OH radical that leads to formation of nitrous acid (HONO). During daytime HONO photo dissociates ( $\lambda < 400$  nm) rapidly back to the reactants. Therefore, HONO formed in the late evening may serve as a night-time reservoir of OH and NO, which are subsequently liberated again the following morning in the sunlight. Studies in the highly polluted Po Valley in Northern Italy have shown an interrelation between simultaneous peaks in NO<sub>x</sub> concentrations and aerosol surfaces and peak HONO concentrations during foggy periods (Notholt et al., 1992). This

was taken as evidence for heterogeneous conversion on aerosol surfaces. Probably this type of conversion plays an important role also in many urban areas over Europe, but so far only few studies have been carried out. Similar heterogeneous reactions can also occur on land surfaces (Harrison et al., 1996), and daytime formation of HONO can take place due to photo-sensitised reduction of NO<sub>2</sub> on humid acid in soil (Stemmler et al., 2006). A recent study furthermore suggest that nitrate in agricultural soil may lead to emission of HONO (Su et al., 2011).

### 3.3 Organic nitrogen compounds in the atmosphere

The presence of atmospheric organic N compounds has been evident for years (Cape et al., 2011; Cornell et al., 2003; Neff et al., 2002), but direct measurements of individual species (except PAN, discussed below) are rather sparse. The recent evidence for organic N comes from analysis of precipitation samples for total N and comparison with inorganic N content, to give “dissolved organic N” (DON) by difference. This approach has been prone to several analytical artefacts (Cape et al., 2001), but DON may in fact contribute up to half of total water-soluble N in precipitation.

The fraction of N in precipitation in organic form depends highly on location, and on whether air masses are of marine or terrestrial origin. DON has been large ignored in estimating environmental consequences of N deposition; such ecological impacts of DON depend on the bioavailability of the organic N, and there is reason to believe that many, if not all, components of DON are biologically available (Krab et al., 2008; Lipson and Nasholm, 2001; Paerl and Whittall, 1999; Qualls and Haines, 1992).

DON appears to have multiple sources, including agricultural activities and many reduced N compounds of biological origin, e.g. urea, amines, amino acids etc. (Cape et al., 2011). There appears to be a DON contribution from marine air masses (Cornell et al., 1995, 2001; Cornell, 2011), and DON proportions are consistently high in unpolluted air, especially in the tropics. For continental/terrestrial samples, annual average concentrations of DON in precipitation appear to correlate with total N and are better correlated with NH<sub>4</sub><sup>+</sup> than with NO<sub>3</sub><sup>-</sup> (Zhang et al., 2012). The overall correlation

## Governing processes for reactive nitrogen compounds

O. Hertel et al.

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

⏪

⏩

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion



suggests a significant anthropogenic source of organic N either directly or via atmospheric reactions, and the better correlation with  $\text{NH}_4^+$  than with  $\text{NO}_3^-$  support the suggestion of the importance of agricultural sources of DON emission to the atmosphere (Cape et al., 2011).

The seasonal variations in DON concentrations are not always correlated with  $\text{NH}_4^+$  concentrations, implying that different sources are involved (Cape et al., 2004). There is limited evidence available from sampling of air directly using a nebulising mist sampler that both gas-phase and particle phase components contribute to water-soluble organic N (WSON) in the atmosphere, which leads to the occurrence of DON in precipitation (Benitez et al., 2010). Organic N has been measured in fog (Zhang and Anastasio, 2001) and cloud water (Hill et al., 2007), but there is some concern that most analyses for DON are made on bulk rainfall samples (i.e. collected using an open funnel) and that a significant fraction of the measured DON might have been dry-deposited on the funnel surface (Benitez et al., 2010). This presents problems of interpretation, but does not remove the problem of identifying the source, composition and fate of organic N compounds.

Some measurements of individual components of reduced organic N in gas, particulate and aqueous phases have been reported and indicate potential sources and fates of these compounds, but in most cases rather small concentrations are measured and these cannot account for the rather high proportions of DON in precipitation. Recent very sophisticated analyses of the composition of DON aimed at identifying functional groups are consistent with an important source involving reduced N, probably via reactions in the atmosphere (Altieri et al., 2012). Correlations between total N and DON in global databases and the remote atmosphere suggest DON represents on average about 25 % of atmospheric N deposition, although this proportion is highly variable in space and time.

Organic nitrogen compounds are also formed as secondary products in the atmosphere from the reaction of various organic compounds with oxidized nitrogen species (e.g. Jenkin and Clemitshaw, 2000). For example, when aldehydes are photo

**BGD**

9, 9349–9423, 2012

## Governing processes for reactive nitrogen compounds

O. Hertel et al.

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

⏪

⏩

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion



dissociated or react with OH, an alkyl radical is formed, which in turn may form peroxy alkyl nitrates that serve as important reservoirs of NO<sub>2</sub> (Fig. 8). The most abundant of these nitrates is the peroxyacetyl nitrate (PAN). PAN is thermally unstable and equilibrium between peroxy acetyl radical and NO<sub>2</sub> on one side and PAN on the other side is established in the boundary layer. High PAN and O<sub>3</sub> concentrations are often observed together during photo chemical smog episodes. Thermal degradation gives PAN a lifetime of ~ 1.7 h at 273 K and 50 h at 263 K. The PAN formation competes with NO degradation of peroxy acetyl radical. This reaction is usually dominating at ppbv levels of NO meaning that PAN and other peroxy alkyl nitrates are usually formed only in the background atmosphere, but substantial PAN concentrations may still be observed in urban areas at relatively low temperatures. The peroxy alkyl nitrates include compounds produced in a similar way as PAN, but generated from biogenic isoprene emissions that may be of importance in southern Europe, and have similar thermal degradation pathways as PAN. PAN is an important atmospheric N reservoir species, but it is probably not an important source of DON (Cornell et al., 2003). PAN deposition has been modelled in many studies (Wu et al., 2012), and the dry deposition is much faster than expected on the basis of its solubility, with dry deposition velocities in daytime up to 10 mm s<sup>-1</sup>, so much is still unknown about the mechanisms of its atmospheric removal (Turnipseed et al., 2006).

#### 4 Dry deposition and bi-directional fluxes

N<sub>r</sub> is monitored in many regional networks across the world, such as the European EMEP programme (www.emep.int) that includes both long-term observations (Tørseth et al., 2012 and campaign studies Aas et al., 2012), the NitroEurope Integrated Project (NEU; www.nitroeuropa.eu; Sutton et al., 2007) the US National Atmospheric Deposition Network (NADP; http://nadp.sws.uiuc.edu/), the Acid Deposition Monitoring Network in East Asia (EANET; www.eanet.cc) and several others. However, these networks measure air concentrations rather than fluxes, and dry deposition is estimated

**BGD**

9, 9349–9423, 2012

### Governing processes for reactive nitrogen compounds

O. Hertel et al.

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

⏪

⏩

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion



using inferential modelling approaches (Erisman et al., 1994; Erisman and Draaijers, 1995), such as using measured air concentrations and meteorology in combination with a dry deposition algorithm (e.g. Flechard et al., 2011; Skj oth et al., 2002) which are underpinned by often sparse databases of campaign based process studies with limited geographical coverage. This is partly due to the fact that instrumentation to measure fluxes of sticky compounds such as NH<sub>3</sub>, HNO<sub>3</sub> or HONO are expensive and labour intensive to operate. The measurement of each individual N<sub>r</sub> compound is technically more challenging than that of CO<sub>2</sub> fluxes. Robust low cost flux measurement approaches are lacking, although recent development of a Conditional Time-Averaged Gradient (COTAG) method (Famulari et al., 2010) shows promise for wide-scale deployment over long periods for short vegetation.

A first European flux measurement network for reactive nitrogen compounds was established within the NitroEurope IP (2011) (Skiba et al., 2009; Sutton et al., 2007; Tang et al., 2009). This network applied at a three-tier approach, where selected N<sub>r</sub> compounds were measured at a network of 13 supersites, using advanced micrometeorological flux measurement techniques. At a further 9 regional sites the novel COTAG systems combined with a low-cost time-integrated approach for measuring soil gas fluxes (System for Inert Gas flux Monitoring by Accumulation, SIGMA, Ambus et al., 2010) were deployed, while deposition was derived at a further 50+ “inferential sites” from concentration measurements, using inferential techniques (Flechard et al., 2011; Sutton et al., 2007; Tang et al., 2009).

