Introduction

Ammonia in the Environment

Ammonia (NH₃) emission from the biosphere to the atmosphere is one of the many unintended consequences of reactive nitrogen (Nᵢ) creation from inert dinitrogen gas (N₂) through symbiotic biological nitrogen fixation (BNF) and the
Haber-Bosch process, and of the agricultural usage of the fixed N\textsubscript{r} for crop and meat production (Sutton et al. 2011). Conversely, NH\textsubscript{3} emission is also one of the main precursors of the nitrogen cascade (Galloway et al. 2003), whereby the N atom of the NH\textsubscript{3} molecule may potentially participate in a number of environmental impacts through a series of pathways and chemical and (micro-)biological transformations in the biosphere. As airborne NH\textsubscript{3} is transported downwind from sources, chemically processed in the atmosphere, and dry- and wet-deposited to the Earth’s surface, it may be converted in air, vegetation, soils and water successively to NH\textsubscript{4}\textsuperscript{+}, NH\textsubscript{3}\textsuperscript{−}, NO, N\textsubscript{2}O, many organic N forms, threatening in terms of air quality, water quality, soil quality, the greenhouse gas balance, ecosystems and biodiversity—5 key threats identified by Sutton et al. (2011).

Quantitatively, NH\textsubscript{3} is currently believed to account for approximately half of all global biospheric, anthropogenic and natural atmospheric N\textsubscript{r} emissions, with N\textsubscript{r} defined and inventoried as the sum of NH\textsubscript{3}–N and oxidized nitrogen NO\textsubscript{x}–N. Global estimates of NH\textsubscript{3} and NO\textsubscript{x} emissions provided by the Emissions Database for Global Atmospheric Research (EDGAR 2011) were 40.6 and 37.2 Tg N year\textsuperscript{−1} for the year 2008, respectively. Agricultural NH\textsubscript{3} emissions dominate and are of the order of 27–38 Tg NH\textsubscript{3}–N year\textsuperscript{−1} (Beusen et al. 2008). Uncertainties in global NH\textsubscript{3} emissions are large, possibly up to 30–40 %, as shown by the variability in other published global figures (e.g. calculated estimates of 75 (50–128), by Schlesinger and Hartley 1992; 45 Tg NH\textsubscript{3}–N year\textsuperscript{−1} by Dentener and Crutzen 1994; 54 Tg NH\textsubscript{3}–N year\textsuperscript{−1} by Bouwman et al. 1997; 43 Tg NH\textsubscript{3}–N year\textsuperscript{−1} by van Aardenne et al. 2001). By comparison, the global biological and industrial N\textsubscript{2} fixation is of the order of 140 Tg N year\textsuperscript{−1} (Galloway et al. 2003), of which NH\textsubscript{3} emissions represent a loss of approximately one-third. The environmental impacts of NH\textsubscript{3} are expected to become more pronounced in many regions of the world where increases in NH\textsubscript{3} emissions are expected to occur during the 21st century, as a result of agricultural intensification and the manifold effects of climatic change on N cycling.

Within the European Union (EU-27), total NH\textsubscript{3} and NO\textsubscript{x} emission estimates are also of the same order, at 3.0 and 2.8 Tg N year\textsuperscript{−1}, respectively (European Environment Agency 2012; Sutton et al. 2011), contributing around 7.5 % of global emissions. Although EU-27 NH\textsubscript{3} emissions declined by 28 % from 1990 to 2010, the share of NH\textsubscript{3} in total European N\textsubscript{r} emissions increased from 44 % to reach the current level of 51 %, because NO\textsubscript{x} emissions almost halved (−47 %) over the same 20 year period (European Environment Agency 2012), due to very significant NO\textsubscript{x} emission abatements in the transport, industry and energy sectors. A range of NH\textsubscript{3} emission projections in Europe tend to indicate either a small increase, or possibly a slow linear decline of the order of ~25 % by the year 2100, while NO\textsubscript{x} emissions are projected to decline exponentially by ~75 % over the same time horizon (Winiwarter et al. 2011).

As oxidised N\textsubscript{r} eventually takes a backseat to reduced N\textsubscript{r} emissions in Europe and N. America, the degree to which NH\textsubscript{3} will control atmospheric chemistry and N deposition to sensitive ecosystems is set to increase over the next few decades. In addition, because NH\textsubscript{3} emissions largely originate in agriculture and
are predominantly the result of biological processes (with the notable exception of biomass burning and forest fires—e.g. R’Honi et al. 2013), they are much more weather/climate sensitive than are NOx emissions, which are dominated by industrial, domestic and traffic combustion processes. With global temperatures expected to rise by a few K, and based on thermodynamic considerations (a volatilisation Q10 of 3–4), agricultural NH3 emissions could increase substantially over the 21st century, although water availability is also a critical factor, favouring mineralisation of organic N sources while suppressing NH3 emissions by allowing dilution and infiltration (Sutton et al. 2013). The net impact of rising temperatures and altered spatial/seasonal precipitation patterns on regional and global NH3 budgets is as yet uncertain, with the uncertainty being compounded by land-use and land-cover changes and evolving agricultural practices (e.g. fertilization rates, spreading techniques, grazing density). Such an assessment will require the development of fully mechanistic, climate-dependent models for the quantification of surface/atmosphere NH3 exchange under global environmental change (Sutton et al. 2013).

**Requirements for Different Ammonia Exchange Models**

Predicting global-change-induced alterations of NH3 emissions and dry deposition is just one out of a range of environmental issues and ecological applications requiring biosphere/atmosphere NH3 exchange modelling, along with e.g. local N deposition impacts assessments (Hertel et al. 2011; Theobald et al. 2004, 2009; Sutton et al. 1998b; Loubet et al. 2009a), air quality studies (Pinder et al. 2007; Wu et al. 2008), and transboundary air pollution flux estimation (Simpson et al. 2012; Berge et al. 1999). Models of surface/atmosphere NH3 exchange have been both developed and applied for a number of purposes and at a large range of spatial scales ranging from the leaf or plant (Massad et al. 2010a), the canopy or ecosystem (Sutton et al. 1998a; Nemitz et al. 2001a; Riedo et al. 2002; Personne et al. 2009), the landscape (Loubet et al. 2009a; Hertel et al. 2006), to the national/regional level (van Pul et al. 2009; Bash et al. 2013) and to the globe (Dentener and Crutzen 1994).

The objectives of the modelling depend on the spatial and temporal scales at which models are ultimately applied. At the field/ecosystem scale, surface exchange models often come as an aid to the interpretation of measured flux data and to further process understanding (e.g. Sutton et al. 1995b; Flechard et al. 1999; Nemitz et al. 2000b; Spindler et al. 2001; Neirynck and Ceulemans 2008; Burkhardt et al. 2009), as the unexplained variability (residuals) points to potential model weaknesses and areas for further improvements. Models may also be used to fill gaps in measured flux time series in order to provide seasonal or annual NH3 exchange budgets (Flechard et al. 2010). In the absence of measured fluxes, but based on local meteorology and measured ambient concentrations at given sites, inferential modelling provides NH3 flux estimates for individual
ecosystems (Smith et al. 2000; Zimmermann et al. 2006; Walker et al. 2008; Zhang et al. 2009; Flechard et al. 2011). At larger (landscape, regional, global) scales, surface/atmosphere schemes are parameterized for different land uses and embedded within modelling contexts that encompass the whole cycle (from an Earth-Atmosphere-Earth perspective) of emission, dispersion, transport, chemistry and deposition (van Pul et al. 2009; Asman et al. 1998).

The process understanding gained over the years from controlled environment studies and field-scale measurements is eventually formalized into soil-vegetation-atmosphere transfer (SVAT) models, which then feed—in simplified, generalized forms—into landscape-scale models (LSMs), regional or global chemistry and transport models (CTMs), and dynamic global vegetation models (DGVMs).

