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## Governing processes for reactive nitrogen compounds in the European atmosphere

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**Abstract.** Reactive nitrogen ( $N_r$ ) compounds have different fates in the atmosphere due to differences in the 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 the 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 generally 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 an aerodynamic 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. Further studies are needed to characterise the sources, air chemistry and removal rates of organic N emissions.

## 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-Bahl et al., 2011b; 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 ammonia ( $NH_3$ ). Strong  $NH_3$  sources contribute to high N loads to nearby ecosystems through a fast dry deposition rate (dry deposition velocities are typically  $5\text{--}20\text{ mm s}^{-1}$ , Flechard et al., 2011, but may approach  $40\text{ mm s}^{-1}$ , Phillips et al., 2004). This process competes with reactions that lead to the formation of aerosol phase ammonium ( $NH_4^+$ ). The latter contributes to  $PM_{2.5}$  and  $PM_{10}$  (mass of aerosols with an aerodynamic diameter less than 2.5 and  $10\text{ }\mu\text{m}$ , respectively) with much longer transport distance and with potential health effects and impact on the radiation balance. The  $NH_4^+$ -containing aerosols are typically assigned low dry deposition velocities (in the order of  $1\text{ to }10\text{ mm s}^{-1}$ , Duyzer, 1994), but uncertainties are large (e.g. Flechard et al., 2011) and there is increasing recognition of the role of chemical divergence (dissociation of  $NH_4NO_3$ ) in producing apparently high deposition velocities (Fowler et al., 2009). In some cases ammonium aerosols can be transported up to 1000 km away, since the aerosols 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 in Europe of about 24 h (according to model calculations with OPS, Johannes van Jaarsveld, personal communication). 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 de-

scribed 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 scavenging process is therefore by chemical conversion. In the case of  $NO_2$ , chemical reaction with hydroxyl (OH) radical forms nitric acid ( $HNO_3$ ) with a typical transformation rate of about 5 % per hour (Hertel, 1995).  $HNO_3$  has a short atmospheric lifetime (on the order of seconds to hours), since it is quickly scavenged by uptake in aerosols, reaction with  $NH_3$  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  $HNO_3$  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 thus shown lifetimes of 7–20 min in air masses processed by clouds containing ice-crystals (von Kuhlmann and Lawrence, 2006). Another US study showed that  $HNO_3$  and PAN concentrations in the urban areas of Houston, Texas were of similar magnitude and could be up to 1.5 ppbv during daytime (Luke et al., 2010). Uptake of  $HNO_3$  into existing aerosols or formation of new aerosols by reaction with  $NH_3$  leads to aerosol bound nitrate ( $NO_3^-$ ) as does the reaction with sea salt (Pryor and Sorensen, 2000). Such  $NH_4^+$  and  $NO_3^-$  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  $NH_4^+$  and  $NO_3^-$  may under certain circumstances – depending on humidity and temperature – be released back to gas phase  $NH_3$  and  $HNO_3$ . The greenhouse gas nitrous oxide ( $N_2O$ ) is excluded in Fig. 1, as it has little impact on atmospheric N deposition. To maintain simplicity of the diagram,  $N_r$  reservoir compounds (e.g. HONO,  $HO_2NO_2$ , 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  $N_r$ . Another example of atmospheric processes affecting the impact of  $N_r$  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  $NO_x$  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 which has little direct impact on human health and the airway irritant  $NO_2$  that, for example, may enhance the impact of airborne allergens (Tunnicliffe et al., 1994). As can be seen from this short introduction, the atmospheric chemistry and governing processes removing  $N_r$  from the atmosphere are both complex and diverse, reflecting the many compounds involved and their different characteristics. To be

**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, 4.2.

	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.

able to relate these processes more effectively, the present paper provides an overview of the governing atmospheric processes of  $N_r$  in the atmosphere, with particular attention to the nature of the emissions and removal processes, including bi-directional exchange where this occurs. This review further develops 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  $N_2O$  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-Bahl et al., 2011b).

## 2 Emissions

$N_r$  is mainly released to the atmosphere as  $NH_3$ ,  $NO_x$ , and in poorly quantified amounts of other gases such as HONO or organic N (e.g. amines). By comparison, emission of  $N_2O$  accounts for a much smaller fraction, representing about 15 % of the total European  $N_r$  emission to the atmosphere (Leip et al., 2011). An overall distribution of the emission of reactive nitrogen compounds and how these have evolved during the period 1985 to 2005 is shown in Table 1.

### 2.1 Ammonia emissions

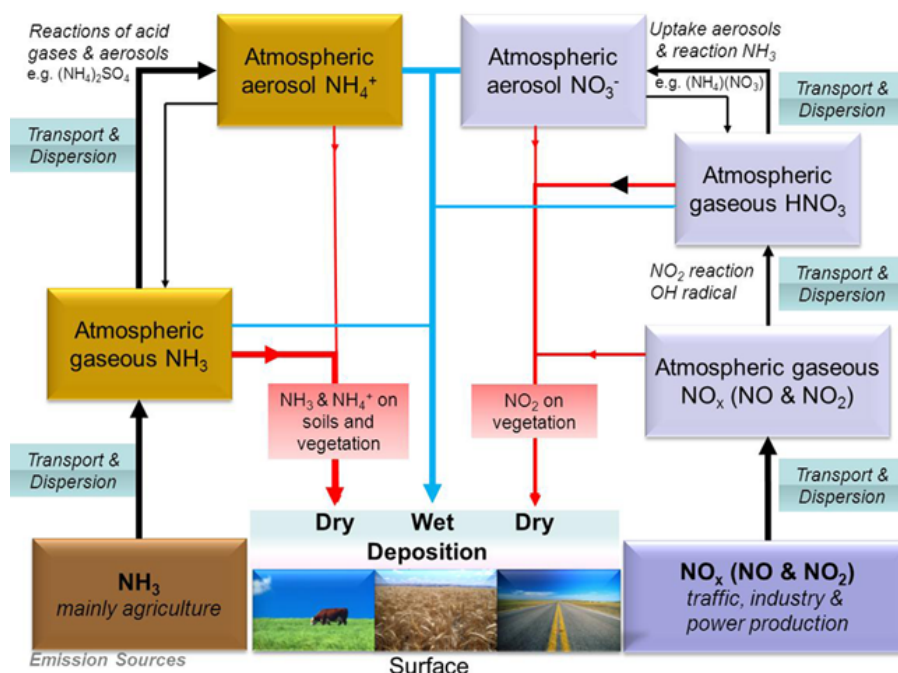
In western countries, agricultural activities contribute 85–98 % of the atmospheric  $NH_3$  emissions (Anderson et al., 2003; Gyldenkarne et al., 2005; Sutton et al., 2000a). These emissions arise in some cases from the direct breakdown of mineral fertiliser, but more importantly from the breakdown of organic matter (such as urea) within animal wastes. The emission process is a chemical and physical process which is highly temperature dependent and varies significantly over day and season (Gyldenkarne 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., 2009);
- application of manure and mineral fertiliser 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 (Gyldenkarne et al., 2005) and vegetation during management and senescence (Sutton et al., 2000b, 2009a, b).

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);





**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 NH<sub>x</sub> (gas phase ammonia (NH<sub>3</sub>) and aerosol phase ammonium (NH<sub>4</sub><sup>+</sup>)), and the right side of the figure illustrates the pathway of nitrogen oxides (NO<sub>y</sub>) compounds in the atmosphere (NO<sub>x</sub> = NO + NO<sub>2</sub> and reaction products). The sketch is a modified version of the figure in Hertel et al. (2006). Reservoir compounds nitrous acid (HONO), peroxy nitric acid (HO<sub>2</sub>NO<sub>2</sub>), PAN (CH<sub>3</sub>C(O)OONO<sub>2</sub>) and PAN-like species like isoprene nitrates as well as nitrate (NO<sub>3</sub>) and dinitrogen pentoxide (N<sub>2</sub>O<sub>5</sub>) radicals have been disregarded in the sketch for simplicity.

- manufacturing processes such as production of fertiliser, glass wool, catalysts and cement (Reis et al., 2009) and <http://edgar.jrc.ec.europa.eu/index.php>;
- humans, pets, landfills, urban areas and sewage systems (Reche et al., 2012; Sutton et al., 2000a);
- non-anthropogenic sources including natural fires from ecosystems (Andreae and Merlet, 2001; Simpson et al., 1999; Yokelson et al., 2007);
- 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 is used in a parameterisation that for a number of years has been considered as the most advanced description of temporal and spatial variations in NH<sub>3</sub> emissions in Europe (Pinder et al., 2006; Zhang et al., 2008). This parameterisation was originally developed for Denmark (Gyldenkerne et al., 2005; Skjøth et al., 2004) for which detailed information about animal numbers, manure management and crop growth is available on farm and

field level. Recent studies have shown that these methodologies can be used in large scale models over both USA (Bash et al., 2012; Cooter et al., 2012) and from Central to Northern Europe (Skjøth et al., 2011). Here it should be noted that the European situation is considerably more difficult than USA. USA provides access to agricultural census data such as animal numbers and crop assignments at the county level (e.g. Cotter et al., 2012), while the available information over the European continent varies (e.g. detailed farm-level information in the Netherlands/Denmark and almost no available information from Belarus/Ukraine). A flexible methodology which is based on both the idea of splitting agricultural sources into sectors and using an open source code is therefore developed within the EU FP7 project ECLAIRE (<http://www.eclaire-fp7.eu>). The methodology in ECLAIRE use the European approach as a starting point as this parameterisation 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, 2009). This

approach therefore allows for gradual replacements of single elements in the existing simple European approach with more advanced methodologies such as process based calculations of mineral fertiliser (Hamaoui-Laguel et al., 2012). We here use split of agricultural emissions into several agricultural sectors as the basis for describing  $\text{NH}_3$  emissions. 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).