A comprehensively detailed spatial coverage of N<sub>r</sub> deposition can only be achieved through numerical modelling (e.g. Simpson et al., 2007, 2011). The gaseous N<sub>r</sub> compounds usually making the biggest contribution to dry deposition are NH<sub>3</sub>, HNO<sub>3</sub> and NO<sub>2</sub>. Their relative contributions to N deposition depend on the pollution climate. In agricultural areas NH<sub>3</sub> may dominate the atmospheric N loading (e.g. Hertel et al., 2006), while in more industrial and urban areas HNO<sub>3</sub> and NO<sub>2</sub> may be more important. This pattern is seen even on the European scale; Simpson et al. (Simpson et al., 2007) showed that reduced nitrogen dominates dry-deposition of N to forests close to

**BGD**

9, 9349–9423, 2012

## Governing processes for reactive nitrogen compounds

O. Hertel et al.

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

⏪

⏩

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion

the main agricultural areas, whereas oxidised nitrogen dry-deposits over much larger transport distances. In addition,  $\text{NH}_3$  deposition depends on the N status of the receiving surface, with fertilised vegetation and vegetation receiving high atmospheric N deposition inputs acting as a less efficient sink or even, in the case of fertilized vegetation, a net source of  $\text{NH}_3$  (Loubet et al., 2009; Sutton et al., 1993). In wet regions, leaf cuticles frequently provide an efficient sink for water soluble gases ( $\text{NH}_3$  and  $\text{HNO}_3$ ), although the cuticular sinks for both gases can become saturated under very dry conditions, increasing the importance of stomata as an uptake pathway (Nemitz et al., 2004b).

#### 4.1 The dry deposition process

Dry deposition is the direct uptake of gases or aerosols at terrestrial or marine surfaces. The dry deposition of gases and particles is a continuous process and governed by their air concentrations, turbulent transport processes in the boundary layer, the chemical and physical nature of the depositing species, and the biological and chemical capability of the surface to capture or absorb the species (e.g. Fowler et al., 2009).

In relation to deposition transport, the boundary layer may be considered to consist of two layers: the fully turbulent layer and the quasi-laminar layer. The quasi-laminar layer is introduced to quantify the way in which pollutant transfer differs from momentum transfer in the immediate vicinity of the surface (Hicks et al., 1987). In this layer, the transport is dominated by molecular diffusion. Once at the surface, the chemical, biological and physical nature of the surface determines the capture or absorption of the gases and particles. Deposition to water surfaces (oceans or fresh waters) may thus be very different from deposition to vegetated surfaces on land.

The deposition process may be considered as a series of resistances, by analogy with an electrical circuit (Monteith and Unsworth, 2008). The resistances refer to the transport processes through the various “layers” defined above: turbulent transfer (usually denoted  $R_a$ ), quasi-laminar ( $R_b$ ) and surface ( $R_c$ ). For a complex surface with several potential absorption sinks (e.g. vegetation) the resistance  $R_c$  may be viewed as

**BGD**

9, 9349–9423, 2012

### Governing processes for reactive nitrogen compounds

O. Hertel et al.

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

⏪

⏩

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion



a network of parallel resistances, representing transfer to the external leaf surface, through stomata, to water on the surface, or through the canopy to the underlying soil surface. The total resistance ( $R_t$ ) is the sum of all the series and parallel resistances ( $R_a + R_b + R_c$ ), and is usually expressed in units of  $\text{s m}^{-1}$ . The inverse of the total resistance ( $1/R_t$ ) is known as the deposition velocity ( $v_d$ ), representing the deposition flux ( $F$ ) normalized by the air concentration at height  $z$  ( $X_z$ ), i.e.  $v_d(z) = -F/X_z$  and has units of  $\text{m s}^{-1}$  (Fowler et al., 2001a; Hicks et al., 1987).

The turbulent transfer resistance ( $R_a$ ) depends upon the height at which the deposition flux is measured, so the total resistance ( $R_t$ ) and deposition velocity ( $v_d$ ) also vary with height above the surface. The transfer flux ( $F$ ) is defined as the product of the air concentration of a gas or particles at height  $z$ , multiplied by the deposition velocity at height  $z$ , and in the absence of competing chemical reactions (Sorensen et al., 2005), does not vary with height, provided that the air concentration is horizontally uniform.

The resistance formulation described above assumes that the concentration of the gas at the absorbing surface is zero (e.g. on the leaf cuticle or within plant stomata). Where this is not the case (see below) the effect can be described mathematically either as a decreased driving force for deposition (concentration difference between height  $z$  and the non-zero surface concentration) or as an increased surface resistance. The interpretation as a surface resistance in this context has the disadvantage that it is unable to simulate the bi-directional fluxes induced by non-zero surface concentrations.

The deposition velocity ( $v_d$ ) is often reported as a constant even though it depends on a set of variables, e.g. wind speed, surface roughness and atmospheric stratification. Joffre (1988) has suggested a parameterization which depends on the meteorological conditions, roughness length and the molecular diffusion coefficient for the compound of interest. The various components of the total transport resistance can be estimated from meteorological data if several assumptions are made concerning spatial and temporal homogeneity (Erisman and Draaijers, 1995; Hicks et al., 1987; Nemitz and Sutton, 2004; Smith et al., 2000). The surface resistance ( $R_c$ ) term depends on the physical and chemical nature of the absorbing surface, and parameterisations should

**BGD**

9, 9349–9423, 2012

## Governing processes for reactive nitrogen compounds

O. Hertel et al.

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

⏪

⏩

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion



be adapted to the surface concerned.  $v_d$  is often expressed as annual or seasonal averages, for the purpose of calculating deposition fluxes as the product of air concentrations and deposition velocities.

When calculating fluxes, deposition velocities need to be used together with concentrations from the same height ( $z$ ) – usually the height at which the concentrations are measured. Tall vegetation causes increased atmospheric turbulence, so  $R_a$  values are smaller, and deposition velocities are larger, than to short vegetation. Consequently, estimating deposition of different components requires knowledge about land-cover as well as the spatial pattern of air concentrations. The air-sea gas exchange of the very soluble gases  $\text{HNO}_3$  and  $\text{NH}_3$  is rate limited by the vertical transport in the boundary layer, because the uptake at the water surface is very fast relative to other commonly studied gases. Of the two very soluble N-gases,  $\text{HNO}_3$  exchange rates tend to be generally larger than  $\text{NH}_3$  due to the higher solubility (i.e. smaller  $R_c$ ), although  $\text{NH}_3$  deposition can be faster to a perfect sink due to its smaller molecular weight than  $\text{HNO}_3$  and correspondingly smaller boundary layer resistance ( $R_b$ ). The less soluble  $\text{NO}_2$  and  $\text{NO}$  gases deposit much slower to the marine surface. The surface resistance is the most important resistance for slightly soluble gases and relates to the transfer velocity  $K_c$ , which is also used for air-sea exchange of other gases like  $\text{CO}_2$ ,  $\text{DMS}$  and  $\text{CH}_4$ .

The surface resistance ( $R_c$ ) is a key parameter for the deposition of a gas and several parameterization schemes have been derived from flux measurements. For sub-micron particles, the transport through the boundary layer is more or less the same as for gases. However, transport of particles through the quasi-laminar layer governing  $R_b$  can differ. For particles with an aerodynamic diameter  $< 0.1 \mu\text{m}$ , deposition is controlled by diffusion, whereas deposition of particles with an aerodynamic diameter  $> 10 \mu\text{m}$  is more controlled by sedimentation (Seinfeld and Pandis, 2006). Deposition of particles with an aerodynamic diameter between  $0.1$  and  $1 \mu\text{m}$  is determined by the rates of impaction and interception and depends heavily on the turbulence intensity. Transfer through the quasi-laminar layer close to the surface presents a considerable restriction on the deposition of  $0.1$ – $1.0 \mu\text{m}$  aerodynamic diameter particles. Uptake of particles

**BGD**

9, 9349–9423, 2012

## Governing processes for reactive nitrogen compounds

O. Hertel et al.

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

⏪

⏩

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion

---

## Governing processes for reactive nitrogen compounds

O. Hertel et al.

---

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

⏪

⏩

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion



by surfaces is thus largely controlled by micro-structures and turbulence intensity. Although a range of theory-based models is available to describe aerosol deposition, they often predict features which conflict with measured deposition rates (Flechard et al., 2011; Petroff et al., 2008; Pryor et al., 2008a, b). For example, methods based on the well-known formulations of Slinn (Slinn, 1982) predict low deposition velocities to forest canopies. Alternative formulae of Zhang et al. (Zhang et al., 2001) predict higher deposition velocities, but no effect of canopy density. Several studies show that  $\text{NH}_4\text{NO}_3$  has higher deposition velocities than sulphates, as a result of the partitioning of  $\text{NH}_4\text{NO}_3$  to the more rapidly depositing  $\text{HNO}_3$  and  $\text{NH}_3$  gases (e.g. Fowler et al., 2009; Nemitz et al., 2004a; Wolff et al., 2010). Deposition of particles containing  $\text{SO}_4^{2-}$ ,  $\text{NO}_3^-$ ,  $\text{Cl}^-$  and  $\text{NH}_4^+$  contributes to the potential acidification and eutrophication (N components) of ecosystems. Compared to gaseous deposition of acidifying compounds onto low vegetation, particle deposition fluxes are usually found to be small. However, unlike wet deposition it takes place all the time and furthermore it is believed that the fluxes of small particles are currently underestimated for very rough surfaces like forests. Erisman et al. (1996) found that deposition of aerosols to the Speulder forest contributed 20 and 40% to the total dry deposition of S and N, respectively. Parameterisations of aerosol dry deposition velocities to forests differ greatly between models (Flechard et al., 2011).