**Ammonia Measurement and Modelling Approaches**

The development, parameterization and validation of models over the years has been, to a large extent, underpinned by the ever-increasing availability of NH$_3$ concentration and/or flux datasets across all scales.

At sub-landscape scales (cuvette, chamber, plot, field), this has stemmed from technological advances in NH$_3$ flux measurement instrumentation, capable of adequate lower detection limits, continuous online analysis for extended periods of time, selective quantification of gaseous NH$_3$ from aerosol NH$_4^+$, together with tolerable troubleshooting and maintenance workloads. In particular, at the field scale, wet denuder systems with automated online detection (Wyers et al. 1993; Blatter et al. 1994; Erisman et al. 2001; Thomas et al. 2009) have helped produce many exchange flux datasets by aerodynamic gradient methods (AGM) or Bowen ratio techniques, both at remote background locations with low (sub-ppb) concentration levels (Flechard and Fowler 1998b; Milford et al. 2001a), and over polluted semi-natural ecosystems and fertilized agricultural systems (Wyers and Erisman 1998; Nemitz et al. 2000a, b; Neirynck and Ceulemans 2008; Sutton et al. 2009b; Flechard et al. 2010; Wolff et al. 2010a; Loubet et al. 2012; Walker et al. 2013). Relaxed eddy accumulation systems have allowed NH$_3$ flux measurements at one single height (Nemitz et al. 2001b; Meyers et al. 2006; Hensen et al. 2009a). In parallel, a range of new generation, fast-response optical and mass spectrometry instruments have emerged over the last 15 year (see von Bobrutzki et al. 2010, for a review and intercomparison), which have proved suitable for eddy covariance (EC) measurements of large (emission) fluxes such as those occurring after the land spreading of manures (Whitehead et al. 2008; Sintermann et al. 2011). However, many of these instruments have yet to realize their full potential for the smaller exchange fluxes typical of unfertilized background situations (Famulari et al. 2004), not least due to aerosol NH$_4^+$ interference and to high-frequency damping losses of NH$_3$ fluctuations from adsorption/desorption within the measurement system, especially air inlet lines and online filters (Ellis et al. 2010; Whitehead et al. 2008).
At landscape/regional/global scales, it is much harder to make flux measurements, and modelled surface/atmosphere exchange cannot easily be directly validated. At the landscape scale, limited use has been made of plume measurements and inverse modelling of strong sources (Hensen et al. 2009b; Flesch et al. 2007; Blackall et al. 2007; Loubet et al. 2009b; Carozzi et al. 2013). However, model evaluation, especially at the regional scale, typically relies on the indirect indicators provided by measured wet deposition of NH$_x$ ($\text{NH}_3 + \text{NH}_4^+$) and, wherever available, ambient NH$_3$. Ammonia concentration measurements as part of spatial networks of atmospheric pollution monitoring using low-cost, long-term sampling, are available in few places worldwide (Tang et al. 2009; Adon et al. 2010). Encouragingly, recent developments in satellite-based infrared spectroscopy to map NH$_3$ concentrations (Clarisse et al. 2009; Shephard et al. 2011; R’Honi et al. 2013) suggest that the monitoring of NH$_3$ from space will help validate large-scale atmospheric models and refine current modelled estimates of regional and global NH$_3$ emissions.

Advances in instrumentation, flux measurements and process understanding since the early 1980s have allowed the atmospheric pollution modelling community to move from a unidirectional paradigm for NH$_3$ (fixed discrete point sources versus diffuse dry deposition everywhere else), to a dynamic bi-directional view, in which sources and sinks alternate in space and time depending on weather, pollution climate and agricultural management (Sutton et al. 2013). The major mechanisms and controls of NH$_3$ exchange have been identified at the substrate, plant, and ecosystem scales, even if there remain substantial gaps in knowledge, but the methodologies and models currently used to estimate emissions and deposition at landscape and regional scales have not all reached comparable levels of complexity. This is only partly due to computational limits (CPU time), as the very detailed processes operating at very short timescales might become prohibitive when run over regional and multi-annual scales. More likely, however, it is often a consequence of the lack of fine resolution, detailed input data required to run the schemes, compounded by the difficulty of turning largely heterogeneous measurement (flux) datasets into a generalised, unified and self-consistent modelling theory.

Scope of the Review

The state of the art of NH$_3$ surface/atmosphere exchange (measurement and modelling) has been examined in a number of reviews, e.g. Sutton et al. (1993c, 1995b, 2007, 2013), Asman et al. (1998), Nemitz et al. (2001a), Hertel et al. (2006, 2012), Loubet et al. (2009a), van Pul et al. (2009), Fowler et al. (2009), Wu et al. (2009), Massad et al. (2010b), Zhang et al. (2010). The present contribution seeks to bring together the most recent advances in measurements, understanding and modelling of surface/atmosphere NH$_3$ exchange over the vegetated land area, including the application of fertilizers, manures and slurry to farmland. Note that although NH$_3$
emissions from farmstead livestock housing and manure storage facilities represent around 20% (and biomass burning an additional 15%) of total emissions globally (EDGAR 2011), these will not be considered specifically. Similarly, sea/air exchange is not treated here, even though marine NH₃ emissions can be substantial, e.g. 30 Gg NH₃-N year⁻¹ over the EMEP grid area (Barrett 1998).

The present paper focuses on bi-directional NH₃ exchange over vegetation and soils in both (semi)-natural vegetation and agricultural systems, as well as uni-directional exchange (emission) fluxes from land-applied mineral N fertilizers and manures. A brief overview is first given of the meteorological, thermodynamic, chemical and biological processes controlling NH₃ emission and uptake at the substrate, plant and ecosystem levels. Existing models of surface exchange are examined at the different scales from the leaf to the globe, with an emphasis on the development of canopy-scale models and their implementation at larger scales (landscape, regional). Although the conceptualization of a model and its parameterization (the calibration of its parameters based on observations) are quite different things, in the surface exchange literature the two terms have sometimes been used interchangeably. The ultimate objective of this work is to integrate current knowledge into a common modelling framework adapted for local, regional and global scale models, and to examine the degree to which measurement and input data are available, or missing, in order to parameterize, and ultimately run, surface/atmosphere exchange models at the different scales.

Processes Controlling NH₃ Emission and Uptake in the Soil/Plant/Atmosphere Continuum

Thermodynamic and Chemical Controls

At the level of each potential NH₃ source or sink in the soil/vegetation system (apoplastic, leaf cuticle, surface water films, leaf litter, soil solution, fertilizer pellets, applied manure), the gaseous NH₃ concentration (NH₃,g) in equilibrium with dissolved [NH₃,aq] and [NH₄⁺] is governed by Henry’s law (Kₗ) and the NH₃ protonation constant (Kₐ) (Seinfeld and Pandis, 2006; see Montes et al. 2009, for a review of Kₐ and Kₗ parameterizations, and Fig. 1a, b). In the context of the atmospheric exchange through stomata with the leaf apoplast, this equilibrium concentration has been called the compensation point, here denoted χₙp; the net gaseous NH₃ flux to or from the air surrounding the substrate then depends on the concentration difference χₙp-χₐ, where χₐ is the ambient NH₃ concentration (Farquhar et al. 1980). This differential between surface and air concentrations can be applied for many substrates: if the concentration gradient is zero then there is no net exchange flux; if χₙp > χₐ then NH₃ emission from the substrate occurs, while with χₙp < χₐ there is a net uptake by the substrate. By convention, a positive flux denotes NH₃ emission, negative indicates deposition or uptake.
Advances in Understanding, Models and Parameterizations …