### 2.1.1 Animal houses and manure storages

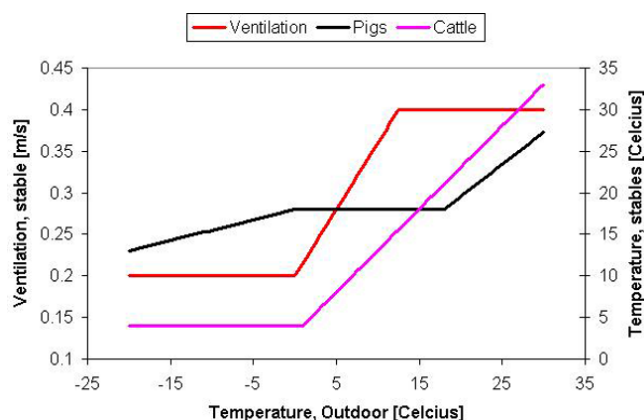
Highly complex surface models have been developed for describing  $\text{NH}_3$  emissions from agricultural buildings (Muck and Steenhuis, 1982; Olesen and Sommer, 1993; Oudendag and Luesink, 1998; Zhang et al., 1994) in order to help better manage emissions. However, in chemistry-transport modelling (CTM) a simplified parameterisation based on wind speed and temperature has proven more practical (Skjøth et al., 2004):

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

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. Equation (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 (Gyldenkerne 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 storage 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 fertiliser

$\text{NH}_3$  emissions from field application of manure and mineral fertiliser 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

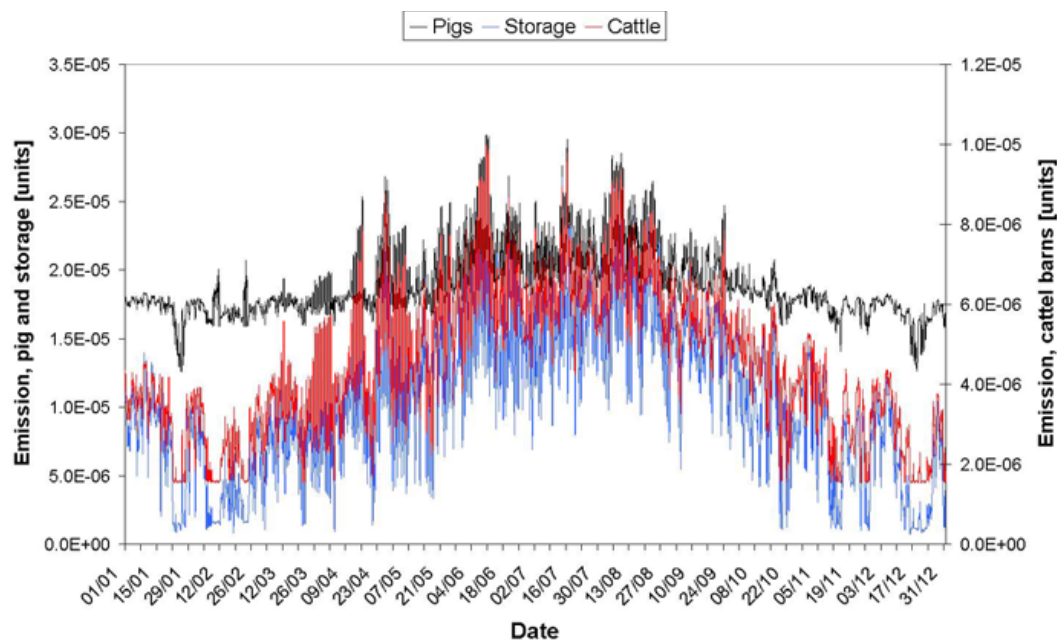


**Fig. 2.** The indoor temperature as function of outdoor ambient temperature for isolated stables (black) and open barns (pink). Isolated stables have a less steep increase in indoor temperatures because the ventilation rate (red) in these stable systems depends on outside temperatures.

direct soil injection or band-spreading methods strongly reduce emissions (Skjøth et al., 2008). National regulations can significantly alter 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. In many southern European countries animals are outdoor most of the year (see Fig. 2 in Skjøth et al., 2011), which means much less manure is collected in the barns. Also for mineral fertiliser there are regional differences, as farmers in Southern Europe start fertiliser 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



**Fig. 3.** Simulations of ammonia emission for open and isolated stables and manure storage using hourly meteorological input of temperature and wind speed. All times 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>.

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 practice (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 Other agricultural sources including plants

Legumes and plants which take up excess fertiliser emit  $\text{NH}_3$  (Larsson et al., 1998). This emission depends on the enrichment of the apoplast with  $\text{NH}_4^+$ , and the so-called compensation point (Farquhar et al., 1980), which is a function of the plant status with respect to growth, stress, etc. (this is described in more detail in Sect. 4.2 on bi-directional fluxes). This emission is still not well described with respect to magnitude, as well as temporal and spatial variation. Emissions from crops are often observed after fertilisation with either manure (Flechar et al., 2010) or mineral fertiliser (Milford et al., 2009). Grazing (Loubet et al., 2002) as well as cutting (Sutton et al., 2001) of grass is known to release  $\text{NH}_3$ . The management of the crops can heavily influence the loss of  $\text{NH}_3$  to the atmosphere. However, little is known about these processes and only few studies of N-exchange between atmosphere and vegetation cover the entire season exploring the full cycle of growth and decay (Wang and Schjørring,

2012). Several mechanistic descriptions of the compensation point have been derived (Massad et al., 2010a; Wichink-Kruit et al., 2012) and these rely strongly on detailed information on agricultural production methods which for several decades have been difficult to obtain and generalise (Hutchings et al., 2001; Skjøth et al., 2011). A global inventory estimate a contribution to global  $\text{NH}_3$  emissions from crops of  $4.4 \text{ Tg NH}_3 \text{ yr}^{-1}$  (Bouwman et al., 1997), but the magnitude of this emission is still subject to discussions and these emissions remain very uncertain.

#### 2.1.5 Wild animals

According to a global inventory wild animals contribute  $0.1 \text{ T NH}_3 \text{ yr}^{-1}$  (Bouwman et al., 1997). Simpson et al. (1999) estimated a contribution from wild mammals in Europe of about  $5 \text{ Gg N yr}^{-1}$ , much less than the estimated  $27 \text{ Gg yr}^{-1}$  from birds (of a total natural contribution of  $65 \text{ Gg N yr}^{-1}$ ). Generally, however, few studies have been carried out to quantify emissions from wild animals, and yet the perhaps best described non-agricultural sources of  $\text{NH}_3$  are probably from colonies of wild seabirds (Blackall et al., 2007; Riddick et al., 2012) 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.

**Table 2.** Available emission inventories containing data on reactive nitrogen compounds.

	Resolution	Reference	link	Comments
EMEP/CORINAIR	50 km × 50 km	Tørseth et al. (2012)	<a href="http://www.emep.int">http://www.emep.int</a>	
EDGAR	0.1° × 0.1°		<a href="http://edgar.jrc.ec.europa.eu">http://edgar.jrc.ec.europa.eu</a>	
GEIA	0.1° × 0.1°		<a href="http://gaiacenter.org">http://gaiacenter.org</a>	
GENEMIS	16.67 km × 16.67 km	Schwartz et al. (2000), Wickert et al. (2001)	<a href="http://gsf.de/eurotrac">http://gsf.de/eurotrac</a>	only 1994 data

### 2.1.6 Catalyst processes, mainly road traffic

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). An inventory from Denmark indicates that Danish road traffic contribute 2.8 % of the national  $\text{NH}_3$  emissions (Gyldenkerne and Mikkelsen, 2007). For the UK the fraction in 2010 was 3.6 % ([http://naei.defra.gov.uk/emissions/emissions\\_2010/summary\\_tables.php?action=unece&page\\_name=NH310.html](http://naei.defra.gov.uk/emissions/emissions_2010/summary_tables.php?action=unece&page_name=NH310.html)). The relative importance of European road traffic as a source of  $\text{NH}_3$  is generally increasing as other sources are going down, at the same time as European road traffic is still increasing.