### 4.2 Bi-directional fluxes of N-containing gases

Although reports of transient  $\text{N}_2\text{O}$  deposition fluxes are increasing in number, e.g. (Flechard et al., 2007), in terms of N inputs, the magnitude of  $\text{N}_2\text{O}$  uptake is small and negligible compared with the main contributors to atmospheric  $\text{N}_r$  deposition. For other  $\text{N}_r$  containing gases there are several parallel pathways of pollutant exchange with vegetation, including adsorption to leaf cuticles, exchange through the stomata with the sub-stomatal cavity and exchange with the soil. All these processes are potentially bi-directional, depending on the relative magnitude of the air concentration and the gaseous concentrations in chemical equilibrium with the leaf surface, the apoplastic

fluid and the soil solution, respectively. The likelihood for uptake increases with the water solubility and Henry's Law coefficient of the gas, which vary several orders of magnitude ([www.mpch-mainz.mpg.de/~sander/res/henry.html](http://www.mpch-mainz.mpg.de/~sander/res/henry.html)).

#### 4.2.1 Nitric acid

5 Due to its high deposition rate,  $\text{HNO}_3$  makes a significant contribution to Nr deposition in regions with high  $\text{NO}_x$  emissions.  $\text{HNO}_3$  is highly water soluble and commonly assumed to deposit at the maximum rate permitted by turbulence, i.e. surface resistance is negligible. This is probably a reasonable approximation for most situations, but emission gradients or reduced uptake rates of  $\text{HNO}_3$  have been observed, probably due to non-zero  $\text{HNO}_3$  surface concentrations in equilibrium with  $\text{NH}_4\text{NO}_3$  deposited to leaf surfaces (Neftel et al., 1996; Nemitz et al., 2004; Zhang et al., 1995). For trace gases with negligible surface resistance, the deposition velocity is sensitive to the atmospheric resistances ( $R_a$  and  $R_b$ ), which over rough surfaces are usually small ( $5\text{--}10\text{ s m}^{-1}$ ). In such conditions, even a small surface resistance would strongly influence deposition rates. Currently there are insufficient field data to show whether  $\text{HNO}_3$  deposition is subject to a surface resistance, and this remains a research priority.

#### 4.2.2 Ammonia

$\text{NH}_3$  is less water soluble than  $\text{HNO}_3$ , and  $\text{NH}_3$  previously absorbed to wet leaf surfaces may more readily be desorbed (re-emitted) as leaf water layers dry out again so long as this has not yet been absorbed into the leaf tissues (Flechard et al., 1999; Sutton et al., 1998). Generally plants contain inorganic N in the form of  $\text{NH}_4^+$  and  $\text{NO}_3^-$ . These nutrients are mainly present in the liquid part (apoplast) between the cells of the plant.  $\text{NH}_4^+$  is an important by-product of plant biochemical pathways resulting in non-zero  $\text{NH}_4^+$  concentrations in the leaf apoplast, which results in non-zero gas-phase concentrations (stomatal compensation points,  $\chi_s$ ) in equilibrium with this  $\text{NH}_{4\text{apo}}^+$  concentration

**BGD**

9, 9349–9423, 2012

### Governing processes for reactive nitrogen compounds

O. Hertel et al.

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

⏪

⏩

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion

at the apoplastic pH (Farquhar et al., 1980; Massad et al., 2010b; Mattsson et al., 2009; Schjørring et al., 1998; Sutton et al., 1995).

Available evidence suggests that  $\text{NH}_4^+$  increases with increasing N supply to the plant, either through fertilization or high atmospheric N inputs (Massad et al., 2010b; Sutton et al., 1995). The system is here an exchange governed by the pH depending relation between  $\text{NH}_3$  and  $\text{NH}_4^+$ . The compensation point  $\chi_s$  is the product of a temperature function describing the Henry's Law equilibrium and the ratio of  $\Gamma_s = [\text{NH}_4^+]_{\text{apo}} / [\text{H}^+]_{\text{apo}}$ . Values of  $\Gamma_s$  range from  $< 100$  for semi-natural vegetation in clean, remote environments to values around 1500 for semi-natural vegetation in environments with high  $\text{N}_r$  deposition, and to  $> 10\,000$  after fertilization (Massad et al., 2010b). At  $10^\circ\text{C}$ , this equates to values of  $\chi_s$  of  $< 0.15$ , 2.3 and  $> 15\ \mu\text{g NH}_3\ \text{m}^{-3}$ , respectively. Emission potentials of fertilized soils and plant litter can be several orders of magnitude larger (Nemitz et al., 2000; Sutton et al., 2009a, b), allowing  $\text{NH}_3$  emitted at the ground to be recaptured by an overlaying plant canopy (Denmead et al., 1976). The decomposition of plant litter has been found to play an important role in agricultural canopies, such as oilseed rape and managed grassland (Denmead et al., 1976; Nemitz et al., 2000; Sutton et al., 2009b, c; Zhang et al., 2010) but is much less certain will often be less important for semi-natural vegetation, especially where litter has lower nitrogen content or is more acidic.

The large range of plant, litter and soil  $\Gamma$  values illustrates that the direction of  $\text{NH}_3$  exchange is often difficult to estimate a priori. There are several reviews in the literature on bi-directional  $\text{NH}_3$  exchange which have compiled data on compensation points (Massad et al., 2010b; Zhang et al., 2010) in order to provide the necessary input for application in atmospheric transport models. The compensation points increase with N input as it is the main driver of apoplast and bulk leaf  $\text{NH}_4^+$  concentrations (Massad et al., 2010a; Sutton et al., 1993; Zhang et al., 2010), but the compensation point also varies between different plant species even when growing under the same conditions (Mattsson et al., 2009) and with growth stage and season (Milford et al., 2009; Riedo et al., 2002).

## Governing processes for reactive nitrogen compounds

O. Hertel et al.

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

⏪

⏩

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion



The stomatal pathway for NH<sub>3</sub> exchange is only available when stomata are open during daytime, and thus deposition to (often wet) leaf surfaces is the dominant pathway during the night, unless soil surfaces provide a major source and are well exposed to the atmosphere. Maps of NH<sub>3</sub> net exchange are particularly uncertain, due to (i) uncertainties in the overall magnitude as well as spatial and temporal patterns of agricultural NH<sub>3</sub> emissions and (ii) the large variability of NH<sub>3</sub> deposition rates to different surfaces.

The development of resistance models to describe bi-directional NH<sub>3</sub> exchange for eventual incorporation in atmosphere transport models has progressed through several clear stages in recent decades. Initially, atmospheric transport models only allowed the application of uni-directional dry deposition, using values of  $v_d$  (Asman and Janssen, 1987). Introduction of the idea of a stomatal compensation point by Farquhar et al. (1980) then led to incorporation of simple compensation point values in atmospheric models (see Fischer, 1987). The next stage of development can with the introduction of the canopy compensation point approach, which offset bi-directional stomatal exchange with uptake onto leaf surfaces (Sutton et al., 1995; Sutton and Fowler, 1993), with this approach subsequently being adopted in both small- and large-scale atmospheric transport models by Dentener and Crutzen (1994); Sorteberg and Hov (1996) and Loubet et al. (2001). For example, Sorteberg and Hov (1996) found a reduction of 0–20 % in total sulphur deposition and a 0–25 % increase in NH<sub>3</sub> deposition when using a bidirectional scheme compared with a simple flux model allowing only NH<sub>3</sub> deposition. Lately Wichink Kruit et al. (2012) found that the inclusion of a compensation point model in the LOTUS-EUROS model redistributed N depositions on the regional scale and provided a considerable increase in NH<sub>3</sub> concentrations over sea areas, thus reflecting observed open sea concentrations (e.g. de Leeuw et al., 2003) to a much larger degree.

While the single layer canopy compensation point approach provided the foundation to simulate competing stomatal and cuticular processes within the plant canopy, it did not address the contribution of ammonia emissions from the ground surface or allow

## Governing processes for reactive nitrogen compounds

O. Hertel et al.

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures



Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion





dynamics of bidirectional cuticular exchange. The latter process was subsequently addressed, allowing simulation of  $\text{NH}_3$  release from drying leaf surfaces, initially using a fixed cuticular pH (Neiryneck et al., 2005; Sutton et al., 1998), and subsequently extended to simulate full leaf surface chemistry, the model solving leaf surface pH according to comprehensive ion exchanges (Burkhardt et al., 2009; Flechard et al., 1999). Until now, this dynamic approach to treating leaf cuticle adsorption and desorption has been considered too complex for incorporation in atmospheric transport models.