Temperature Effects and the R Ratio ([NH₄⁺]/[H⁺])

Thermodynamics dictate that any warming of the substrate, at constant substrate pH, theoretically results in a displacement of dissolved NH₃ to the gas phase, promoting NH₃ emission or at least opposing uptake by the substrate from the air. The relationship of $\chi_{cp}$ to temperature is exponential (Seinfeld and Pandis 2006), with a warming of 45 K roughly translating into a doubling of the compensation point (Fig. 1c) for a given $[\text{NH}_4^+] / [\text{H}^+]$ ratio in the liquid phase (Flechard and Fowler 2008). The $[\text{NH}_4^+] / [\text{H}^+]$ ratio is henceforth termed $\Gamma$ and characterises the NH₃ emission potential, normalised for temperature. Measured values of $\Gamma$ have been shown to be vastly variable (up to 5 orders of magnitude difference) between various parts of plant canopies, e.g. leaf surface water, soil, litter, bulk leaf tissue and the apoplast, e.g. in grassland (Sutton et al. 2009b; Burkhardt et al. 2009) and in maize (Walker et al. 2013), but the different $\chi_{cp}$ values all respond in the same way to temporal temperature changes as long as $\Gamma$ is constant.

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**Fig. 1** Thermodynamic controls of the air/solution NH₃/NH₄⁺ partitioning. a and b compilation by Montes et al. (2009) of published values, parameterizations and temperature dependencies of Henry’s law coefficients ($K_h$) and dissociation constants ($K_a$); the curve numbers refer to specific experiments cited in Montes et al. (2009), for solutions ranging from pure water to slurries and high activity solutions; c theoretical equilibrium air NH₃ concentration of a 100 μM NH₄⁺ solution as a function of temperature and showing the effect of pH in the range 4–7.5, calculated according to Sutton et al. (1994); d fitting of a theoretical thermodynamic curve to micrometeorologically measured surface NH₃ ($Z_0'$) concentrations over Scottish peatland, resulting in a best fit for the $[\text{NH}_4^+] / [\text{H}^+]$ ratio ($\Gamma$) of 132 for the moorland ecosystem (Flechard and Fowler 1998b)
In practice, it is clear that ecosystem N and NH$_4^+$ pools are ever changing and that $\Gamma$ values may undergo diurnal, seasonal and annual cycles in response to weather, phenology, senescence, etc., such that the theoretical temperature response with respect to NH$_3$ fluxes is not necessarily verified in the long term. Modelling approaches based on the temperature response of a $\Gamma$ emission potential should therefore theoretically also deal with temporal $\Gamma$ dynamics in the various parts of an ecosystem.

In the atmosphere, the reversible equilibrium of the gas/aerosol NH$_3$/HNO$_3$/NH$_4$NO$_3$ triad is also temperature (and relative humidity, RH) sensitive (Mozurkewich 1993), with likewise a displacement of aerosol-phase NH$_4^+$ and NO$^-$ towards gaseous NH$_3$ and HNO$_3$ in warmer (and drier) conditions. Depending on the relative mixing ratios of NH$_3$, HNO$_3$ and NH$_4$NO$_3$, and on temperature and RH in the air column within and just above vegetation, gas/particle inter-conversion may alter the net NH$_3$ flux, as exchange velocities for gas-phase NH$_3$ and aerosol-phase NH$_4^+$ are different (Brost et al. 1988; Nemitz et al. 2004; see Section “Vertical Distribution of Sources and Sinks Within and Above Ecosystems”).

Surface/Substrate pH and Acid/Base Ratio

Substrate pH is also a major chemical control of NH$_3$ fluxes; for a constant [NH$_4^+$] in solution the compensation point increases by a factor of 3.2 for every additional 0.5 pH unit, and by 10 for every pH unit (Fig. 1). Thus the wide range of pH values, and their temporal variations, typically encountered in plants and on other environmental surfaces, clearly show the importance of using accurate values in models of both emission from fertilizers and background bi-directional exchange. Apoplastic pH typically varies in the range 5–7 (Farquhar et al. 1980; Schjoerring et al. 1998; Hill et al. 2002; Massad et al. 2008), and a range of stress factors can induce temporal variations (Felle and Hanstein 2002). The pH of the apoplast can increase by a few tenths of a unit in drought-stressed plants (Sharp and Davies 2009), while both NH$_3$ and CO$_2$ can also alkalinize the apoplast (Hanstein and Felle 1999; Felle and Hanstein 2002). In grassland, Loubet et al. (2002) reported a sharp rise in apoplastic pH (from ~6.5 to ~7.5) as grazing animals were introduced to the pasture. Leaf age can be a factor; in perennial Luzula sylvatica, young leaves were found to have much higher apoplastic pH than old leaves, leading to 4 to 10-fold higher NH$_3$ compensation points (Hill et al. 2002).

On external leaf surfaces, the pH of rain and dew is typically acidic, in the range 3.5–6 (Burkhardt et al. 2009; Flechard et al. 1999), but alkaline conditions may also occur in plant surface wetness, resulting from the presence of soil particles (Sutton et al. 1993a; Walker et al. 2013). Also, instantaneous or chronic exposure to elevated NH$_3$ levels is likely to raise surface pH and affect the magnitude of the surface exchange flux (Wu et al. 2009).

Jones et al. (2007) showed that the non-stomatal resistance ($R_{ns}$) of moorland plants to the uptake of atmospheric NH$_3$ increased linearly with ambient NH$_3$ concentration in the range 0–100 $\mu$g m$^{-3}$. This indicates that at high ambient NH$_3$
levels, the non-stomatal dry deposition process is self-limiting as the cuticle and other canopy surfaces may become NH$_3$-saturated and a high pH strongly suppresses the effective NH$_3$ solubility. Such situations occur typically in the vicinity of point sources such as animal production facilities (Loubet et al. 2009a), where ambient concentrations decrease exponentially with distance, from typically >100 µg m$^{-3}$ within the nearest 50 m of animal buildings and manure storage areas down to less than 10 µg m$^{-3}$ within a kilometer (Walker et al. 2008).

The concurrent dry and wet deposition of acidic atmospheric gases and aerosols contributes to the regulation of plant surface pH, and much depends on the prevailing pollution climate, the occurrence and duration of surface wetness, and the relative abundances of NH$_3$ (the major atmospheric base) and of atmospheric acids (Erisman and Wyers 1993; Flechard et al. 1999). Thus plant surface (cuticle, wetness) pH is the main (if implicit) underlying mechanism that accounts for some parameterizations for non-stomatal resistance to NH$_3$ deposition, developed in a range of publications (Erisman et al. 1994; Nemitz et al. 2001a; Massad et al. 2010b; Wichink-Kruit et al. 2010), and based on the atmospheric molar ratios of NH$_3$ to SO$_2$ or NH$_3$ to total acids (SO$_2$, HNO$_3$, HCl), as proxies of surface alkalinity/acidity.

For field applied manures, the pH of cattle and pig slurries is typically in the range 7.5–8, but values down to 6.3 and up to 9.0 have been reported (Sintermann et al. 2012). This, combined with the natural variability of soil pH across agricultural landscapes in which manures are applied to land, contributes to the large variability in fluxes and NH$_3$ emission factors (EF) (Genermont and Cellier 1997; Segaard et al. 2002; Sommer et al. 2003; Sintermann et al. 2012). It should be noted that farmers typically monitor and manage soil pH to insure it is in an optimal range for the crop being produced and models should take this into account when estimating NH$_3$ fluxes for agricultural crops.