### 2.1.7 Manufacturing processes

On a global scale, industrial processes have been estimated to contribute  $0.2 \text{ Tg NH}_3 \text{ yr}^{-1}$  (Bouwman et al., 1997), meaning that this is a minor part of the overall  $\text{NH}_3$  emission. This estimate fits well with a recent Danish inventory indicating a contribution of 0.8 % from manufacturing processes for Denmark (Gyldenkerne and Mikkelsen, 2007).

### 2.1.8 Humans, pets, landfill, urban sources and sewage systems

Evaporation from  $\text{NH}_4^+$  containing salts is a possible source of  $\text{NH}_3$  in Southern and Central European cities where strongly elevated levels have been found: Barcelona (Reche et al., 2012), Rome (Perrino et al., 2002) and Rijeka (Croatia) (Alebic-Juretic, 2008). Danish sewage systems have been estimated to emit  $\text{NH}_3$  of the same magnitude as has been reported for manufacturing processes (0.8 % of the national  $\text{NH}_3$  emission). In a global inventory  $\text{NH}_3$  emissions from humans and pets have been estimated to be  $3.2 \text{ Tg NH}_3 \text{ yr}^{-1}$  (Bouwman et al., 1997).

### 2.1.9 Other non-anthropogenic sources

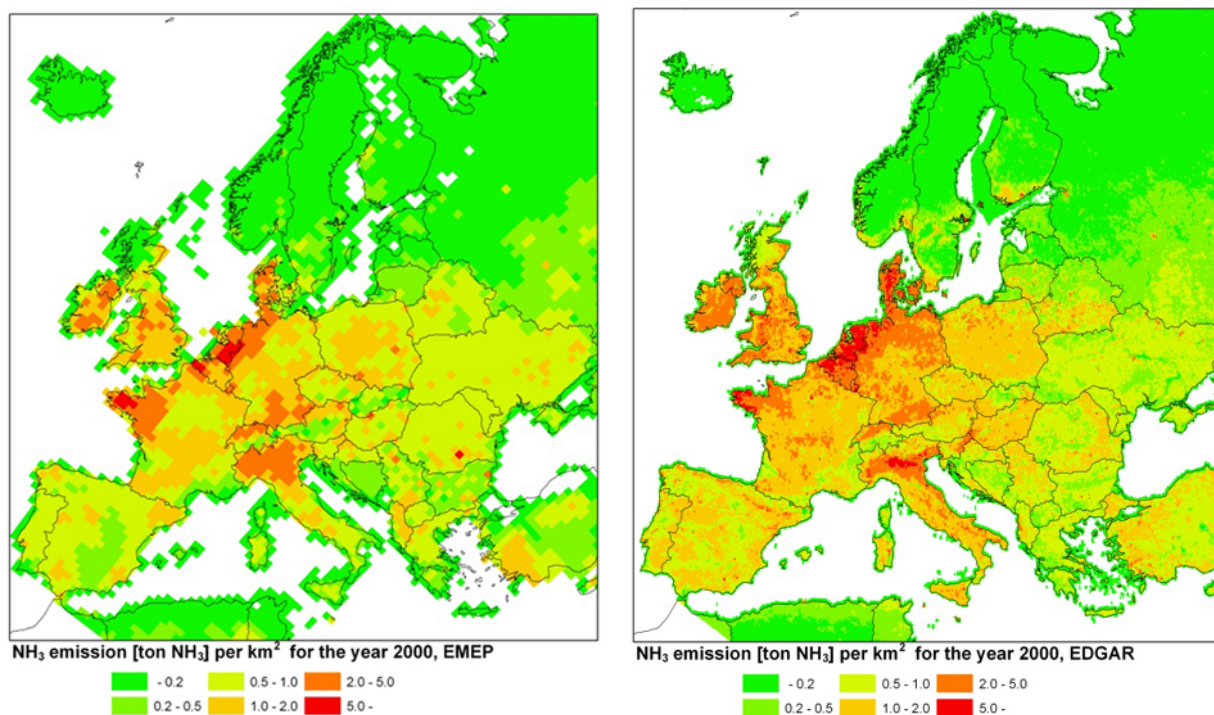
Estimates of emissions from biomass burning in a global inventory indicate that this is a relatively important source of  $\text{NH}_3$  and globally contributes  $10.3 \text{ Tg NH}_3 \text{ yr}^{-1}$  (Andreae and Merlet, 2001). A slightly older estimate is about 30 % smaller ( $7.2 \text{ Tg NH}_3 \text{ yr}^{-1}$ ) (Bouwman et al., 1997), indicat-

ing that these estimates also have significant uncertainties. Simpson et al. (1999) estimated about  $5 \text{ Gg N yr}^{-1}$  from European wildfires. Just as crops receiving excess N may emit  $\text{NH}_3$  (see Sect. 2.1.4), the vegetation in natural and semi-natural ecosystems may also release  $\text{NH}_3$  under certain conditions. This emission will mainly be important for ecosystems in the vicinity of intensive agricultural sources. Although poorly quantified, this is likely to be a minor source of emission.

### 2.1.10 Spatial distribution

Table 2 contains a listing of available emission inventories regarding reactive nitrogen compounds with information about resolution and references/links. EMEP (the Co-operative Programme for Monitoring and Evaluation of the Long-range Transmission of Air Pollutants in Europe; <http://www.emep.int>; Tørseth et al., 2012) and CORINAIR (CORE Inventory AIR emissions) have compiled inventories of annual mean emissions at the European level with a spatial resolution of  $50 \text{ km} \times 50 \text{ km}$ . 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  $0.1^\circ \times 0.1^\circ$  resolution, and EUROTRAC (EUREKA project on the transport and chemical transformation of trace constituents in the troposphere over Europe) and GENEMIS (GENERation and Evaluation of eMISSION data; <http://www.gsf.de/eurotrac>) compiled emission inventories for the year 1994 with a grid resolution of  $16.67 \text{ km} \times 16.67 \text{ km}$  (Schwarz et al., 2000; Wickert et al., 2001). The GENEMIS data have in some cases been used to redistribute EMEP emission to higher resolution for subsequent years, assuming unchanged relative distributions (Hertel et al., 2002). The need for high resolution inventories has been recognised 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



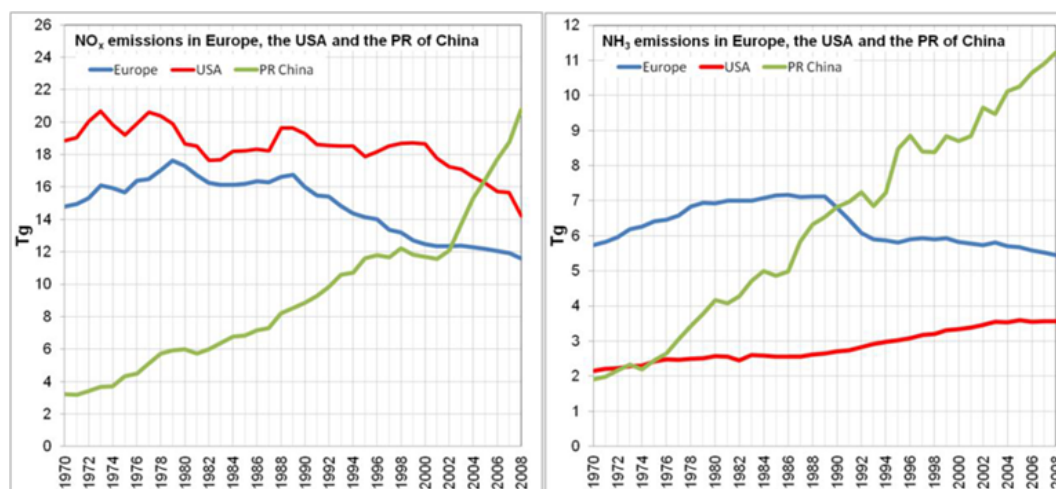


**Fig. 4.** Spatial distributions of annual  $\text{NH}_3$  emissions in Europe for the year 2000 based on: (left) EMEP at  $50 \text{ km} \times 50 \text{ km}$  to the (right) EDGAR at  $0.1^\circ \times 0.1^\circ$ .

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 appointed by the single countries. One result is that the two inventories do not agree with respect to amounts of emissions within national boundaries. Such differences can be substantial (de Vries et al., 2011a, b). 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 have been considered to contribute some of the largest uncertainties in these models (Reis et al., 2009). Recently it has been shown that the existing methodology in the European emission inventory system that rely on national emission factors and neglecting climatic effects on ammonia emissions can be substantial (Skj  th and Geels, 2012). The differences between the different emission data sets and known uncertainties (e.g. impact on emissions from meteorology and climate) 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).

### 2.1.11 Long-term trends

$\text{NH}_3$  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). Over the next decade only very limited reductions in ammonia emissions are expected for both the European domain and for individual countries (Slent   et al., 2009). The uncertainties in these trends, including the small response of  $\text{NH}_3$  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; Skj  th 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  $\text{NH}_3$  concentrations in Eastern Europe were due to a reduction in  $\text{SO}_2$  emissions (resulting in longer atmospheric lifetime of  $\text{NH}_3$ ) a factor which also influenced the  $\text{NH}_3$  trends in the Netherlands, Denmark and elsewhere. The observed trends in  $\text{NH}_3$  concentrations are therefore a result of changes in atmospheric chemistry as well as changes in  $\text{NH}_3$  emissions. Furthermore, due to the dependency on temperature, the climate change includes a climate penalty on ammonia emissions (Skj  th and Geels, 2012). This penalty and potential feedback mechanisms are currently not accounted for in neither CTMs nor climate models.