By contrast to cuticular adsorption/desorption, the inclusion of a two-layer canopy compensation point approach has been considered more tractable for general model application. The basis of this approach was introduced by Nemitz et al. (2000, 2001), and has since formed the foundation for generalized empirical parameterizations estimated by Massad et al. (2010b) and Zhang et al. (2010) for use in atmospheric dispersion models e.g. Gore et al. (2009).

Experimental studies have shown that over the sea the atmospheric fluxes of  $\text{NH}_3$  may also be upward or downward (Lee et al., 1998; Quinn et al., 1988; Sørensen et al., 2003) depending on the meteorological conditions and the relationship between the pH and  $\text{NH}_4^+$  concentration in the upper surface waters on the one side, and the  $\text{NH}_3$  concentrations in ambient air just above the water surface on the other side. The bi-directional  $\text{NH}_3$  flux over sea is expressed as a simple compensation point exchange with the water surface:  $F = V_e (C_{\text{eq}} - C_{\text{air}})$ , where  $v_e$  is the exchange velocity between air and sea (that equals  $1/(R_a + R_b)$ ),  $C_{\text{eq}}$  is the  $\text{NH}_3$  concentration in the air at equilibrium with the  $\text{NH}_x$  in the water, and  $C_{\text{air}}$  is the actual ambient air concentration of  $\text{NH}_3$ . The ambient air  $\text{NH}_3$  concentration at equilibrium was expressed by Asman et al. (1994) as a function of  $\text{NH}_x$  concentration in sea water, activity coefficients for  $\text{NH}_3$  and  $\text{NH}_4^+$  in sea water, Henry's law coefficient for  $\text{NH}_3$  and pH in sea water (Asman et al., 1994). The formulation has since been applied into the Lagrangian ACDEP model (Sørensen et al., 2003) using a distribution of  $\text{NH}_x$  concentrations in sea water from Barret (Barrett, 1998), where the results showed a redistribution of N deposition in the coastal region off the coast of the Netherlands. On a global basis Johnson et al. (2008)

## Governing processes for reactive nitrogen compounds

O. Hertel et al.

[Title Page](#)[Abstract](#)[Introduction](#)[Conclusions](#)[References](#)[Tables](#)[Figures](#)[⏪](#)[⏩](#)[◀](#)[▶](#)[Back](#)[Close](#)[Full Screen / Esc](#)[Printer-friendly Version](#)[Interactive Discussion](#)

emphasised the importance of temperature control on air-sea  $\text{NH}_3$  exchange over the remote ocean and argued that  $\text{NH}_3$  emissions were much more likely over warm waters than over cold (Johnson et al., 2008). Similarly, results with the ACDEP model suggested high sensitivity to both  $\text{NH}_4^+$  concentration and pH of the sea water (Sørensen et al., 2003).

### 4.2.3 Nitrogen monoxide

$\text{NO}$  is rather water-insoluble and there is no efficient mechanism for  $\text{NO}$  to react on the surface or inside leaves, so its deposition rate is rather slow. By contrast, soils are a source and sink for  $\text{NO}$ . Some of these soil emissions of  $\text{NO}$  are oxidized to  $\text{NO}_2$  (and possibly  $\text{HNO}_3$ ) within plant canopies, and taken up more efficiently than  $\text{NO}$  and thus the behaviour of  $\text{NO}$  still needs to be taken into account in surface – atmosphere exchange.

### 4.2.4 Nitrogen dioxide

Plant uptake of  $\text{NO}_2$  is slower than that of the more water soluble gases ( $\text{HNO}_3$ ,  $\text{NH}_3$ ), but it is a significant contributor to N deposition. The  $\text{NO}_2$  deposition to vegetation is primarily regulated by stomata, and for most plants the internal resistance is negligible, and  $\text{NO}_2$  deposition velocities may thus be computed from knowledge of stomatal resistance or conductance (Thoene et al., 1991). Studies indicate a small effective stomatal compensation point for  $\text{NO}_2$  for some plant species, in the range of  $> 0$  to 2 ppb; e.g. an American experimental study found a value of 1.5 ppb for the canopy compensation point for  $\text{NO}_2$  over deciduous forest (Horii et al., 2004). However, the underlying process is not currently understood, and some laboratory work has failed to reproduce the field observations. Because of its low water solubility, deposition to (and reaction with) surface water, including sea water, is also slow (Cape et al., 1993).

**BGD**

9, 9349–9423, 2012

## Governing processes for reactive nitrogen compounds

O. Hertel et al.

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

⏪

⏩

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion

## 4.2.5 Nitrous acid

The biosphere/atmosphere exchange of HONO is generally bi-directional, and day-time concentrations of HONO are low, as it is rapidly photolysed in sunlight. With solubility similar to NH<sub>3</sub>, HONO is deposited to vegetation under most conditions. Observations of HONO emission have been attributed to production of HONO at surfaces, e.g. through the reaction of NO<sub>2</sub> with NO on wet surfaces (Harrison et al., 1994) or NO<sub>2</sub> reduction on humic acid (Stemmler et al., 2006). In connection with an experimental study, a parameterisation of bi-directional fluxes of both NH<sub>3</sub> and HONO was applied for estimating dry deposition of N to the Amazon basin from measured ambient air concentrations (Trebs et al., 2006).

## 4.2.6 Organic nitrogen compounds

Newly developed instruments have resulted in new measurements indicating deposition rates of PAN (and other PAN-like compounds) that are significantly larger than classical predictions (Turnipseed et al., 2006; Wolfe et al., 2009), especially to wet vegetation. Thus the lifetime of PAN with respect to deposition may be shorter than previously thought. In addition, PAN is water-insoluble and the comparably large deposition fluxes to wet surfaces indicate that the current mechanistic understanding of the deposition process is incomplete. There are parallels to the deposition of O<sub>3</sub>, which also appears to exhibit larger deposition rates to wet surfaces than can be explained by its solubility (Fowler et al., 2001b). The importance of alkyl nitrates has recently been demonstrated for Blodgett Forest, Sierra Nevada, USA (Farmer et al., 2006), although it appears that the pollution climate of their site is unusual. Nevertheless, information is lacking to form a robust picture of the importance of these compounds across the full range of European conditions. Although amines have been measured as emitted from agricultural activities (Schade and Crutzen, 1995), there is currently no information on their dry deposition. Amines play a significant role in atmospheric new particle formation. Petäjä et al. (2011) showed that under atmospherically relevant conditions

**BGD**

9, 9349–9423, 2012

### Governing processes for reactive nitrogen compounds

O. Hertel et al.

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

⏪

⏩

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion



amines are needed to explain cluster formation (Petäjä et al., 2011). Also several laboratory experiments have pointed out that amines have a marked enhancing impact on particle formation (Berndt et al., 2010; Murphy et al., 2007). Kurten et al. used quantum chemistry calculations and concluded that amines are far more efficient than ammonia in stabilizing small H<sub>2</sub>SO<sub>4</sub> -clusters in the atmosphere (Kurten et al., 2008).

## 5 Wet deposition

Unlike dry deposition, the wet deposition processes are indirect in that rain, hail and snow are vectors for conveying gaseous and aerosol compounds to the surface. The simple precipitation collectors applied in monitoring networks contrast appreciably with the underlying physical and chemical pathways of solutes into the collected precipitation sample. There is also significant uncertainty in the relative magnitudes of dry deposition of trace chemical species as gases and aerosols onto the collecting equipment although this issue can be minimised in “wet only” collectors which are covered during dry periods.

### 5.1 Wet scavenging of aerosols

The bulk of the NH<sub>4</sub><sup>+</sup> and NO<sub>3</sub><sup>-</sup> aerosol mass is present in the size range 0.1 to 1.0 μm (diameter). These aerosols are removed through interception by falling rain or snow, a process known as washout or by incorporation of the aerosol into cloud droplets within clouds, a process known as rainout. Washout is responsible for 10% to 20% of the N<sub>r</sub> in wet deposition. The aerosol scavenging within cloud occurs through a number of physical and chemical pathways while the gases are incorporated through solution and oxidation processes. The phoretic process includes diffusiophoresis, in which aerosol particles are transported in the direction of a mean flux of vapour molecules. In the case of a cloud droplet growing by vapour diffusion of water molecules towards the droplet surface, aerosols would move along the vapour flux towards the growing

## Governing processes for reactive nitrogen compounds

O. Hertel et al.

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

⏪

⏩

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion



droplet. Additional phoretic mechanisms are presented by electrical and thermal gradients (electrophoresis and thermophoresis respectively). The phoretic processes contribute relatively small amounts of the solute in cloud water (Goldsmith et al., 1963).