*Meteorological Controls*

Weather affects ecosystem/atmosphere NH$_3$ exchange directly through the physical effects of wind speed, turbulence, global radiation, atmospheric stability and water (rainfall, dewfall, snowfall, evapotranspiration). The enhancement by wind speed and surface friction of NH$_3$ volatilisation rates after slurry spreading or inorganic fertilizer application is well documented, with the effect being quantified by the aerodynamic resistance ($R_a$) to heat and trace gas transfer (Genermont and Cellier 1997; Segaard et al. 2002; Sommer et al. 2003). After slurry spreading, the radiative heating of the surface drives the evaporation of water from deposited manure and possibly the formation of a crust, which adds an additional surface resistance ($R_c$) to the aerodynamic ($R_a$) and the laminar boundary layer ($R_b$) resistances to emission (Sommer et al. 2003).

Unstable atmospheric conditions favour convective mixing and NH$_3$ volatilisation, although they tend to co-occur with warm days with strong evaporation
and high vapour pressure deficit (VPD), during which a slurry crust may form. Rainfall at the time of spreading tends to suppress NH$_3$ emission by diluting thick slurry and facilitating infiltration into the soil, where NH$_4^+$ ions adsorb to sites of cation exchange; however, after a dry period rainfall may dissolve the dry slurry crust and solubilise NH$_4^+$, which then becomes available for volatilisation. Similarly, short-lived NH$_3$ emission pulses may be triggered by rainfall after dry weather spells, for example in agricultural soils amended with mineral fertilizer and up to several weeks following fertilization (Walker et al. 2013), or in natural alkaline soils in arid environments, such as, e.g. the Mojave Desert (McCalley and Sparks 2008).

The same meteorological drivers similarly impact patterns of background and bi-directional exchange. Large wind speeds and unstable conditions reduce $R_a$ and thus tend to increase emissions from the canopy (upward fluxes) as well as dry deposition (downward fluxes). However, large wind speeds also increase NH$_3$ dispersion (Loubet et al. 2009a) and thus tend to reduce ambient NH$_3$ concentration levels close to point sources (Flechard and Fowler 1998a), such that, although the exchange velocity is higher (higher turbulence, lower $R_a$), the dry deposition flux may not be greater (Flechard and Fowler 1998b).

**Leaf Surface Wetness**

The control by rainfall and dewfall is more straightforward, with leaf-surface water generally acting as a more efficient sink for highly water-soluble NH$_3$ than does a dry cuticle, and water droplets also physically blocking stomatal apertures (Zhang et al. 2003), all favouring dry deposition and limiting emission by the ecosystem. Water droplets, and also thin water films formed by deliquescent particles on leaf surfaces (Burkhardt and Eiden 1994), are often acidic and increase the affinity and sink potential of the canopy for atmospheric NH$_3$ (Flechard and Fowler 1998b), as well as for NH$_3$ emitted by the underlying soil and leaf litter (Nemitz et al. 2000a). Burkhardt and Eiden (1994) also describe a \textit{“wick”} effect of microscopic water films, by which the migration of NH$_4^+$ ions between the apoplast and the external cuticle, along stomatal guard cell walls, is controlled by pH and NH$_4^+$ concentration gradients. Similarly, Sutton et al. (1995a) describe transcuticular fluxes of NH$_4^+$ between apoplast and leaf surface. Contrary to direct gaseous NH$_3$ transfer through stomates, such liquid-phase mediated transfers are controlled by the presence of free water and are controlled by relative humidity and/or the hygroscopicity of particles at the surface, but they do contribute to the net canopy-scale NH$_3$ flux.

The succession of wet and dry meteorological phases, such as nocturnal/diurnal cycles of dew formation and evaporation, and brief showers followed by sunny spells, may lead to alternating patterns of NH$_3$ uptake and re-emission from plant leaf surfaces. Cases of NH$_3$ desorption from cuticles following leaf surface water evaporation have been reported (Sutton et al. 1995c, 1998a; Flechard et al. 1999; Neirynck and Ceulemans 2008), demonstrating the reversibility of the
non-stomatal uptake process. Further, recent NH\textsubscript{3} flux measurements over maize, coupled with surface water pH observations and controlled experiments, suggest that wet leaf surfaces may actually occasionally provide a less efficient sink for NH\textsubscript{3} than dry cuticles, as a result of trans-cuticular base cation leaching and the presence of alkaline soil particles, both raising the pH of surface wetness (Walker et al. 2013).

All the processes described above are dependent on prevailing meteorological conditions, with surface wetness being controlled by the ratio of rainfall to evapotranspiration (driven by atmospheric VPD, wind speed and net radiation), while soil particle emission (erosion) is governed by wind speed, soil dryness, as well as agricultural activities, e.g. tillage. Air, vegetation and soil temperatures control a host of plant physiological (Section “Plant Physiological Controls”), soil and microbiological processes (Section “Soil and Microbial Processes”). Plant growth and root NH\textsubscript{4}\textsuperscript{+} intake, microbial activity, ammonification (microbiological NH\textsubscript{4}\textsuperscript{+} fixation from N\textsubscript{2}), nitrification (microbiological oxidation of NH\textsubscript{4}\textsuperscript{+} into NO\textsubscript{−}), soil respiration (mineralisation of soil organic matter) and leaf litter decay, all generally increase with temperature (given adequate water and nutrient supply) and regulate the dynamics of ecosystem NH\textsubscript{4}\textsuperscript{+} pools and NH\textsubscript{3} exchange fluxes.

**Plant Physiological Controls**

Vegetation may be a net source or a net sink for NH\textsubscript{3}, depending on the nitrogen status of plants and thus (indirectly) on the influx of nitrogen into the ecosystem, whether by fertilization of through atmospheric deposition (Massad et al. 2010b), providing a negative feedback where long-term NH\textsubscript{3} deposition tends toward ecosystem saturation (Sutton et al. 1993c). The present section focuses on the physiological parameters controlling the NH\textsubscript{x} status of the apoplast of green leaves (defined as the intercellular space where water and solutes can move freely), stems and inflorescences, and to some extent of senescing attached leaves.

**The Stomatal Compensation Point**

Meyer (1973) was the first to recognize that NH\textsubscript{3} is present (as NH\textsubscript{3,aq} and NH\textsubscript{4}\textsuperscript{+}) in intercellular fluids on the cell walls of the mesophyll cells of leaves (the apoplast), so that a compensation point air concentration of NH\textsubscript{3} exists. This was later shown in measurements by Lemon and Van Houtte (1980) and most famously by Farquhar et al. (1980). Prior studies using dynamic chamber measurements had typically shown consistent uptake by plant leaves and a direct control by stomatal conductance (e.g. Hutchinson et al. 1972), but the NH\textsubscript{3} concentrations applied to the chamber inlet were often much greater than typical ambient levels encountered in the field (0.1–10 μg m\textsuperscript{−3}), and above the stomatal compensation point (χ\textsubscript{s}), precluding emissions from the apoplast. Since then, many controlled environment
studies have shown linear relationships between ambient NH$_3$ ($\chi_a$) concentration and the NH$_3$ flux, with a bi-directional exchange switching from an emission at low $\chi_a$ levels to an uptake at higher $\chi_a$ levels, the switch occurring at $\chi_s$ (Sutton et al. 1995b; Husted et al. 1996; Schjoerring et al. 1998; Hill et al. 2001).