**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  $N_r$  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.

## 2.2 Nitrogen oxide emissions

$NO_x$  is generated at high temperatures in combustion processes mainly from oxidation of free atmospheric nitrogen ( $N_2$ ). The main source categories are

- road transport,
- other mobile sources including shipping,
- industrial production,
- energy production from fossil fuels,
- $NO$  emissions from soils.

In addition to these categories smaller  $NO_x$  contributions are related to aviation, and biomass and agricultural burning (2 to 3 %), and other small contributions are related waste incineration and lightning (< 1 %) (Vestreng et al., 2009). These smaller contributions are not treated further in this paper.

### 2.2.1 Emissions of $NO_x$ from road traffic

Road traffic is one of largest sources of  $NO_x$  emissions contributing 20 % to 25 % of total European  $NO_x$  emissions (Vestreng et al., 2008). The relative share of the  $NO_x$  emission from vehicles varies significantly from country to country; according to recent EMEP inventory from 4 % in Canada to 71 % for Russia (Mareckova et al., 2009). The most important sub-category for road transport is from heavy duty vehicles that (according to EMEP) contribute 9 % of the road traffic  $NO_x$  emission in Belarus but 41 % in Austria and Georgia (Mareckova et al., 2009). The  $NO_x$  emissions

from European road traffic on individual roads can be calculated using the COPERT (Computer Programme to calculate Emissions from Road Transport) model, which is available from: <http://www.emisia.com/copert/General.html>. The COPERT model contains emission factors for different vehicle categories with aging factors and driving conditions in different road types.

### 2.2.2 Other mobile sources including shipping

European  $NO_x$  emission from aircraft were generated within the EU Project QUANTIFY, and data are available from the homepage: [www.pa.op.dlr.de/quantify](http://www.pa.op.dlr.de/quantify). Shipping is one of the major sources of  $NO_x$  emissions in Europe contributing 10 to 15 % (Vestreng et al., 2008).

### 2.2.3 Industrial production

Industrial production is another important source category for  $NO_x$  emissions and contribute 10 to 15 % to European emissions (Vestreng et al., 2008).

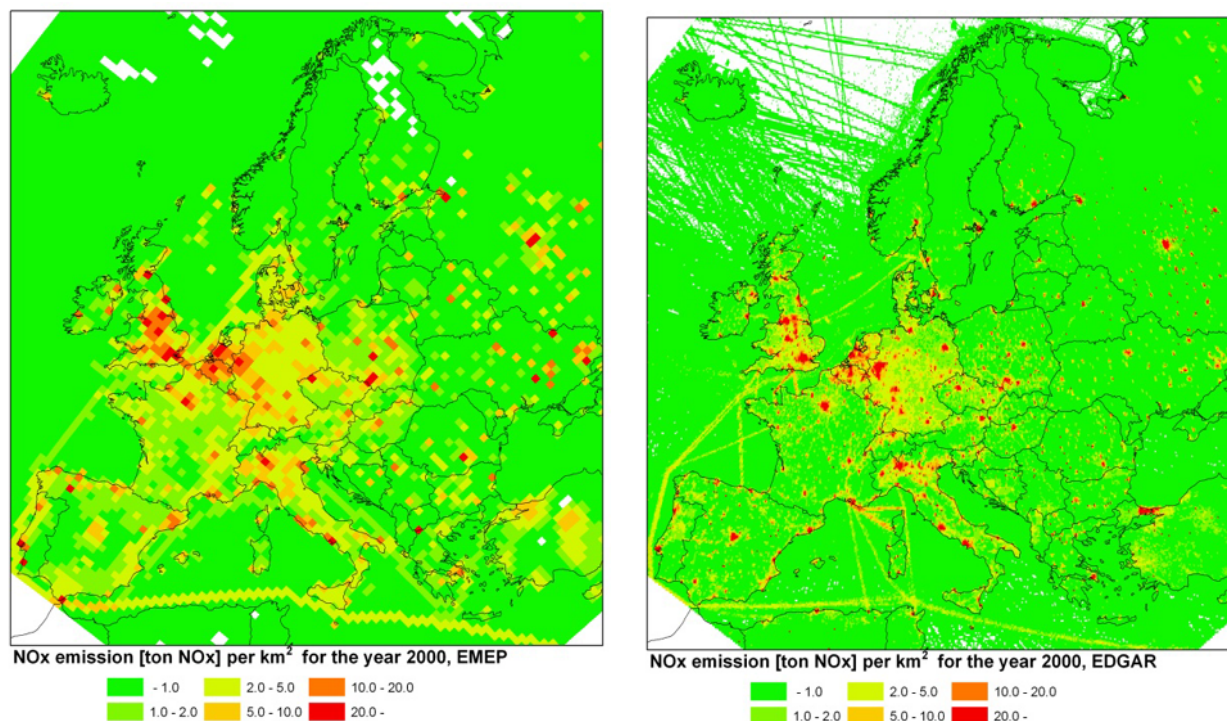
### 2.2.4 Energy production using fossil fuels

Energy production is the other major source of  $NO_x$  emissions and contributes – just like road traffic – 20 to 25 % of total European  $NO_x$  emissions (Vestreng et al., 2008).

### 2.2.5 Emissions of $NO$ from soils

In a global perspective,  $NO_x$  emissions from soil contribute > 40 % (Davidson and Kinglerlee, 1997) (IGAC Newsletter, December 2000), and > 10 % for some European countries (Butterbach-Bahl et al., 2004; Simpson et al., 1999;





**Fig. 6.** Spatial distributions of annual  $\text{NO}_x$  emissions in Europe for the year 2000 based on: (left) EMEP at  $50 \text{ km} \times 50 \text{ km}$  to the (right) EDGAR at  $0.1^\circ \times 0.1^\circ$ .

Skiba et al., 1997; Stohl et al., 1996). The soil contribution to a country's NO budget is much smaller for industrialised countries with high fossil fuel combustion rates than in countries with small fossil fuel combustion rates. This is especially the case for countries in warm dry climate zones, as these conditions favour microbial NO production (Skiba et al., 1997). Estimates of emissions of  $\text{NO}_x$  from agricultural soil varies by more than a factor of two and are mainly related to mineral and organic fertiliser applications (de Vries et al., 2011c). Emissions that are related to application of fertiliser are estimated to account for 4 % of the soil emissions (IGAC Newsletter, 2000) and up to 65 % for the USA (Hall et al., 1996). Rural agricultural areas receiving N fertilisers in countries with long dry periods are probably the largest sources of soil NO where they can be of the same order of magnitude as  $\text{N}_2\text{O}$  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 both point to European forest soils 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; Kesik et al., 2005).

## 2.2.6 Spatial distribution

Annual inventories of  $\text{NO}_x$  emissions are available from similar sources as for  $\text{NH}_3$  (Sect. 2.1), 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 large 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.

## 2.2.7 Temporal variation

Road traffic follows very regular temporal patterns. In Northern Europe the typical urban road traffic pattern includes 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  $\text{NO}_x$  (Hertel et al., 2008; Menut et al., 2012). In some parts of Southern Europe there may even be four rush hours or a generally elevated traffic flow throughout the daytime hours (Bigi et al., 2012; Vignati et al., 1996) 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 (Berkowicz et al., 1996; Bigi et al., 2012; Vignati et al., 1996). Various studies have focussed on how to change transportation habits in urban areas, encouraging bicycle use, and discouraging rush-hour commuting (Murtagh et al., 2012; Su and Zhou, 2012; Zhao et al., 2012; Zhou, 2012). A UK study in this respect showed that car commuting is socio-economically structured in several ways and, for example, that high educational groups are more likely to adjust working hours to avoid rush hours (Goodman et al., 2012). Furthermore, highways are used for transport over longer distances 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  $\text{NO}_x$  emissions in winter in Northern Europe, whereas increased demand for air conditioning in summer is expected to lead to increased  $\text{NO}_x$  emissions in summer in Southern Europe. These differences in seasonal  $\text{NO}_x$  emissions have been shown to be reflected in the seasonal pattern of ozone concentrations in the urban areas in Europe (Pires, 2012).