Aerosols may also be captured by cloud droplets following Brownian diffusion to the droplet surface and rates of Brownian diffusion vary strongly with particle size, being significant for particles smaller than 100 nm in diameter. However, diffusion rates are very small relative to molecular diffusion and diffusional mechanisms make only minor contributions to the wet removal pathway. The remaining minor process leading to capture of aerosols by cloud droplets is impaction and interception. As implied in the name these processes lead to the capture of aerosols by droplets when one is unable to follow the streamlines of airflow around the other and the aerosol and droplet collide. The bulk of the aerosol N in cloud water is incorporated through the activation of aerosols containing  $\text{NO}_3^-$  or  $\text{NH}_4^+$  into cloud droplets. The N containing aerosols are effective cloud condensation nuclei and are readily incorporated into cloud droplets through the nucleation scavenging pathway. Thus the main route is nucleation scavenging for aerosol  $\text{NO}_3^-$ , and  $\text{NH}_4^+$  (Pruppacher and Jaenicke, 1995). The pathway for below-cloud wet scavenging of the gaseous  $\text{N}_r$  depends on the solubility and reactivity of the specific gas.  $\text{NH}_3$  and  $\text{HNO}_3$  are highly soluble, and clouds and rain remove these gases effectively from the air. The contribution of NO and  $\text{NO}_2$  to dissolved N in precipitation is very small as these gases are not very soluble. Wet deposition is monitored by simple methods (precipitation collectors) analysed for major anthropogenic ions  $\text{SO}_4^{2-}$ ,  $\text{NO}_3^-$ ,  $\text{NH}_4^+$ ,  $\text{H}^+$  and marine ions  $\text{Cl}^-$ ,  $\text{Na}^+$ ,  $\text{Mg}^{2+}$ . The networks of collectors for precipitation chemistry are much less dense than precipitation collectors for measurement of rainfall amounts, due to costs of chemical analysis. Furthermore, precipitation chemistry collectors are located at a height above ground to reduce contamination from ground based sources (Erisman et al., 2003), and the practice of locating collectors above the ground reduces the capture of small droplets due to aerodynamic screening by the collector (Dammgen et al., 2005).

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## Governing processes for reactive nitrogen compounds

O. Hertel et al.

---

[Title Page](#)[Abstract](#)[Introduction](#)[Conclusions](#)[References](#)[Tables](#)[Figures](#)[Back](#)[Close](#)[Full Screen / Esc](#)[Printer-friendly Version](#)[Interactive Discussion](#)

## 5.2 Distribution between dry and wet deposition

The relative contributions to deposition from dry and wet deposition change with distance from source as primary pollutant concentrations decline and oxidation from gas to particle removes gas phase species which dry deposit quickly. Thus areas more than a few hundred km from sources receive most of their  $N_r$  deposition in precipitation (Bartnicki et al., 2011; de Leeuw et al., 2003; Hertel et al., 2003), except possibly for forests whose aerodynamic roughness maximises dry-deposition (Simpson, 2006 1537/id). In regions in which the amounts of precipitation are large, wet deposition dominates the  $N_r$  loads, as in most of the uplands of Europe. However, it is not simply the precipitation amount that needs to be considered in assessing the relative contributions of wet and dry deposition. The processes leading to orographic enhancement of rainfall amount have a profound effect on the overall scavenging of pollutants from the atmosphere.

## 5.3 Orographic effects

The meteorological process which enhances precipitation in much of maritime northern Europe is the seeder – feeder mechanism, in which orographic cloud, formed over hills and mountains is washed out by precipitation falling from higher levels in the troposphere (Bergeron, 1965). Mountains are very effective in increasing rainfall and wet deposition by the seeder-feeder process in which low level hill cloud droplets are washed out by falling precipitation from higher levels. The hill cloud is more polluted than higher level cloud because boundary layer aerosols are effectively activated into cloud droplets as they are forced to rise and cool over the hills and mountains. The seeder – feeder effects on precipitation amount have been simulated in process-based models and are able to simulate observed spatial patterns in precipitation (Carruthers and Choularton, 1983). Models have also been used to simulate the wet deposition of pollutants over mountains (Dore et al., 1990) and compared with detailed campaign measurements in an upland area. Extending the modelling of orographic enhancement of wet deposition to the country scale has enabled detailed spatially resolved wet deposition maps to be

**BGD**

9, 9349–9423, 2012

### Governing processes for reactive nitrogen compounds

O. Hertel et al.

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

⏪

⏩

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion



generated (Dore et al., 1990). As orographic enhancement of wet deposition has been shown to be a major contributor to the total deposition in upland Britain the explicit inclusion of the process in deposition maps has been regarded as a routine component of wet deposition mapping (<http://www.rotap.ceh.ac.uk>). The resulting wet, and total N deposition maps show a strong influence of altitude and require a grid resolution on the same scale as the complex topography to reproduce (< 10 km). Thus deposition modelling and mapping at a 50 km × 50 km scale fails to capture the spatial structure in wet deposition (Fig. 8; Simpson et al., 2011). In principle the models are able to simulate the process, but the grid resolution of both the underpinning meteorological model and the model applied for deposition calculations needs to be able to capture the topographic scale of the variability and also take into account the spatial variation in the emissions (Dore et al., 2012), which for the UK area have been shown to be a spatial resolution of at least 1 km (Dore et al., 2012).

#### 5.4 Cloud droplet deposition

Unlike aerosols in the size range 0.1–1.0 μm, which are not deposited efficiently on vegetation, the hill cloud droplets are large enough (3 to 10 μm in diameter), to impact efficiently on vegetation (Fowler et al., 1990); this deposition pathway is termed cloud deposition or occult deposition. For the UK it provides a very small contribution to the total but it is important for hills which are frequently shrouded in cloud. As the concentrations of major ions in hill cloud are enhanced, this deposition pathway leads to the exposure of vegetation to very high concentrations ( $\text{SO}_4^{2-}$ ,  $\text{NH}_4^+$ ,  $\text{NO}_3^-$ , 1000 μeq l<sup>-1</sup> to 2000 μeq l<sup>-1</sup>) (Fowler et al., 1990). The orographic enhancement of wet deposition is not included in the assessments of wet deposition in all countries, and for regions with only small areas of upland, this will not lead to significant underestimates in wet deposition. However, for areas of Europe in which seeder – feeder scavenging of pollutant represents a substantial contribution to total deposition, it is important to simulate the

**BGD**

9, 9349–9423, 2012

### Governing processes for reactive nitrogen compounds

O. Hertel et al.

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

⏪

⏩

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion

process in mapping regional wet deposition, to avoid underestimating wet deposition and exceedances of critical loads.

## 6 Conclusions

The emissions of  $N_r$  compounds are mainly related to releases of anthropogenic origin. These releases have typically strong diurnal and in some case also seasonal variations that relate to human activities ( $NO_x$  and  $NH_3$ ) and climatic variables (mainly  $NH_3$ ).  $N_r$  compounds are subject to rapid transformations in the atmosphere which profoundly affect their deposition rates. Thus  $NH_3$  has a fast deposition rate but it also reacts with acids to form  $NH_4^+$  salt aerosols which have low dry deposition velocities, but are liable to wet deposition. As a consequence, then  $NH_4^+$  containing aerosols are readily capable of long-range transport.  $NO_x$  emissions are mainly in the form of  $NO$ , but the emissions from road traffic contain an important fraction of primary  $NO_2$  which has increased in recent years.  $NO$  is rapidly converted to  $NO_2$ , which in turn converts to  $HNO_3$  either through daytime reaction with the  $OH$  radical, or by heterogeneous conversion processes at night.  $HNO_3$  is subject to rapid deposition, but may also convert to  $NH_4NO_3$  which is subject to long-range transport.  $NO_2$  may also be converted by reaction on land and aerosol surfaces to  $HONO$ , which is capable of photolysis to form  $OH$  radicals. Ambient air concentrations of  $N_r$  compounds are generally fairly well (often within  $\pm 20\%$ – $30\%$ ) reproduced by state-of-the-art models, while estimates of deposition are considerably more uncertain (often more than  $\pm 50\%$ ). A significant part of the uncertainty in current CTMs is related to the fluxes with the surface, both emissions and deposition. Studies of fluxes of  $N_r$  are therefore in considerable need, especially for sensitive ecosystems. These flux studies need to include detailed field studies, parameterization, application and testing of models covering both the biosphere and the atmosphere such as CTMs. Furthermore, long-term trends show very different emission patterns for  $NH_3$  and  $NO_x$ , where emissions in Europe and the US show marked declines for  $NO_x$  and moderate reductions or stagnation for  $NH_3$ . In contrast, emissions

### Governing processes for reactive nitrogen compounds

O. Hertel et al.

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

◀

▶

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion





## Governing processes for reactive nitrogen compounds

O. Hertel et al.

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

⏪

⏩

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion



from China, for instance, show a steep increase in the past decade and continue to grow rapidly (Reis et al., 2009). These variations will cause considerable changes in the atmosphere-biosphere system, where most N related feedback mechanisms remain to be studied (Arneth et al., 2010). Long-term studies of biosphere-atmosphere interactions in relation to  $N_f$  that take full cascade of effects into account is therefore highly needed.