The stomatal compensation point is the equilibrium NH$_3$ concentration associated with the [NH$_4$] concentration in the apoplast, which results from the balance in healthy leaves of several production and consumption processes. These include: NH$_4^+$ import via the xylem; active (unidirectional) NH$_4^+$ transport into leaf cell cytoplasm and vacuole; passive (bi-directional) NH$_3$ transport between apoplast and cells; NH$_4^+$ assimilation within the cytoplasm into amino acids via the glutamine synthetase/glutamate synthetase (GS/GOGAT) cycle; and NH$_4^+$ generation by, e.g. photorespiration, nitrate reduction, protein turnover and lignin biosynthesis (Joy 1988; Schjoerring et al. 1998, 2002; Massad et al. 2008, 2010a). The experimental inhibition of GS by methionine sulfoximine in barley in the laboratory (Schjoerring et al. 1998), or the use of barley mutants with a reduced GS activity (Mattsson and Schjoerring 1996), both lead to NH$_4^+$ accumulation in the apoplast and dramatic increases in stomatal NH$_3$ emissions, demonstrating the critical role of GS (and GOGAT) in avoiding NH$_4^+$ accumulation in leaf tissues and regulating NH$_3$ emission.

**Apoplastic PH**

It is worth noting that, as the stomatal compensation point is not simply a function of [NH$_4^+$] in the apoplast, but rather a direct function of the [NH$_4^+$]/[H$^+$] ratio (or $\Gamma$) in the apoplast ($\Gamma_a$) (Section “Thermodynamic and Chemical Controls”), $\chi_s$ increases exponentially with pH. Any internal physiological regulation of apoplastic [H$^+$] that does not have a commensurate effect on [NH$_4^+$] therefore systematically affects $\chi_s$ and the stomatal NH$_3$ flux. Unlike intracellular pH, which must be maintained within a narrow range (7.2–7.5) to allow all plant metabolic functions to proceed, apoplastic pH is rather variable due to a fairly low passive buffer capacity (Felle and Hanstein 2002). The necessary regulation of intracellular pH is responsible for proton transfers across the cytoplasmic membrane, leading to apoplastic pH changes (Massad et al. 2008). In addition, plant responses to environmental stress factors such as drought have also been shown to affect apoplastic pH (Felle and Hanstein 2002; Sharp and Davies 2009), as do variations in ambient soluble trace gas (NH$_3$, CO$_2$) concentrations (Hanstein and Felle 1999). Thus small fluctuations in membrane transport, gas exchange (stomatal conductance) and intercellular mass exchange impact apoplastic pH (Felle and Hanstein 2002). Apoplastic pH is also believed to be influenced by N nutrition (Raven 1988), even if the effect is unclear (Massad et al. 2008). Plant species relying on NO$^-$ nutrition and assumed to assimilate NO$^-$ in the shoots tend to have higher apoplastic pH, while vegetation relying on mixed N sources (NH$_4^+$, NO$^-$, organic N) and more likely to favour root assimilation tend to exhibit lower apoplastic pH values (Hoffmann et al. 1992).
Plant Nitrogen Nutrition

Plant nitrogen uptake and status, development stage and species all affect $\Gamma_s$, resulting in diurnal and seasonal fluctuations at the ecosystem scale (Schjoerring et al. 1998; Massad et al. 2008, 2010b). The form of inorganic nitrogen (either $\text{NH}_4^+$ or $\text{NO}^-\text{O}$) being taken up by roots has been shown to impact stomatal $\text{NH}_3$ emission significantly, with emissions from $\text{NH}_4^+$-fed barley being a factor 10 higher than those from $\text{NO}^-\text{O}$-fed plants, consistent with higher leaf tissue $[\text{NH}_4^+]$ and higher xylem $\text{NH}_4^+$ concentration, given equivalent N contents of the nutrient solution (Mattsson and Schjoerring 1996).

Such effects of N form may have consequences for spatial distributions of $\Gamma_s$ values across landscapes, since well-aerated agricultural soils are generally $\text{NH}_3$-rich and $\text{NH}_4^+$-poor, while in permanent grasslands, heathlands and mature forests the opposite situation prevails (Schjoerring et al. 1998). Even though it is often assumed that all $\text{NH}_4^+$ is assimilated in the roots prior to transport to the shoots as amino acids, some studies have shown that at least a fraction of $\text{NH}_4^+$ might be transported prior to assimilation (Massad et al. 2008). By contrast, upon absorption by roots, $\text{NH}_3$ can either be reduced to $\text{NH}_4^+$ in root cells, stored in root cell vacuoles, exported via the xylem to the leaves or expelled to the outside of the root. Thus the $\text{NH}_4^+$ abundance in xylem and in the apoplast of leaves depends both on the soil $[\text{NH}_4^+]/[\text{NH}_3]$ ratio and on the balance of root assimilation, transport and storage in roots. Further, although $\Gamma_s$ generally increases with increasing N supply (Mattsson and Schjoerring 1996), and preferentially with $\text{NH}_4^+$ supply to the roots for several plant species, the relationship between the amount of N absorbed by the roots and the compensation point is not straightforward because of a possible masking effect due to apoplastic pH change (Mattsson and Schjoerring 2002; Massad et al. 2008).

High concentrations of N and $\text{NH}_4^+$ in bulk leaf tissues are expected to result in high $\Gamma_s$ values (Schjoerring et al. 1998). Mattson et al. (2009a) measured apoplastic pH and $\text{NH}_4^+$ concentrations of the eight most abundant graminaceae species of a fertilized grass sward in N. Germany, using the apoplastic extraction by vacuum infiltration technique (Husted and Schjoerring 1995). This direct method for the determination of $\Gamma_s$ is based on the measurement of the leaf apoplastic $\text{NH}_4^+$ concentration and pH by means of extraction with successive infiltration and centrifugation of leaf segments (Husted and Schjoerring 1995). The measured apoplastic $\text{NH}_4^+$ concentrations differed by almost one order of magnitude between species, while apoplastic pH values also varied from 6.0 to 6.9. The resulting $\Gamma_s$ values ranged from about 30 to over 700 and correlated very strongly (linearly) to bulk leaf $[\text{NH}_4^+]$, with the consequence that three out of eight grass species with the highest stomatal compensation points could behave as $\text{NH}_3$ sources, while the remaining five species were consistent sinks throughout the 3 week measurement campaign. Such variations in stomatal $\text{NH}_3$ emission potentials among species within the same habitat demonstrate the challenge of modelling the exchange at the ecosystem scale.
Massad et al. (2010b) compiled 60 published values of $\Gamma_s$ for non-managed (non-fertilized) ecosystem types including forests, heathlands and moorlands (average 502, range 3–5604), and 96 published $\Gamma_s$ values for managed systems including croplands, and fertilized and/or grazed grasslands (average 782, range 16–5233). In addition to data obtained using the vacuum infiltration technique, the data included estimates by cuvette-based controlled experiments and by field-scale micrometeorological flux measurements. Massad et al. (2010b) concluded that the key driver of $\Gamma_s$ appears to be the total N input to the ecosystem (whether by fertilization, atmospheric deposition, or both), and that $\Gamma_s$ values were positively and exponentially related to bulk tissue $[\text{NH}_4]^+$. Fertilized agricultural ecosystems generally show higher $\Gamma_s$ values than semi-natural vegetation, although very large $\Gamma_s$ values were also reported for example over polluted forest sites in the Netherlands and Belgium, which have been subjected to high nitrogen deposition loads for decades (Neirynck and Ceulemans 2008; Wyers and Erisman 1998).

**Temporal Variations**

The apoplastic $\Gamma_s$ ratio undergoes temporal variations on seasonal (Fig. 2) and diurnal timescales. Seasonal variations are expected to occur since the assimilation, transport and turnover of nitrogen change dramatically with plant developmental stage, and the seasonal NH$_3$ exchange pattern may vary for different types of vegetation depending on which processes dominate the actual N utilization (Schjoerring et al. 1998).