### 2.2.8 Long-term trends and projections

European  $\text{NO}_x$  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) (Reis et al., 2009) as a result of regulation of emissions, e.g. 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  $\text{NO}_x$  emissions from electricity generation. Such factors make projections of future trends in  $\text{NO}_x$  emissions rather uncertain. International shipping has been steadily increasing, as have the associated emissions, but reductions will result from implementation of the HELCOM convention that includes  $\text{NO}_x$  Emission Control Areas (NECA) for the Baltic Sea and the North Sea ([http://www.helcom.fi/Recommendations/en\\_GB/rec28E\\_13/](http://www.helcom.fi/Recommendations/en_GB/rec28E_13/)). In urban streets,  $\text{NO}_2$  has not followed the decreasing trend observed for  $\text{NO}_x$  (Carslaw et al., 2007; Lambrecht, 2007). The explanation is that the  $\text{NO}_2$  to  $\text{NO}_x$  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  $\text{NO}_x$  emission but with a higher  $\text{NO}_2$  fraction in the remaining emission (Carslaw and Beevers, 2005). EU limit values for  $\text{NO}_2$  are therefore currently exceeded in many European cities (Hertel and Goodsite, 2009), and this will persist until stricter EURO 6 norms are im-

plemented from 2015 (<http://eur-lex.europa.eu/LexUriServ/LexUriServ.do?uri=CELEX:32007R0715:EN:NOT>).

## 2.3 Emissions of organic nitrogen

Primary emissions of organic N are poorly understood, and much of what is measured as atmospheric organic N may have arisen from atmospheric reactions of VOCs with ammonia or oxidised N. The oceans, biomass burning and intensive animal production are a source of amines (Ge et al., 2011), but these are thought to make a small contribution ( $0.3 \text{ Tg N yr}^{-1}$ ) to overall organic N emissions on a global scale. A recent model has estimated global annual emissions of organic N as  $27 \text{ Tg N}$ , of which around half is attributed to human activities, but with very large uncertainty (range 2 to  $95 \text{ Tg N}$ ) (Kanakidou et al., 2012).

## 3 Transformations

In the following a description of the most important chemical transformation processes for  $\text{N}_r$  is provided with highlights concerning where these processes play a significant role.

### 3.1 $\text{NH}_x$ chemistry in the atmosphere

New aerosol particles are formed in gas phase reactions between  $\text{NH}_3$  and gas phase acids. However,  $\text{NH}_3$  may also deposit onto existing atmospheric particles. Whenever sulphuric acid ( $\text{H}_2\text{SO}_4$ ) is present,  $\text{NH}_3$  practically always reacts with this compound in an irreversible process forming aerosol phase ammonium bisulphate ( $\text{NH}_4\text{HSO}_4$ ) and ammonium sulphate ( $(\text{NH}_4)_2\text{SO}_4$ ). Naturally this reaction depends on the availability of  $\text{H}_2\text{SO}_4$  which is currently decreasing in concentrations in Europe – this is, for example, illustrated in measured sulphate contents in precipitation in Estonia that has decreased up to a factor of four over the time period 1994 to 2005 (Treier et al., 2008). Early experiments (1960s) showed that 50 % of the available  $\text{NH}_3$  was converted to  $\text{NH}_4^+$  in about 35 min at concentration levels in those days ( $20 \mu\text{g m}^{-3} \text{ SO}_2$  and  $2.7 \mu\text{g m}^{-3} \text{ NH}_3$  in UK in the 1960s) (Mckay, 1971).  $\text{H}_2\text{SO}_4$  is formed from gas phase oxidation of  $\text{SO}_2$  by hydroxyl (OH) radical or from aerosol phase conversion by hydrogen peroxide ( $\text{H}_2\text{O}_2$ ) and ozone ( $\text{O}_3$ ). The latter process is pH dependent, and may be catalysed by  $\text{NH}_3$ , since uptake of  $\text{NH}_3$  increases the pH of aerosols (ApSimon et al., 1994; Junge and Ryan, 1958).

The rate of conversion of gas phase  $\text{SO}_2$  and  $\text{NH}_3$  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 (RH – Relative Humidity), the limiting factor for the transformation is the molecular diffusion of  $\text{NH}_3$  to the acid particles, whereas at low RH only between 10 and 40 % of the collisions between  $\text{NH}_3$  gas molecules and  $\text{H}_2\text{SO}_4$ -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  $\text{NH}_3$  (Daumer et al., 1992). Whereas the  $\text{NH}_3$  reaction with  $\text{H}_2\text{SO}_4$  may generally be considered as irreversible, 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), but gaseous  $\text{NH}_3$  may also result from evaporation of cloud and rain droplets.

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 a major secondary pollutant from oxidation of  $\text{NO}_x$ . Measurements of fluxes over forests have indicated  $\text{HNO}_3$  concentrations in the range of 600 to 700 pptv (Pryor and Klemm, 2004). Changes in temperature and/or humidity will lead to changes in the partitioning between gas phase  $\text{NH}_3/\text{HNO}_3/\text{HCl}$  and aerosol phase  $\text{NH}_4\text{NO}_3/\text{NH}_4\text{Cl}$ , with increasing humidity and decreasing temperature moving the partitioning towards the aerosol phase compounds.

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 atmospheric acidity in response to overall growth in  $\text{NO}_x$  and  $\text{SO}_2$  anthropogenic emissions (Legrand et al., 2002); global  $\text{NO}_x$  and  $\text{SO}_x$  emissions are still increasing whereas European emissions are decreasing. New measurement data indicate that in NW Europe during summertime, HCl concentrations are similar (in terms of mixing ratio) to those of  $\text{HNO}_3$  (Schaap et al., 2011). However, due to its higher volatility,  $\text{NH}_4\text{Cl}$  concentrations are generally much lower than  $\text{NH}_4\text{NO}_3$  concentrations. HCl is emitted from anthropogenic sources including waste incineration and domestic coal burning in Europe which was historically also a significant source of  $\text{NH}_x$  in the atmosphere (Sutton et al., 2008), but is also released in displacement reaction in sea spray particles when these take up  $\text{HNO}_3$  (Wall et al., 1988). HCl is not carried by most chemical transport models (CTMs). However, the assumption that the measured HCl is mainly produced by sea-salt reaction with  $\text{HNO}_3$  needs

to be reconciled with CTMs which tend already to under-predict  $\text{NO}_3^-$  levels. This displacement is still most likely the explanation for HCl concentrations of up to 250 pptv observed in the marine boundary layer (Harris et al., 1992). Measurements and model calculations for the Baltic Sea have indicated  $\text{HNO}_3$  concentrations in the range around 300 to 600 pptv (de Leeuw et al., 2003). 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 therefore been given considerable scientific attention as, for example 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).

$\text{NO}_3^-$  is found in coarse as well as fine mode aerosol as a result of the displacement reaction, whereas  $\text{NH}_4^+$  is present predominantly 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 (coarse mode particles have higher deposition velocities).

### 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; only 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 (Carslaw, 2005). However, in the tropospheric boundary layer the distribution between NO and  $\text{NO}_2$  is governed to a large degree by  $\text{O}_3$  that reacts quickly with NO to form  $\text{NO}_2$ . In sunlight  $\text{NO}_2$  photo-dissociates (wavelengths 290 to 420 nm) to form 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 stabilises the formed  $\text{O}_3$  molecule (Seinfeld and Pandis, 2006). These reactions all have time scales of seconds to minutes. The rate of reaction between 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 will either lead to close to complete consumption of all the  $\text{O}_3$  in the conversion of NO to  $\text{NO}_2$ , or to close the conversion of all NO to  $\text{NO}_2$  (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 (Palmgren et al., 1996). During daylight, the main fate of  $NO_2$  is to undergo photolysis, which remakes  $O_3$  and NO (Dickerson et al., 1982). This photolysis reaction has a typical rate coefficient under summer conditions in the mid-afternoon at mid-latitudes 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 remade 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). An analysis of long-term  $O_3$  trends at EMEP monitoring sites showed a mixed pattern (Torseth et al., 2012). A substantial year-to-year variability in  $O_3$  concentrations is caused by varying meteorological conditions and this variability has made it difficult to separate trends related to emission change from other effects. For the Nordic countries the data indicated a reduction over the years in the occurrence of very low concentrations. The most pronounced change in the frequency distribution was seen for the UK sites and for the sites in the Netherlands, showing a reduction in the higher values. Smaller changes are seen in Germany, while in Switzerland and Austria, no change is seen in the frequency distribution of  $O_3$ .

The hydroxyl (OH) radical initiates the oxidation of a wide range of compounds in the atmospheric boundary layer (Finlayson-Pitts and Pitts, 1986). OH is formed in the daytime in the presence of sunlight (Jenkin and Clemmshaw, 2000). The photo-dissociation of  $O_3$  leads to the formation of both  $O(^3P)$  and  $O(^1D)$  radicals; a fraction of the latter reacts with water vapour to form two OH radicals. This reaction is in competition with  $O(^1D)$ 's reaction with 3rd body  $O_2$  or  $N_2$  molecules to form  $O(^3P)$ , which in turn reforms  $O_3$ . The OH radicals initiate most of the degradation of hydrocarbons in the atmosphere, a chain of reactions that, for example, lead to the formation of high  $O_3$  concentrations during summer. 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 (Seinfeld and Pandis, 2006). 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 to the CO oxidation, leading to formation of alkyl peroxy rad-

icals (Jenkin and Clemmshaw, 2000) that subsequently convert NO to  $NO_2$  in the same way as the  $HO_2$  radical does. 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 (concentrations  $> 1 \text{ ppmv}$ ), e.g. inside the exhaust pipe of vehicles and inside emitting chimneys, a 3rd order reaction can take place between two NO molecules and  $O_2$  with a reaction rate coefficient of  $2.3 \times 10^{-38} \text{ cm}^6 \text{ molecules}^{-2} \text{ s}^{-1}$  (about  $3.7 \times 10^{-59} \text{ ppbv}^{-2} \text{ s}^{-1}$ ) (Hampson and Gavin, 1978) to give off  $NO_2$ . The further transformation of  $NO_2$  by reaction with OH to give  $HNO_3$  takes place with a typical rate of about 5 % per hour in the troposphere.