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---

**Governing  
processes for  
reactive nitrogen  
compounds**

---

O. Hertel et al.

[Title Page](#)[Abstract](#)[Introduction](#)[Conclusions](#)[References](#)[Tables](#)[Figures](#)[⏪](#)[⏩](#)[◀](#)[▶](#)[Back](#)[Close](#)[Full Screen / Esc](#)[Printer-friendly Version](#)[Interactive Discussion](#)

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---

**Governing  
processes for  
reactive nitrogen  
compounds**

---

O. Hertel et al.

---

[Title Page](#)[Abstract](#)[Introduction](#)[Conclusions](#)[References](#)[Tables](#)[Figures](#)[⏪](#)[⏩](#)[◀](#)[▶](#)[Back](#)[Close](#)[Full Screen / Esc](#)[Printer-friendly Version](#)[Interactive Discussion](#)

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## Governing processes for reactive nitrogen compounds

O. Hertel et al.

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

⏪

⏩

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion

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---

## Governing processes for reactive nitrogen compounds

O. Hertel et al.

---

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures



Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion



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---

## Governing processes for reactive nitrogen compounds

O. Hertel et al.

---

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

⏪

⏩

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion



---

## Governing processes for reactive nitrogen compounds

O. Hertel et al.

---

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

⏪

⏩

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion



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---

## Governing processes for reactive nitrogen compounds

O. Hertel et al.

---

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

◀

▶

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion

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## Governing processes for reactive nitrogen compounds

O. Hertel et al.

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

◀

▶

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion

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**Governing  
processes for  
reactive nitrogen  
compounds**O. Hertel et al.

---

[Title Page](#)[Abstract](#)[Introduction](#)[Conclusions](#)[References](#)[Tables](#)[Figures](#)[⏪](#)[⏩](#)[◀](#)[▶](#)[Back](#)[Close](#)[Full Screen / Esc](#)[Printer-friendly Version](#)[Interactive Discussion](#)

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O. Hertel et al.

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

⏪

⏩

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion

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**Governing  
processes for  
reactive nitrogen  
compounds**O. Hertel et al.

---

[Title Page](#)[Abstract](#)[Introduction](#)[Conclusions](#)[References](#)[Tables](#)[Figures](#)[⏪](#)[⏩](#)[◀](#)[▶](#)[Back](#)[Close](#)[Full Screen / Esc](#)[Printer-friendly Version](#)[Interactive Discussion](#)

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## Governing processes for reactive nitrogen compounds

O. Hertel et al.

---

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

⏪

⏩

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion



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---

## Governing processes for reactive nitrogen compounds

O. Hertel et al.

---

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

⏪

⏩

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion



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## Governing processes for reactive nitrogen compounds

O. Hertel et al.

---

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

⏪

⏩

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion

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## **Governing processes for reactive nitrogen compounds**

O. Hertel et al.

---

[Title Page](#)

[Abstract](#)

[Introduction](#)

[Conclusions](#)

[References](#)

[Tables](#)

[Figures](#)

[⏪](#)

[⏩](#)

[◀](#)

[▶](#)

[Back](#)

[Close](#)

[Full Screen / Esc](#)

[Printer-friendly Version](#)

[Interactive Discussion](#)



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---

## Governing processes for reactive nitrogen compounds

O. Hertel et al.

---

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

⏪

⏩

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion



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---

## Governing processes for reactive nitrogen compounds

O. Hertel et al.

---

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

⏪

⏩

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion



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## Governing processes for reactive nitrogen compounds

O. Hertel et al.

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

◀

▶

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion

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## Governing processes for reactive nitrogen compounds

O. Hertel et al.

---

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

⏪

⏩

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion



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## Governing processes for reactive nitrogen compounds

O. Hertel et al.

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

⏪

⏩

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion



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## Governing processes for reactive nitrogen compounds

O. Hertel et al.

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

⏪

⏩

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion

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## Governing processes for reactive nitrogen compounds

O. Hertel et al.

[Title Page](#)
[Abstract](#)
[Introduction](#)
[Conclusions](#)
[References](#)
[Tables](#)
[Figures](#)




[Back](#)
[Close](#)
[Full Screen / Esc](#)
[Printer-friendly Version](#)
[Interactive Discussion](#)


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## Governing processes for reactive nitrogen compounds

O. Hertel et al.

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

◀

▶

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion



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**Governing  
processes for  
reactive nitrogen  
compounds**

O. Hertel et al.

---

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

⏪

⏩

◀

▶

Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion



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## Governing processes for reactive nitrogen compounds

O. Hertel et al.

**Table 1.** Comparison of European emissions of  $N_r$  in Tg, using the nitrogen content of both  $NH_3$  and  $NO_x$  to analyse the overall trend development of  $N_r$  release into the atmosphere. Emission figures based on the EDGAR\* emission inventory, v4.2.

	1985		1990		1995		2000		2005	
	Tg	Share (%)	Tg	Share (%)	Tg	Share (%)	Tg	Share (%)	Tg	Share (%)
Agriculture	5.98	55.3%	5.68	54.2%	4.80	52.8%	4.76	55.3%	4.65	55.5%
Road transport	1.73	16.0%	2.09	20.0%	1.85	20.3%	1.51	17.6%	1.34	15.9%
Other mobile sources	0.36	3.3%	0.26	2.5%	0.24	2.6%	0.24	2.8%	0.26	3.1%
Power generation	1.43	13.2%	1.38	13.2%	1.31	14.4%	1.18	13.8%	1.27	15.2%
Industrial production	0.87	8.0%	0.65	6.2%	0.53	5.9%	0.51	5.9%	0.48	5.8%
Other	0.44	4.1%	0.41	3.9%	0.36	4.0%	0.40	4.6%	0.38	4.5%
Total	10.81		10.47		9.08		8.60		8.38	
			-3%		-16%		-20%		-23%	

\* Source: European Commission, Joint Research Centre (JRC)/Netherlands Environmental Assessment Agency (PBL). Emission Database for Global Atmospheric Research (EDGAR), release version 4.2. <http://edgar.jrc.ec.europa.eu>, 2011.

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

⏪

⏩

◀

▶

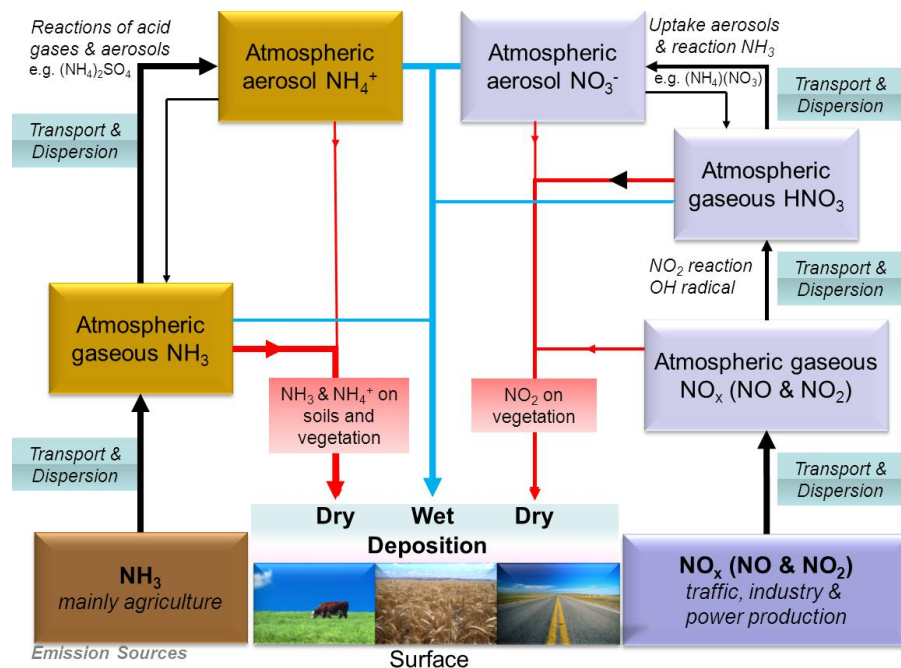
Back

Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion



**Fig. 1.** A sketch which illustrates the various pathways of reactive nitrogen compounds in the atmosphere. The left side of the figure illustrates the atmospheric pathways of  $\text{NH}_x$  (gas phase ammonia ( $\text{NH}_3$ ) and aerosol phase ammonium ( $\text{NH}_4^+$ )), and the right side of the figure illustrates the pathway of nitrogen oxides ( $\text{NO}_y$ ) compounds in the atmosphere ( $\text{NO}_x = \text{NO} + \text{NO}_2$  and reaction products). The sketch is a modified version of the figure in Hertel et al. (2006).