In two barley (Hordeum vulgare) cultivars grown in hydroponics, Husted et al. (1996) showed a marked decrease in the NH$_3$ stomatal compensation point in the period from tillering to anthesis, followed by an increase during senescence. In a fertilized ryegrass (Lolium perenne) pasture, van Hove et al. (2002) found that mean spring and summer apoplastic $[\text{NH}_4]^+$ were a factor 2–3 lower than in autumn and winter, but no distinct trend for apoplastic pH. Similarly, in a beech (Fagus sylvatica) forest, Wang et al. (2011) measured a gradual decrease of $\Gamma_s$ from leaf expansion (June) ($\Gamma_s > 150$) until the mid-season (August) ($\Gamma_s < 100$), followed by an increase during late season and approaching senescence ($\Gamma_s > 170$). Consequently, during the two (early season and late season) $\Gamma_s$ peaks, the leaves could act as an NH$_3$ source, while during the mid-season stomatal uptake prevailed. The authors concluded that a low glutamine synthetase activity in young, emerging beech leaves as well as in senescent leaves and hence, a low capacity for $\text{NH}_4^+$ assimilation, resulted in increased concentrations of tissue and apoplastic $\text{NH}_4^+$. Cellular breakdown during senescence and the associated catabolism of proteins, amino acids and chlorophyll liberates large amounts of $\text{NH}_4^+$, which is no longer assimilated and raises the NH$_3$ emission potential of plants, even before leaves drop to the litter on the ground surface (Mattsson and Schjoerring 2003). Age-related differences in the NH$_3$ compensation point of
25

Advances in Understanding, Models and Parameterizations …

Luzula sylvatica were also found to be considerable (Hill et al. 2002), with both apoplastic pH and NH$_4^+$ concentrations increasing during leaf expansion and declining prior to senescence.

Diurnal patterns of $\Gamma_s$ are generally less systematic than seasonal ones, even if there can be a large degree of hour-to-hour variability (Sutton et al. 2000; Herrmann et al. 2009; Flechard et al. 2010). Although diurnal cycles of NH$_3$ exchange fluxes have been observed in e.g. Brassica napus (Husted et al. 2000), Hordeum vulgare (Schjoerring et al. 1993) and tropical grassland (Trebs et al. 2006), with highest NH$_3$ emission rates typically occurring during the daytime and low rates at night, much of the observed diurnal variability in fluxes may be attributed to the temperature effect rather than to $\Gamma_s$ (Sutton et al. 2000; Personne et al. 2009). Reported diurnal variations in apoplastic NH$_4^+$ and H$^+$ concentrations often do not follow any particular trend (Husted et al. 2000; vanHove et al. 2002), even if some observations in a mixed graminae sward did tend to indicate higher $\Gamma_s$ values during the day than at night (Herrmann et al. 2009), especially after the grass was cut and fertilized.

Fig. 2 Seasonal variations of: a stomatal compensation point in Hordeum vulgare (Husted et al. 1996); b apoplastic [NH$_4^+$] in fertilized and grazed Lolium perenne grassland (van Hove et al. 2002); c apoplastic $\Gamma_s$ in fertilized and grazed Lolium perenne grassland in two adjacent fields (North and South) (Loubet et al. 2002); d apoplastic $\Gamma_s$ in Lolium perenne/Poa trivialis grassland (Wichink-Kruit et al. 2010); and e apoplastic [NH$_4^+$] in Fagus sylvatica (Wang et al. 2011). In b, F and S indicate application of artificial fertilizer (calcium nitrate) and slurry, respectively; M mowing and G grazing by cows. In c, vertical lines indicate management events: dotted lines cut; bold line fertilization; bold dashed lines grazing. The $\Gamma$ symbol represents the ratio [NH$_4^+$]/[H$^+$].
Fertilization Effects on the Apoplastic Emission Potential

Agricultural management (fertilization, animal grazing, grass cutting) is another source of temporal variability for $\Gamma_s$. A number of studies have shown that, in managed agricultural systems, field fertilizer application results in a $\Gamma_s$ peak during the days following the application and usually a return to the pre-fertilization value within one to two weeks. Mattsson and Schjoerring (2002) demonstrate that leaf apoplastic $\text{NH}_4^+$ is a highly dynamic pool, closely reflecting changes in the external (e.g. root) N supply. In fertilized *Lolium perenne* grassland, Loubet et al. (2002) measured an increase in both apoplastic [$\text{NH}_4^+$] and $\Gamma_s$ by up to two orders of magnitude immediately following the application of ammonium nitrate fertilizer, but the effect was short-lived, lasting only two weeks (Fig. 2). Mattsson et al. (2009b) also observed a sharp (factor 10) increase in the apoplastic $\text{NH}_4^+$ concentration of newly emerging leaves after cutting and fertilization of mixed grassland, whereby the NH$_3$ compensation point peaked the day after the fertilizer was applied and thereafter decreased over the following 10 days until reaching the same level as before fertilization. Smaller increases in $\Gamma_s$ associated with grass cuts and grazing have also been reported (Milford et al. 2001b; van Hove et al. 2002; Loubet et al. 2002; Wang and Schjoerring 2012).

Stomatal Conductance

Another major physiological control of NH$_3$ exchange fluxes at the leaf and plant level is the regulation of stomatal opening and conductance, through which the gaseous exchange between the sub-stomatal cavity and the atmosphere is mediated. Stomatal conductance ($G_s$) has long been known to be controlled by global radiation ($R_g$) or photosynthetically active radiation (PAR), air temperature ($T_a$), vapour pressure deficit (VPD), and soil water content (SWC) (Jarvis et al. 1976; Emberson et al. 2000a, b). Heat and drought stress cause stomata to close during the daytime, reducing $G_s$, evapotranspiration, CO$_2$ assimilation and the stomata/atmosphere transfer of trace gases including NH$_3$. For example, NH$_3$ flux measurements over soybean during dry summer conditions showed much suppressed stomatal exchange fluxes, and the bulk of the exchange dominated by non-stomatal fluxes, due to limited soil water availability and drought affecting stomatal opening during the afternoon (Walker et al. 2006). Those authors pondered whether their results were representative of soybean within their study area, but it should be stressed that such measurements are extremely valuable to characterize NH$_3$ exchange in dry conditions and regions of the world, since a large majority of existing field NH$_3$ flux datasets are representative of reasonably well-watered conditions in temperate climates.

Further, research over the past 20–30 year has shown the impact of rising CO$_2$ (Ainsworth and Rogers 2007) and O$_3$ (Wittig et al. 2007) concentrations on stomatal conductance, with expected reductions of $G_s$ of the order of $-20$ to $-30\%$ for elevated CO$_2$ and $-10$ to $-20\%$ for elevated O$_3$. Within the context of global
change, such impacts on $G_s$ should be accounted for when considering present and future scenarios of NH$_3$ exchange.

**Soil and Microbial Processes**

Many processes within the soil profile and on the soil surface lead to an NH$_4^+$ pool being present and available for exchange with the air column above the ground. Within the topsoil and particularly the root zone of any land ecosystem, the NH$_4^+$ pool is depleted by root absorption, by nitrification, by microbial immobilization, and by emission to the atmosphere; it is replenished by atmospheric deposition, by symbiotic N$_2$ fixation (BNF) and ammonification, by microbial turnover, by mineralization of soil organic matter (SOM) and of N-containing root exudates, and by the decay of leaf litter on the ground surface. Adsorption and binding to negatively charged clay mineral and organic colloids represent a transient pool, while dilution and infiltration through the deeper soil layers decrease the emission potential. In addition, in fertilized agricultural systems, the large and sporadic inputs of mineral and organic forms of N lead to sudden increases in available N and particularly NH$_4^+$, often well in excess of the instantaneous plant and microbial demand. In keeping with the $\Gamma_s$ terminology adopted for the apoplastic [NH$_4^+$/[H$^+$]] ratio, corresponding terms may be defined for the topsoil layer ($\Gamma_{soil}$), for the leaf litter ($\Gamma_{litter}$), or collectively a ground layer term ($\Gamma_g$). Figure 3 illustrates how typical values measured for $\Gamma_{soil}$ and $\Gamma_{litter}$ far outweigh (by 2–3 orders of magnitude) $\Gamma_s$ values in fertilized cut grassland, especially during the days following the application of fertilizer.