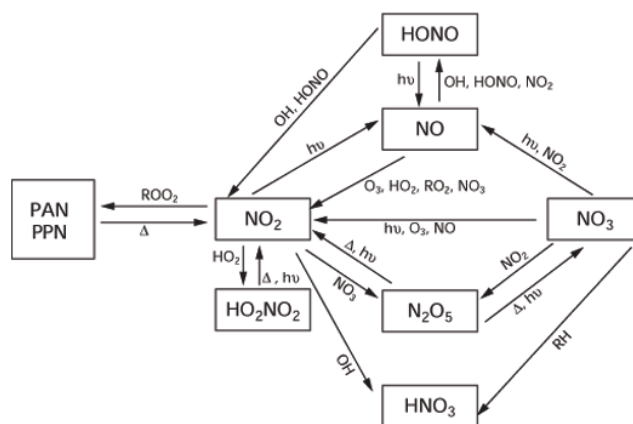
At night, the nitrate ( $NO_3$ ) radical has a similar but less important role in the degradation of hydrocarbons in the atmosphere to that of OH radical in daytime (for an extensive review of the chemistry, see Wayne et al. (1991)). Despite the considerably lower reactivity compared with OH, its higher peak concentrations in the nighttime troposphere allow the  $NO_3$  radical to play a major role in the transformation of organic compounds. The  $NO_3$  radical is formed during nighttime from the reaction of  $O_3$  with  $NO_2$ . Dinitrogen pentoxide ( $N_2O_5$ ) formed in the reaction of  $NO_2$  with  $NO_3$  is a reservoir compound for the  $NO_3$  radical at low temperatures, but it is broken down to its precursors  $NO_2$  and  $NO_3$  at higher temperatures in the dark. The typical nighttime  $NO_3$  radical concentrations in the atmospheric boundary layer are of the order  $10^7$  to  $10^8 \text{ molecules cm}^{-3}$  (which is in the pptv 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  $NO_3$  radical and  $N_2O_5$  radical photo-dissociate so fast that the concentrations of these compounds are insignificant. In the tropospheric boundary layer the photolysis of  $NO_3$  radicals (with a typical noon lifetime of about 5 s) follows two different wavelength dependent paths forming NO and  $O_2$  (wavelength  $< 700 \text{ nm}$ ) or  $NO_2$  and  $O(^3P)$  (wavelength  $< 580 \text{ nm}$ ). Close to pollution sources from combustion processes, e.g. road traffic or power plants, the  $NO_3$  radical is quickly removed by reaction with NO leading to formation of two  $NO_2$  molecules. In urban areas, emissions of NO rapidly destroy  $NO_3$  close to ground level. However, above the surface at night  $NO_2$  and  $O_3$  can exist together in the absence of NO (as it has all been converted to  $NO_2$ ), leading to  $NO_3$  formation, as observed on a 160 m high tower in London (Benton et al., 2010). During the night, the heterogeneous conversion of  $N_2O_5$  to  $HNO_3$  is an important process. The lifetime of  $N_2O_5$  with respect to this removal process is of the order of minutes in the tropospheric boundary layer. This production of  $HNO_3$  may in winter be as important as daytime conversion of  $NO_2$  by OH radical. As already described, particulate nitrate ( $NO_3^-$ ) is formed when  $HNO_3$  reacts with  $NH_3$  and forms new aerosols, and when it sticks to existing particles in the atmosphere. In addition organic nitrate may be formed from gaseous  $NO_2$  on the surfaces of aerosols in

other heterogeneous reactions. The  $\text{NO}_3$  radical attacks alkanes by hydrogen abstraction in a similar way to the reactions of the OH radical, and again leading to formation of a peroxy radical ( $\text{RO}_2$ ) that may oxidise an NO molecule to  $\text{NO}_2$ . Also for alkenes, the attack of the  $\text{NO}_3$  radical is similar to the reactions of the OH radical; the  $\text{NO}_3$  radical adds to the double bond. This reaction is followed by rapid  $\text{O}_2$  addition which leads to the production of a peroxy radical. Reaction of soil emissions of NO to give  $\text{NO}_2$ , which then reacts with atmospheric OH has been suggested to provide an in-canopy source of  $\text{HNO}_3$  (Farmer et al., 2006).  $\text{HNO}_3$  concentrations usually peak during the day, regulated by the emissions of  $\text{NO}_x$ , photochemical activity and the gas/aerosol equilibrium of  $\text{NH}_4\text{NO}_3$  shifting towards the gas phase with increasing temperature and decreasing relative humidity (see, e.g. Andersen and Hovmand, 1994).

A reaction similar to that between  $\text{NO}_2$  and OH, but less important, is the reaction between NO and OH that leads to formation of nitrous acid (HONO) (Finlayson-Pitts and Pitts, 1986). During daytime HONO photo dissociates ( $\lambda < 400 \text{ nm}$ ) 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 (Schiller et al., 2001). Studies in the highly polluted Po Valley in Northern Italy have shown an interrelation between simultaneous peaks in  $\text{NO}_x$  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, which has also been shown in UK studies (Harrison and Collins, 1998). Probably this type of conversion plays an important role also in many urban areas over Europe, but so far only a few studies have been carried out, and according to a recent review the mechanism are still poorly understood (Indarto, 2012). 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  $\text{NO}_2$  on humid acid in soil (Stemmler et al., 2006). A recent study furthermore suggests 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 (Walker et al., 2012b). 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.



**Fig. 7.** Illustration of the interaction between the various nitrogen oxide ( $\text{NO}_y$ ) 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).

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 are 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, 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  $\text{NH}_4^+$  than with  $\text{NO}_3^-$  (Zhang et al., 2012). The overall correlation 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 (Lin et al., 2010), 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-dissociated or react with OH, an alkyl radical is formed, which in turn may form peroxy alkyl nitrates (Tanner et al., 1988) that serve as important reservoirs of NO<sub>2</sub> (Fig. 7). The most abundant of these nitrates is 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 (Gaffney et al., 1999a). Thermal degradation gives PAN a lifetime of approximately 2 h at 273 K and 50 h at 263 K. The PAN formation competes with NO degradation of peroxy acetyl radical (Finlayson-Pitts and Pitts, 1986). This reaction is usually dominant at ppbv levels of NO, meaning that PAN and other peroxy alkyl nitrates are usually formed only in the background atmosphere (Gaffney et al., 1993), but substantial PAN concentrations may still be observed in urban areas at relatively low temperatures (Gaffney et al., 1999b). The peroxy alkyl nitrates include compounds produced in a similar way to PAN, but generated from biogenic isoprene emissions that may be of importance in Southern Europe, and have similar thermal degradation pathways to PAN. PAN is an important atmospheric N reservoir species, but it is probably not an important source of DON (Cornell et al., 2003).

#### 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](http://www.emep.int)) that includes both long-term observations (Tørseth et al., 2012) and campaign studies (Aas et al., 2012), the former NitroEurope Integrated Project (NEU; [www.nitroeuropa.eu](http://www.nitroeuropa.eu); Sutton et al., 2007) the US National Atmospheric Deposition Network (NADP; <http://nadp.sws.uiuc.edu/>) (Lamb and Bowersox, 2000), the Acid Deposition Monitoring Network in East Asia (EANET; [www.eanet.cc](http://www.eanet.cc)) (Totsuka et al., 2005) and several others. However, these networks measure air concentrations rather than fluxes, and dry deposition is estimated using inferential modelling approaches (Erisman et al., 1994; Erisman and Draaijers, 1995; Meyers et al., 1998), such as using measured air concentrations and meteorology in combination with a dry deposition algorithm (e.g. Flechard et al., 2011; Skjøth et al., 2002) which are underpinned by often sparse databases of campaign based process studies or long-term monitoring in both cases 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 fluxes of each individual N<sub>r</sub> compound is therefore 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 recent inter-comparison study on NH<sub>3</sub> fluxes deploying various techniques concluded that the major challenge is still obtaining consistently reliable NH<sub>3</sub> concentrations (Milford et al., 2009).