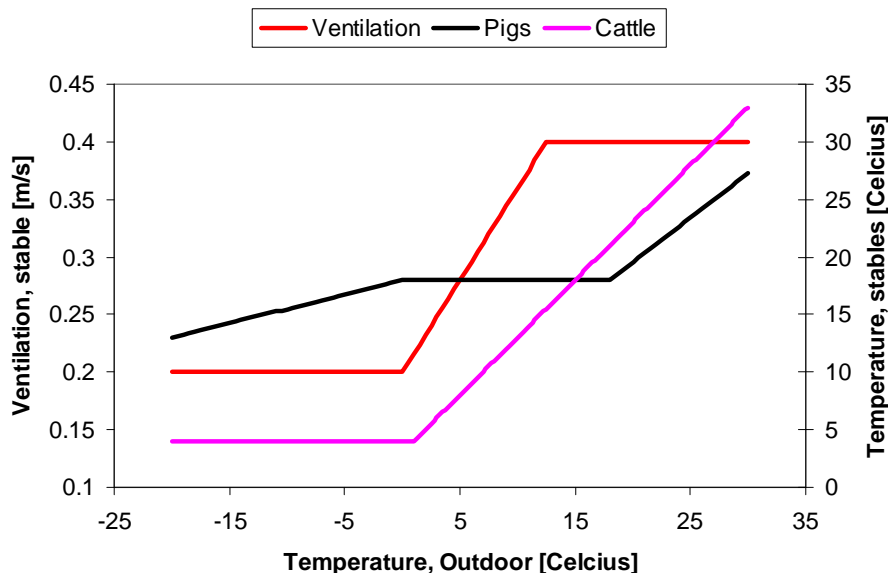
**Governing processes for reactive nitrogen compounds**

O. Hertel et al.

Title Page	
Abstract	Introduction
Conclusions	References
Tables	Figures
◀	▶
◀	▶
Back	Close
Full Screen / Esc	
Printer-friendly Version	
Interactive Discussion	

Governing processes for reactive nitrogen compounds

O. Hertel et al.



**Fig. 2.** The indoor temperature as function of outdoor ambient temperature for isolated stables (black) and open barns (pink). Isolated stables have a less steep increase in indoor temperatures because the ventilation rate (red) in these stable systems depends on outside temperatures.

Discussion Paper | Discussion Paper | Discussion Paper | Discussion Paper | Discussion Paper

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

⏪

⏩

◀

▶

Back

Close

Full Screen / Esc

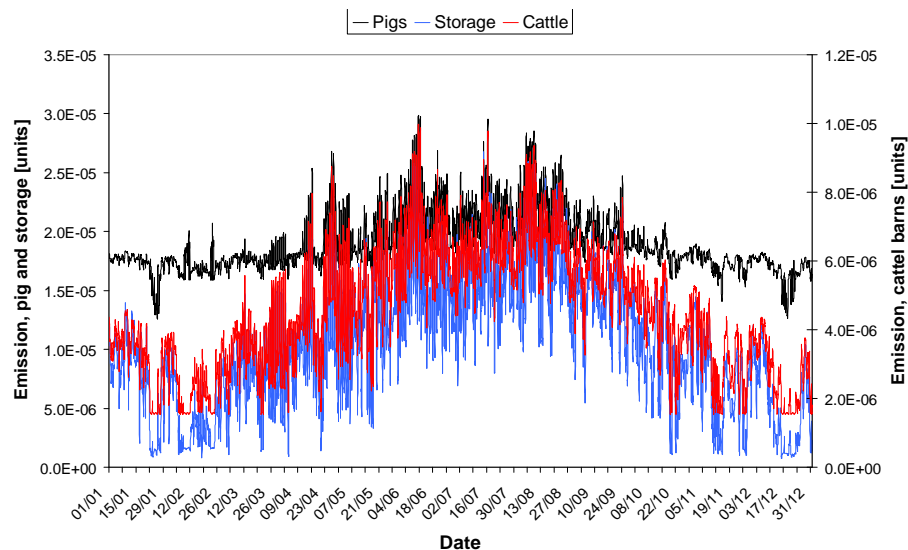
Printer-friendly Version

Interactive Discussion



## Governing processes for reactive nitrogen compounds

O. Hertel et al.



**Fig. 3.** Simulations of ammonia emission for open and isolated stables and manure storage using hourly meteorological input of temperature and wind speed. All time series were created by using meteorological data for the year the year 2007 and the emission model available at <http://www.atmos-chem-phys.net/11/5221/2011/acp-11-5221-2011.html>.

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

⏪

⏩

◀

▶

Back

Close

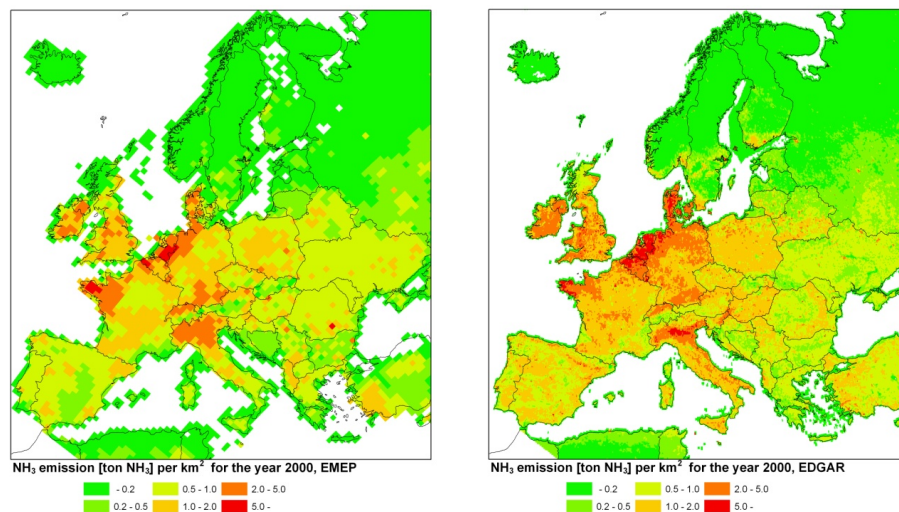
Full Screen / Esc

Printer-friendly Version

Interactive Discussion

## Governing processes for reactive nitrogen compounds

O. Hertel et al.



**Fig. 4.** Spatial distributions of annual  $\text{NH}_3$  emissions in Europe for the year 2000 based on: (left EMEP at  $50 \text{ km} \times 50 \text{ km}$  to the right) EDGAR at  $0.1^\circ \times 0.1^\circ$ .

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

⏪

⏩

◀

▶

Back

Close

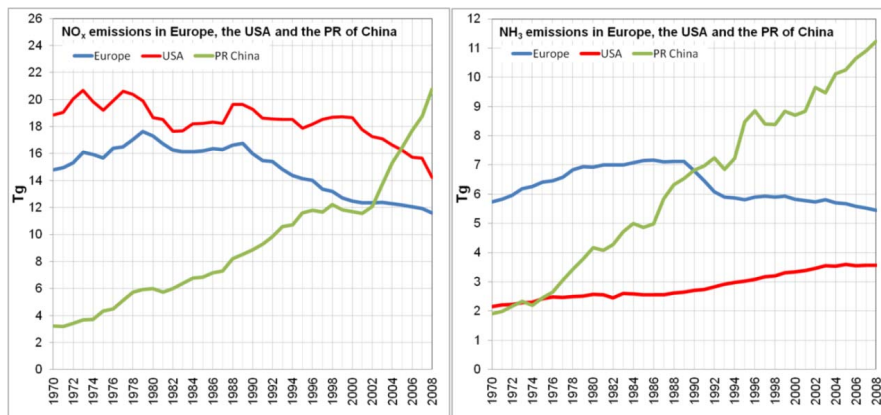
Full Screen / Esc

Printer-friendly Version

Interactive Discussion

## Governing processes for reactive nitrogen compounds

O. Hertel et al.



**Fig. 5.** Development of emissions of nitrogen oxides (left) and ammonia (right) in three world regions (Europe, USA, PR China) between 1970 and 2008 based on the Emissions Database for Global Atmospheric Research (EDGAR) version 4.2 (Unit: Tg). For a more detailed analysis of N<sub>x</sub> emission trends, see Reis et al. (2009). Source: European Commission, Joint Research Centre (JRC)/Netherlands Environmental Assessment Agency (PBL). Emission Database for Global Atmospheric Research (EDGAR), release version 4.2. <http://edgar.jrc.ec.europa.eu>, 2011