![Graph showing time course of $\Gamma$ values in different compartments of a mixed grassland ecosystem](image)

**Fig. 3** Time course of estimated $\Gamma$ values (ratio of [NH$_4^+$/[H$^+$]]) in different compartments of a mixed grassland ecosystem (from Sutton et al. 2009b). The grass was cut on 29 May and lifted for silage on 31 May. Fertilizer (100 kg N ha$^{-1}$ as calcium ammonium nitrate) was applied on 5 June.
Soil Background Emission Potential

Ammonium and ammonia are naturally present in soils as a product of microbial turnover and soil organic matter mineralisation, while fertilization (mineral and organic) as well as grazing in grasslands both supply large quantities of reduced N to agricultural soils. However, soil NH$_4^+$ is depleted by root uptake during the growing season, and by nitrification in well-aerated soils, while the soil NH$_3$ emission potential ($\Gamma_{\text{soil}}$) also largely depends on soil pH. One of the earliest studies on this effect made regional scale estimates of ammonia emission from soils based on mineralization rates, although at that time field verification of the modelled fluxes were missing (Dawson 1977).

In a more recent study over grassland, David et al. (2009) also identified the underlying soil as a strong potential source, but only after the grass was cut and for a short period of time (~1 day), and even then the soil potential emission was a factor of 3 lower than that of the leaf litter. However, few publications have ever shown soil below vegetation to be an ammonia source, be it below a grassland canopy in summer (Sutton et al. 1993b), under barley (Schjoerring et al. 1993), or oilseed rape (Nemitz et al. 2000a).

Neftel et al. (1998) actually suggested that soil must be a sink for NH$_3$ in a triticate field, since their semi-permeable membrane setup for direct measurements of NH$_3$ concentration in the soil showed consistently low (<0.1 µg m$^{-3}$) concentrations. This was despite large measured [NH$_4^+$] values in soil KCl extracts, which, accounting for the soil pH of 6.5, should have resulted in soil pore space NH$_3$ concentrations 2 orders of magnitude higher than those measured. They concluded from this discrepancy that the largest part of the estimated ammonium content in the soil was not in the liquid phase, but was instead adsorbed to solid soil particles, and thus not available for gas exchange with open porosity and the atmosphere. Similarly, Nemitz et al. (2000a) measured much lower NH$_3$ concentrations at a depth of −0.1 m within the soil than just above the leaf litter of oilseed rape, ruling out the possibility that the underlying soil may have been an NH$_3$ source in that study, and pointing to the importance of substantial NH$_3$ gradients at the air-soil-litter interface. There are altogether few reports of soil emission potentials for vegetated canopies in the literature that clearly distinguish the soil emission potential and flux from those associated with the leaf litter or the whole canopy (see Massad et al. 2010b for a review).

Soil Emissions After Fertilizer and Manure Application

Ammonia emission from the soil layer is most important after fertilization, especially if the fertilizer is urea-based or organic manure (Genermont et al. 1998; Segaard et al. 2002; Meyers et al. 2006; Sintermann et al. 2012). At the European scale, the land-spreading of organic manures is believed to contribute around 30–40 % of total NH$_3$ emissions (Sintermann et al. 2012, and references therein). Values of $\Gamma_{\text{soil}}$ typically increase by one or several orders of magnitude after slurry
spreading (Flechard et al. 2010). Although Fig. 3 indicates that $I_{\text{litter}}$ was a factor of 10 higher than $I_{\text{soil}}$ in the grassland system in Sutton et al. (2009b), even after fertilization, this study dealt with mineral fertilizer, and the situation is quite different for organic manures. A dominant mechanism of NH$_3$ loss to the atmosphere is the hydrolysis of urea and/or uric acid present in large quantities in animal wastes i.e. urine, slurries and farm yard manures, by the urease enzyme present in the excreted faeces and also in the soil. This leads to large concentrations of dissolved NH$_x$ and thus a high pH, promoting NH$_3$ volatilisation from the liquid phase. Urea hydrolysis also produces dissolved inorganic carbon, and the subsequent volatilisation of CO$_2$ increases pH, while NH$_3$ volatilisation decreases pH and is in principle self-limiting.

Apart from meteorological effects (Section “Meteorological Controls”), the most important processes controlling NH$_3$ volatilisation are the ion production and buffering processes controlling the pH of the slurry/soil liquid, the solid chemistry that determines precipitation of NH$_4^+$ to slurry dry matter, the physical processes controlling the movement of slurry liquid into and within the soil, the interaction of slurry liquid with soil cation exchange capacity (CEC) (Sommer et al. 2003; Genermont and Cellier 1997). Note that the method of field application (splash plate, trailing hose, trailing shoe, soil injection) is also expected to make a difference, with “low emission” application techniques being promoted in a number of countries to abate field losses (Sintermann et al. 2012; Carozzi et al. 2013).

Soil pH is expected to be a critical parameter controlling the magnitude of the percentage loss of volatilised NH$_3$ to the total NH$_x$ content of land-spread slurry, with emissions being effectively suppressed (<5 % loss) at soil pH 5 and potentially reaching over 50 % at pH 7 (Genermont and Cellier 1997; Loubet et al. 2009a). However, in practice there remain questions regarding the extent to which soil pH influences NH$_3$ losses from surface applied fertilizer and manures, as incomplete mixing may typically occur. Thus when and where soil pH affects the flux is a very complex question.

Soil microbial nitrification of the applied manure or fertilizer NH$_4^+$ depletes the NH$_x$ pool and thus may potentially limit the NH$_3$ emission potential in the days following field spreading. Whether nitrification significantly reduces NH$_3$ emission factors depends on nitrification rates, which have been shown to be extremely variable. For example Felber et al. (2012) measured very fast conversion of applied NH$_4^+$ from cattle slurry to NO$_-$ in top soil (0–10 cm) of grassland, with most of the NH$_4^+$ being nitrified within 2 days. By contrast, Laubach et al. (2012) found that nitrification proceeded more slowly in grassland fertilized with cattle urine, as soil [NH$_4^+$] only decreased by half over a week, although here soil [NH$_4^+$] was roughly a factor of 50 higher than in the study by Felber et al. (2012). Such variability highlights the need to give nitrification proper consideration in models of NH$_3$ volatilisation.

Emissions from synthetic fertilizers depend on the form of inorganic N applied but are typically smaller per unit added N than from manures. Emission from urea-based compounds are larger than from ammonium nitrate fertilizers, which do not raise soil solution pH. The use of urease inhibitors has been shown to reduce and
delay NH$_3$ volatilisation from urea in a number of field trials, including in a fertilized maize field, in which Walker et al. (2013) detected two distinct emission pulses spread over one month.