A first European flux measurement network for reactive nitrogen compounds was established within the NitroEurope IP (Skiba et al., 2009; Sutton et al., 2007; Tang et al., 2009). This network applied 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 nine 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 presently 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 and meteorological conditions. In agricultural areas NH<sub>3</sub> may dominate the atmospheric N loading (e.g. Hertel et al., 2006), while in

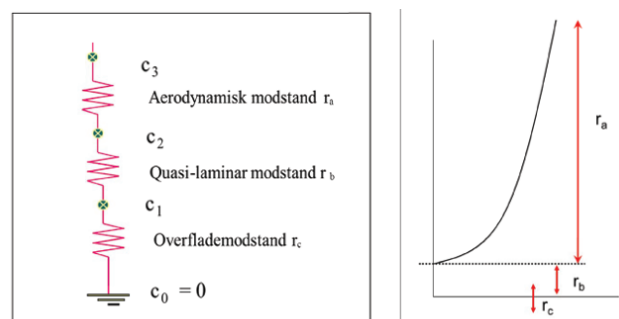
more industrial and urban areas  $\text{HNO}_3$  and  $\text{NO}_2$  may be more important. This pattern is seen even on the European scale; Simpson et al. (2007) showed that reduced nitrogen dominates dry deposition of N to forests close to the main agricultural areas, whereas oxidised nitrogen dry 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 fertilised vegetation, a net source of  $\text{NH}_3$  (Loubet et al., 2009; Sutton et al., 1993; Walker et al., 2012a). 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). Challenges in modelling  $\text{NH}_3$  fluxes over vegetation have been addressed in a number of papers (Pleim et al., 2012; Wu et al., 2009). PAN deposition has been modelled in many studies (Wu et al., 2012); the dry deposition is much faster than expected on the basis of its solubility, with dry deposition velocities in daytime that may exceed  $15 \text{ mm s}^{-1}$  and nighttime values of half this magnitude (Doskey et al., 2004; Schrimpf et al., 1996), so much is still unknown about the mechanisms of its atmospheric removal (Turnipseed et al., 2006).

#### 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 ( $R_c$ ) (see the Sketch in Fig. 8). For a complex surface with several potential absorption sinks (e.g. vegetation) the resis-



**Fig. 8.** Sketches illustrating the resistance method with the three resistances – the aerodynamic resistance, the quasi-laminar sub-layer resistance, and the surface resistance. The concentration at the surface is here assumed to be zero.

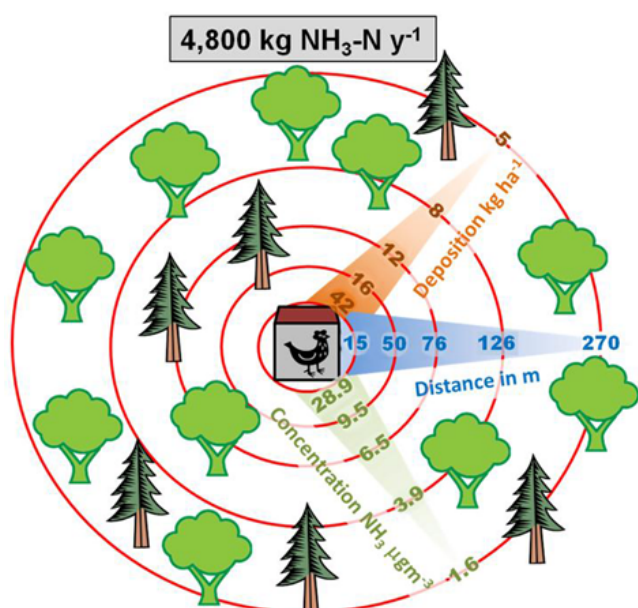
tance  $R_c$  may be viewed as a network of parallel resistances, representing transfer to the external leaf surface, through stomata, to water on the surface, or through the canopy to the underlying soil surface. The total resistance ( $R_t$ ) is the sum of all the series and parallel resistances ( $R_a + R_b + R_c$ ), 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 ( $F$ ) normalised 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 within the so-called surface layer (typically 10–100 m), 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 parameterisation 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





**Fig. 9.** Farm scale  $\text{NH}_3$  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).

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 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  $R_d$  values are smaller, and deposition velocities are larger, than to short vegetation (Cellier and Brunet, 1992). Consequently, estimating deposition of different components requires knowledge about land cover as well as the spatial pattern of air concentrations. Figure 8 illustrates farm-scale  $\text{NH}_3$  emission and deposition. 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 (Spokes and Jickells, 2005). 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 solu-

ble  $\text{NO}_2$  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  $\text{CO}_2$ , DMS (dimethyl sulphide) and  $\text{CH}_4$ .

The surface resistance ( $R_c$ ) is a key parameter for the deposition of a gas and several parameterisation schemes have been derived from flux measurements. For submicron 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 strongly 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 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 (1982) predict low deposition velocities to forest canopies. Alternative formulae of 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. (1997) 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 all surfaces, and especially 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

$N_r$  deposition. For other  $N_r$  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 of uptake increases with the water solubility and Henry's Law coefficient of the gas, which vary over 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

Due to its high deposition rate,  $HNO_3$  makes a significant contribution to  $N_r$  deposition in regions with high  $NO_x$  emissions.  $HNO_3$  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  $HNO_3$  have been observed, probably due to non-zero  $HNO_3$  surface concentrations in equilibrium with  $NH_4NO_3$  deposited to leaf surfaces (Neftel et al., 1996; Nemitz et al., 2004b; Zhang et al., 1994). 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  $HNO_3$  deposition is subject to a surface resistance, and this remains a research priority.

#### 4.2.2 Ammonia

$NH_3$  is less water soluble than  $HNO_3$ , and  $NH_3$  previously absorbed to wet leaf surfaces may more readily be desorbed (re-emitted) as leaf water layers dry out again provided that it 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  $NH_4^+$  and  $NO_3^-$ . These nutrients are mainly present in the liquid part (apoplast) between the cells of the plant.  $NH_4^+$  is an important by-product of plant biochemical pathways resulting in non-zero  $NH_4^+$  concentrations in the leaf apoplast, which results in non-zero gas-phase concentrations (stomatal compensation points,  $\chi_s$ ) in equilibrium with this  $NH_{4\text{apo}}^+$  concentration at the apoplastic pH (Farquhar et al., 1980; Massad et al., 2010b; Mattsson et al., 2009; Schjörriing et al., 1998; Sutton et al., 1995).

Available evidence suggests that  $NH_{4\text{apo}}^+$  increases with increasing N supply to the plant, either through fertilisation or high atmospheric N inputs (Massad et al., 2010b; Sutton et al., 1995). The system here is an exchange governed by the pH-dependent relation between  $NH_3$  and  $NH_4^+$ . The compensation point  $\chi_s$  is the product of a

temperature function describing the Henry's Law equilibrium and the ratio of  $\Gamma_s = [NH_{4\text{apo}}^+]/[H_{\text{apo}}^+]$ . The Henry's law coefficient for  $NH_3$  ( $\text{mol atm}^{-1}$ ) may be expressed as  $H_{NH_3} = 56\text{EXP}(4092(1/T - 1/298.15))$  (Dasgupta and Dong, 1986). 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  $N_r$  deposition, and to  $> 10,000$  after fertilisation (Massad et al., 2010b). At  $10^\circ\text{C}$ , this equates to values of  $\chi_s$  of  $< 0.15$ , 2.3 and  $> 15\text{ }\mu\text{g NH}_3\text{ m}^{-3}$ , respectively. Emission potentials of fertilised soils and plant litter can be several orders of magnitude larger (Nemitz et al., 2000; Sutton et al., 2009a, b), allowing  $NH_3$  emitted at the ground to be recaptured by an overlying 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 it 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  $NH_3$  exchange is often difficult to estimate a priori. There are several reviews in the literature on bi-directional  $NH_3$  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 this is the main driver of apoplast and bulk leaf  $NH_4^+$  concentrations (Massad et al., 2010b; 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).

The stomatal pathway for  $NH_3$  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  $NH_3$  net exchange are particularly uncertain, due to (i) uncertainties in the overall magnitude as well as spatial and temporal patterns of agricultural  $NH_3$  emissions, and (ii) the large variability of  $NH_3$  deposition rates to different surfaces.

The development of resistance models to describe bi-directional  $NH_3$  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 simple 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 was 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). This approach was subsequently adopted in both small- and large-scale atmospheric transport models by Den-  
tner and Crutzen (1994) (Sorteberg and Hov, 1996; 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  $\text{NH}_3$  deposition when using a bidirectional scheme compared with a simple flux model allowing only  $\text{NH}_3$  deposition. Lately Wichink-Kruit et al. (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  $\text{NH}_3$  concentrations over sea areas, thus reflecting observed open sea concentrations (e.g. de Leeuw et al., 2003) to a much greater 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 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 (Neirynck 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 generalised empirical parameterisations estimated by Mas-sad 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 (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) 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

NO is rather water-insoluble and there is no efficient mechanism for 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 NO. Some of these soil emissions of NO are oxidized to  $\text{NO}_2$  (and possibly  $\text{HNO}_3$ ) within plant canopies, and taken up more efficiently than NO and thus the behaviour of NO still needs to be taken into account in surface–atmosphere exchange.