Title Page

Abstract

Introduction

Conclusions

References

Tables

Figures

⏪

⏩

◀

▶

Back

Close

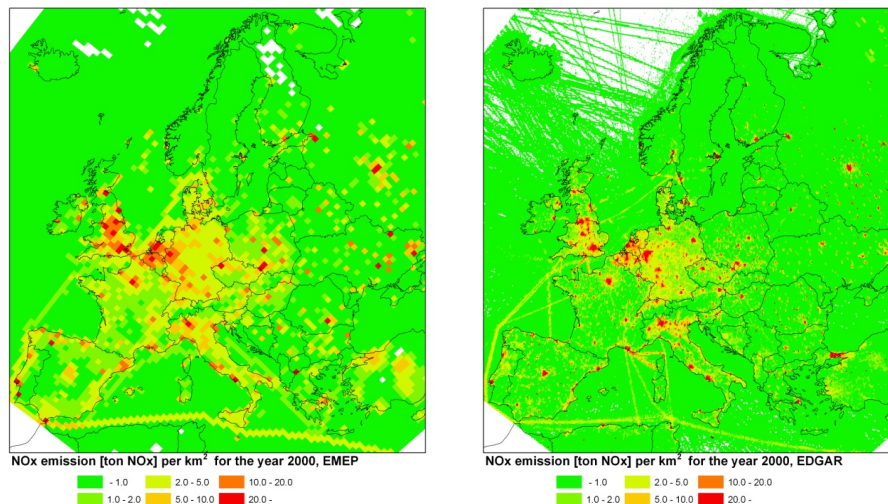
Full Screen / Esc

Printer-friendly Version

Interactive Discussion

**Governing processes for reactive nitrogen compounds**

O. Hertel et al.



**Fig. 6.** Spatial distributions of annual NO<sub>x</sub> emissions in Europe for the year 2000 based on: (left EMEP at 50 km × 50 km to the right) EDGAR at 0.1 × 0.1°.

Title Page

Abstract Introduction

Conclusions References

Tables Figures

⏪ ⏩

◀ ▶

Back Close

Full Screen / Esc

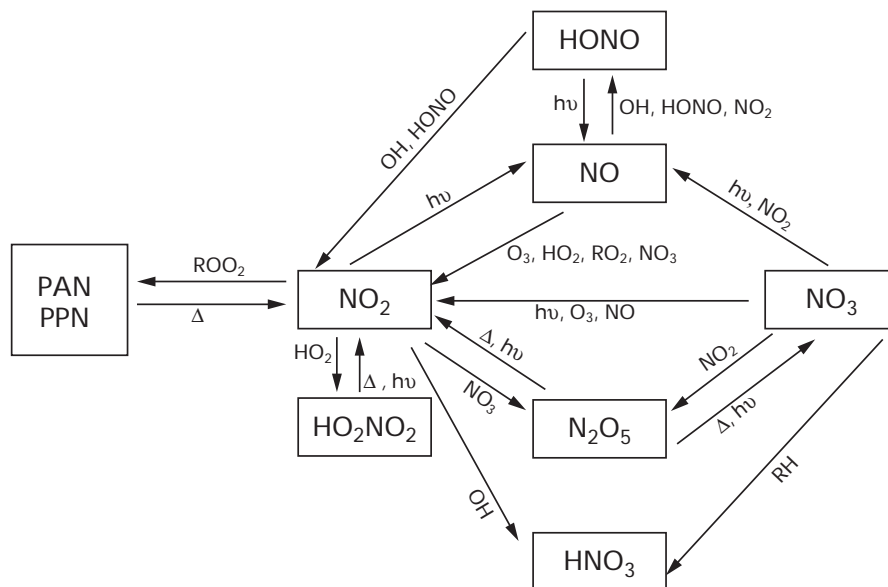
Printer-friendly Version

Interactive Discussion

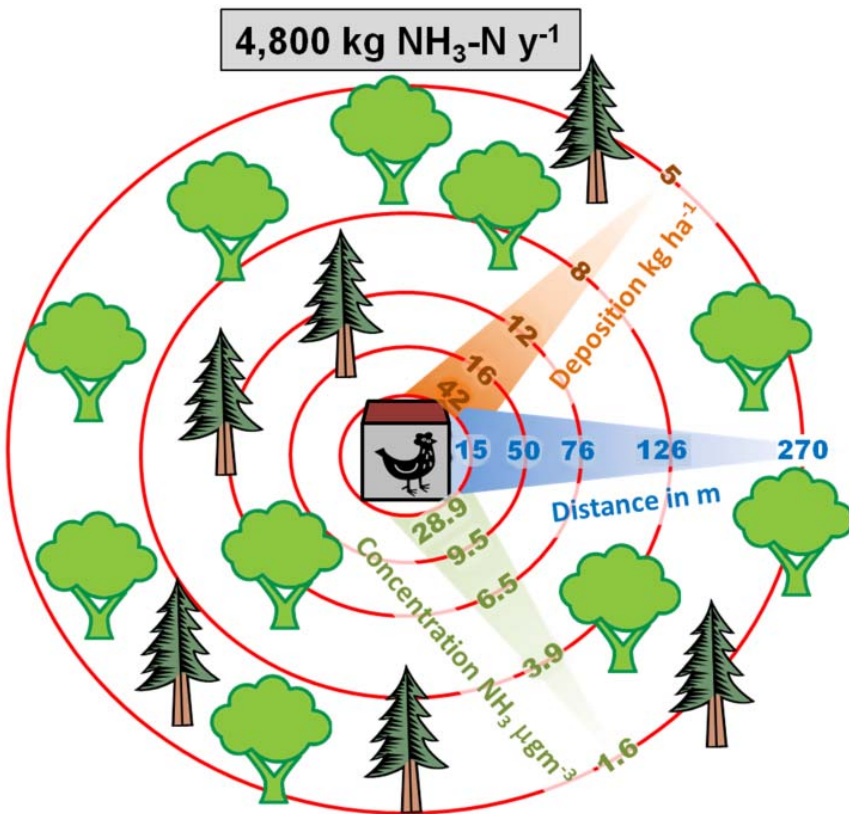


## Governing processes for reactive nitrogen compounds

O. Hertel et al.



**Fig. 7.** Illustration of the interaction between the various nitrogen oxide (NO<sub>y</sub>) compounds in the tropospheric boundary layer. The symbol  $\Delta$  represents energy leading to thermal degradation,  $h\nu$  solar radiation leading to photo dissociation and RH a hydrocarbon reacting with the species in question. PPN is a notation for other peroxy nitrates than PAN. Source (Derwent and Hertel, 1998).

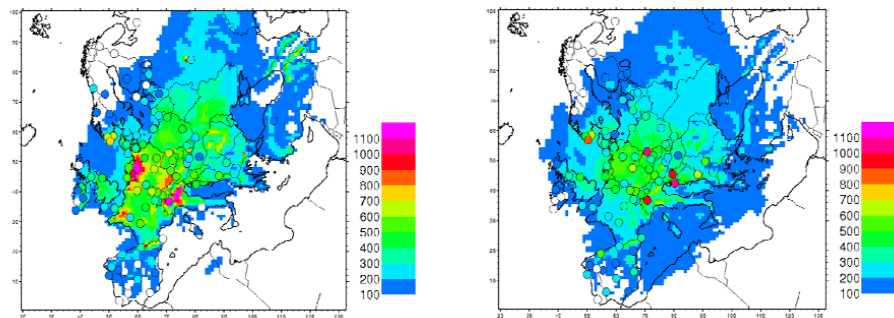


**Fig. 8.** Farm scale NH<sub>3</sub> emission and deposition, illustrating the rapid fall-off in deposition levels with distance from source (adapted from Fowler et al., 1998 and modified from version presented in Simpson et al., 2011).

**Governing processes for reactive nitrogen compounds**

O. Hertel et al.

Title Page	
Abstract	Introduction
Conclusions	References
Tables	Figures
◀	▶
◀	▶
Back	Close
Full Screen / Esc	
Printer-friendly Version	
Interactive Discussion	



**Fig. 9.** Comparison of modelled and observed annual wet deposition of to the left  $\text{NH}_x$  and to the right  $\text{NO}_3^-$  ( $\text{HNO}_3$  and aerosol phase  $\text{NO}_3^-$ ). Data are for 2001 in the EMEP model with observations. The bullets depict observations with the same colour bas as the modelled field. Measured annual deposition is calculated by using the measured precipitation amount and the  $\text{NO}_3^-$  and  $\text{NH}_4^+$  concentration in precipitation. Source: Simpson et al. (2011).

**Governing processes for reactive nitrogen compounds**

O. Hertel et al.

Title Page

Abstract Introduction

Conclusions References

Tables Figures

⏪ ⏩

◀ ▶

Back Close

Full Screen / Esc

Printer-friendly Version

Interactive Discussion