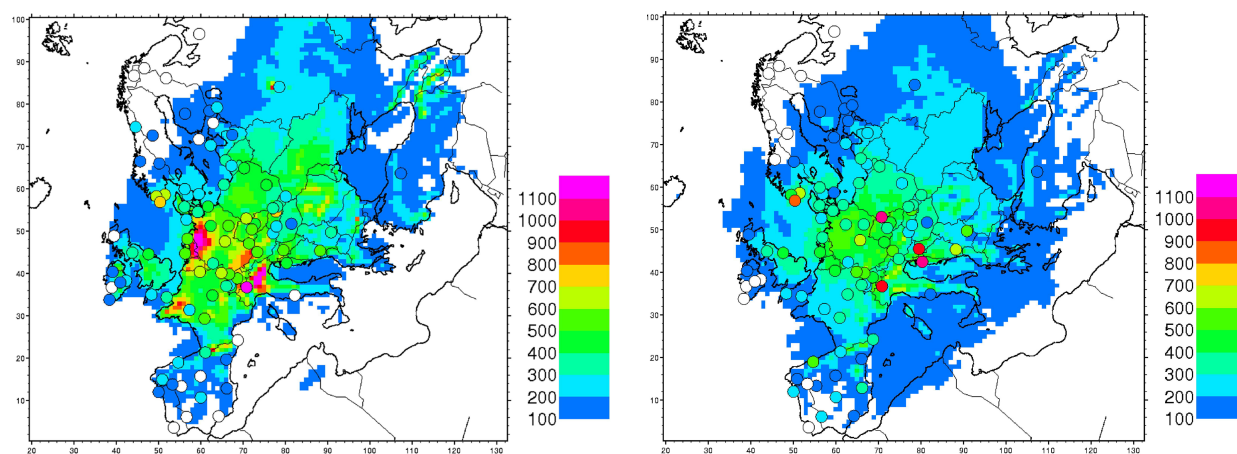
#### 4.2.4 Nitrogen dioxide

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 ppbv 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).

#### 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  $\text{NH}_3$ , 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  $\text{NO}_2$  with NO on wet surfaces (Harrison et al., 1994) or  $\text{NO}_2$  reduction on humic acid (Stemmler et al., 2006). In connection with an experimental study, a parameterisation of bi-directional fluxes of both  $\text{NH}_3$  and HONO





**Fig. 10.** 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  $\text{NO}_3^-$ ). Modelled data are from the EMEP MSC-W model, observations from the EMEP network, both for the year 2001. The bullets depict observations with the same colour base 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 (Simpson et al., 2011).

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  $\text{O}_3$ , 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 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  $\text{H}_2\text{SO}_4$ -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. Wet deposition measurements are performed in the EMEP network ([www.emep.int](http://www.emep.int)) (Tørseth et al., 2012).

### 5.1 Wet scavenging of aerosols

The bulk of the  $\text{NH}_4^+$  and  $\text{NO}_3^-$  aerosol mass is present in the size range 0.1 to  $1.0\ \mu\text{m}$  (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 rain-out. Washout is responsible for 10 to 20 % of the  $\text{N}_r$  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 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: 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  $\text{N}_\text{r}$  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).

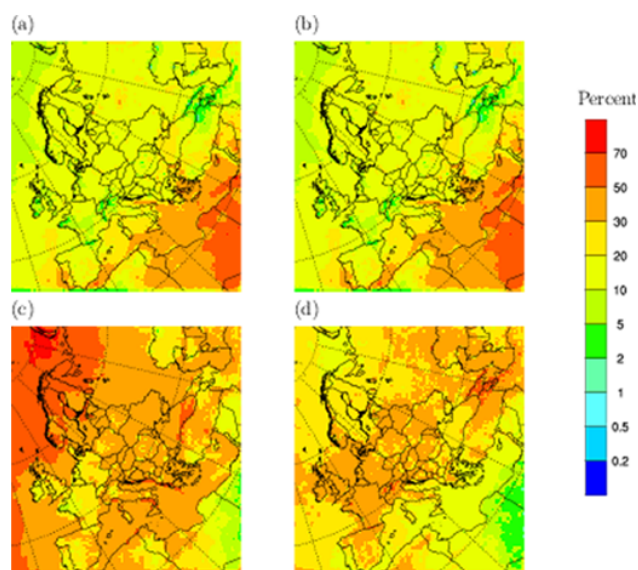
## 5.2 Distribution between dry and wet deposition

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  $\text{N}_\text{r}$  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 et al. (2006) presented calculations of the wet and dry fraction of both oxidised and reduced nitro-

gen over Europe, calculated with the EMEP MSC-W model for the year 2000. In these calculations, the dry deposition of  $\text{N}_\text{r}$  exceeded wet deposition in central and southern Europe, whereas wet deposition predominated in other areas. Figure 11 shows calculations with the latest version of the EMEP model (Simpson et al., 2012) for the year 2009, and for the grid-averaged deposition. These results (and similar for coniferous forest, not shown), reveal a different distribution for this later year. The wet deposition of oxidised  $\text{N}_\text{r}$  compared to reduced  $\text{N}_\text{r}$  is also apparent for these results. The dry deposition fraction is significantly lower in central Europe than shown in Simpson et al. (2006). Some of this difference is likely due to revisions in the model's dry deposition schemes (including land-cover characteristics) over the years, but some is also driven by reductions in the emissions of  $\text{NO}_x$  and  $\text{SO}_x$ . As discussed in (Fagerli and Aas, 2008), reduced  $\text{SO}_x$  emissions give a reduced formation of ammonium sulphate, hence allowing faster conversion of  $\text{HNO}_3$  to longer-lived particulate-nitrate. This also implies a switch from dry deposition of  $\text{HNO}_3$  to wet deposition of nitrate. In regions in which the amounts of precipitation are large, wet deposition dominates the  $\text{N}_\text{r}$  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. It should be noted however that CTMs typically cannot account for some of the more complex processes which affects  $\text{N}_\text{r}$  deposition, for example those of occult deposition or so-called orographic enhancement (both discussed below) can have a profound effect on the overall scavenging of pollutants from the atmosphere (Fowler et al., 2009).

## 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 generated (Dore et al., 1990). As



**Fig. 11.** Calculated percentage contributions to total nitrogen deposition for the year 2009: (a) dry deposition of oxidised  $N_r$ ; (b) dry deposition of reduced  $N_r$ ; (c) wet deposition of oxidised  $N_r$ ; (d) wet deposition of reduced  $N_r$ . Calculations with EMEP MSC-W model (Simpson et al., 2012).

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  $50 \text{ km} \times 50 \text{ km}$  scale fails to capture the spatial structure in wet deposition (Fig. 10; 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 has 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  $0.1\text{--}1.0 \mu\text{m}$ , which are not deposited efficiently on vegetation, the hill cloud droplets are large enough (3 to  $10 \mu\text{m}$  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 ( $\text{SO}_4^{2-}$ ,  $\text{NH}_4^+$ ,  $\text{NO}_3^-$ , 1–2 mM) (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 process in mapping regional wet deposition, to avoid underestimating wet deposition and exceedances of critical loads.

## 6 Conclusions and perspectives

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 ( $\text{NO}_x$  and  $\text{NH}_3$ ) and climatic variables (mainly  $\text{NH}_3$ ).  $N_r$  compounds are subject to rapid transformations in the atmosphere which profoundly affect their deposition rates. Thus  $\text{NH}_3$  has a fast deposition rate but it also reacts with acids to form  $\text{NH}_4^+$  salt aerosols which have low dry deposition velocities, but are liable to wet deposition. As a consequence, then  $\text{NH}_4^+$  containing aerosols are readily capable of long-range transport.  $\text{NO}_x$  emissions are mainly in the form of NO, but the emissions from road traffic contain an important fraction of primary  $\text{NO}_2$  which has increased in recent years. NO is rapidly converted to  $\text{NO}_2$ , which in turn is converted to  $\text{HNO}_3$  either through daytime reaction with the OH radical, or by heterogeneous conversion processes at night. There is a strong need for further research on heterogeneous mechanisms of oxidation of nitrogen dioxide to nitric acid. While the general principles are known, the determinants of atmospheric conversion rates are very uncertain, despite the fact that these processes are the only ones operative at night, and can make a substantial contribution to nitrate formation.  $\text{HNO}_3$  is subject to rapid deposition, but may also react to form  $\text{NH}_4\text{NO}_3$  which is subject to long-range transport.  $\text{NO}_2$  may also be converted by reaction on land and aerosol surfaces to HONO, which is photolysed to form OH radicals. Ambient air concentrations of  $N_r$  compounds are generally fairly well (often within  $\pm 20\text{--}30\%$ ) 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. There is therefore considerable need for studies of  $N_r$  fluxes, especially for sensitive ecosystems. These flux studies need to include detailed field studies, parameterisation, application and testing of models, such as CTMs, covering both the biosphere and the atmosphere such as CTMs. Furthermore, long-term trends show very different emission patterns for  $\text{NH}_3$  and  $\text{NO}_x$ , where emissions in Europe and the US show marked declines for

NO<sub>x</sub> and moderate reductions or stagnation for NH<sub>3</sub>. In contrast, emissions from China, for instance, show a steep increase over 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<sub>r</sub> that take the full cascade of effects into account are therefore urgently needed.

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