Despite extensive trials with a large literature over the last 25 year and good fundamental understanding of NH$_3$ losses from field-applied manures and fertilizers (e.g. Segaard et al. 2002, and the Ammonia Loss from Field-applied Animal Manure (ALFAM) database), there remain substantial uncertainties in field-scale NH$_3$ fluxes and the associated emission factors (EF). Sintermann et al. (2012) compiled and reviewed over 350 EF measurements published between 1991 and 2011 and raised questions about the representativeness, and possible overestimation, of NH$_3$ fluxes measured in medium-sized (20 m diameter) plots by mass balance methods such as the integrated horizontal flux (IHF) approach. The authors point out that emerging NH$_3$ volatilisation flux measurements at the field (>1 ha) scale over the last 5–10 year generally indicate much lower (~factor 2) NH$_3$ losses, typically below 30% of slurry NH$_4$ content, than did many medium-sized plot measurements carried out in the early 1990s (typically 50–80% losses), with serious implications for local and regional scale NH$_3$ budgets. A recent reassessment by Neftel et al. (2013) of EF measurements made in Switzerland in the early 1990s, using the $z_{\text{inst}}$ (simplified IHF) method (Wilson et al. 1982), hinted that these early EF values may have been significantly overestimated due to a combination of at least three factors, all leading to a systematic overestimation: over-speeding of the cup anemometers near the ground, cross-interference of plots located at distances of 70 m, and inadequate values of the $z_{\text{inst}}$ scaling factor. Such careful reanalyses of historical EF datasets from other countries might provide clues for the apparent discrepancies, or inconsistencies, reported in Sintermann et al. (2012).

Emission Potential of the Leaf Litter and Influence of Plant and Ecosystem N Cycling

Apart from fertilizer-induced NH$_3$ volatilisation, significant emissions may also occur from soil in barren land and in senescent plant canopies where leaf litter on the soil surface contributes to emissions (Sutton et al. 2009b; Massad et al. 2010b). Ammonia emissions from the leaf litter, even if understood in principle, remain very uncertain due to the limited number of studies (e.g. Denmead et al. 1976; Harper et al. 1987; Nemitz et al. 2000a; Mattsson and Schjoerring 2003; David et al. 2009; Wang and Schjoerring 2012). The literature generally indicates very large $I_{\text{litter}}$ values but their temporal dynamics are poorly understood. By contrast to mineralization rates of plant litter incorporated into soils (e.g. Nicolardot et al. 1995), little is known about processes within detached leaves lying on the ground surface. Schjoerring et al. (1998) argued that NH$_4^+$ production by mineralization and liberation in the leaf tissue are coupled to degradation
of chlorophyll and of soluble proteins in detached senescent leaves; this is by contrast to senescing leaves that are still attached to the plant, which still have a relatively efficient N remobilisation and are able to avoid accumulation of correspondingly high \( \text{NH}_4^+ \) levels by transfer to other parts of the plant.

For the ground leaf litter, it has been assumed that \([\text{NH}_4^+]\) is controlled by the litter water content, by mineralization and nitrification rates as well as the amount of \([\text{NH}_4^+]\) released to the atmosphere as \(\text{NH}_3\) (Nemitz et al. 2000a). The \(\text{NH}_3\) emission potential of the leaf litter \(I_{\text{litter}}\) is first and foremost dependent on the initial bulk N content of senescent leaves as they become detached from the plant; N-rich leaves are obviously more likely than N-poor leaves to liberate large amounts of \(\text{NH}_4^+\) via mineralisation on the ground. The nitrogen content of plant residues is controlled by contrasting processes in perennial woody species and in annual or biennial non-woody plants, as detailed hereafter.

**Role of Translocation on the Leaf Litter Nitrogen Content of Trees**

In trees, the litter N content is controlled by the ratio of ecosystem-internal N cycling (litter production, mineralisation, root uptake) to tree internal N cycling (assimilation, translocation, storage). These processes ensure that large amounts of N remain available to the plant and are moderately protected against immobilisation in stable soil organic compounds or losses via leaching and gaseous emission (Wang et al. 2013). The N status of attached senescing leaves is controlled by the degree to which N is retranslocated from such leaves into the rest of the tree before leaf fall. The re-translocation is directed either into woody roots and/or the trunks in deciduous species, or from previous years leaves into the youngest age class needles in conifers. The resulting reduction in foliar N content may be expressed as the fraction of N re-translocation relative to the initial N content in the green leaves.

Comparing three European forests subject to contrasting atmospheric N deposition loads, Wang et al. (2013) found that this N re-translocation efficiency was lowest in a Douglas fir stand (37 %) subject to very large (45 kg N ha\(^{-1}\) year\(^{-1}\)) N deposition, compared to a temperate beech forest (70 %) and a boreal pine stand (62 %) exposed to much lower N deposition (ca. 20 and 5 kg N ha\(^{-1}\) year\(^{-1}\), respectively). The boreal pine site thus returned the lowest amount of N via foliage litter to the soil, while the temperate Douglas fir stand returned the highest amount of litter N to the ground. The authors concluded that forests activate very different mechanisms to reduce N losses in foliage litter production: (i) increased N re-translocation efficiency, (ii) increased leaf longevity, (iii) decreased foliage N content and and (iv) decreased foliage mass. Despite the lowest leaf longevity and highest leaf N contents, the beech canopy reduced the N losses via leaf litter production by having very efficient N re-translocation prior to leaf fall.
Nitrogen Content in Leaf Litter and Other Residues in Crops and Grassland

Nitrogen re-allocation from ageing leaves to younger leaves, to growing seeds and to storage for the next growing season may also occur in annual and biennial non-woody plants, such as many agricultural crops, and in perennial grasslands (Wang and Schjoerring 2012). However, in many cases all the non-harvested above-ground biomass eventually returns to soil, either as litterfall during the growing season, or after harvest. Thus the soil layer is the ultimate resting place for the non-harvested stem and foliar N, both from bottom-canopy senescent leaves dropping to litter during the growing season, as well as litterfall following complete senescence or harvest. In a ryegrass (Lolium perenne) grassland, Wang and Schjoerring (2012) found that green photosynthesizing leaves generally had the largest total N concentration, followed by stems and inflorescences. By contrast, the lowest total N content occurred in senescent leaves, indicative of N re-allocation. The situation was reversed for the bulk $\Gamma$ ratio (total leaf tissue $[\text{NH}_3^+] / [\text{H}^+]$), with green leaves and stems generally showing substantially lower $\Gamma$ values than senescent leaves and litter. Thus, although remobilization had reduced total N concentrations in senescent leaves and litter, mineralization of organic N compounds still lead to much higher bulk $[\text{NH}_3^+]$ values than in green leaves.

Many studies have observed large NH$_3$ concentrations near the ground surface and litter in closed canopies (e.g. Denmead et al. 1976; Nemitz et al. 2000a), resulting from the production and accumulation of NH$_3^+$ by mineralisation of litter organic compounds. In mixed grassland, David et al. (2009) defined the litter as the sum of both senescing attached leaves and dead/decomposing detached leaves. By means of dynamic chamber measurements (cuvette), they found that emissions from the litter were the largest source in the canopy and that emissions were higher from wet than from dry litter. They also found that peak NH$_3$ emissions from litter leaves occurred both after a step decrease and a step increase of air relative humidity, due to a change in either increased evaporation or increased mineralization. This was consistent with the findings within an oilseed rape canopy by Nemitz et al. (2000a), who demonstrated with a simple dynamic litter model that shrinking liquid pools within the leaf litter lead to more concentrated NH$_3^+$ pools and increased emissions. Here, measurements of within-canopy vertical NH$_3$ concentration profiles, from a depth of $-0.1$ m in the soil up to the top of the oilseed rape canopy (1.4 m), showed mean in-soil and top-canopy concentrations of the same order (1–2 $\mu$g m$^{-3}$), but much higher concentrations ($\sim 9$ $\mu$g m$^{-3}$) just above the leaf litter. This information, coupled with the inverse Lagrangian modelling technique (ILT) (Raupach 1989) to determine the vertical distribution of NH$_3$ concentration, sources and sinks within the canopy, demonstrated the existence of a large emission potential within decomposing litter leaves on the soil surface, which was consistent with previous studies (e.g. Denmead et al. 1976). However, in order to simulate diurnal variations of the measured NH$_3$ concentration at the surface of the leaf litter ($\chi_{\text{litter}}$), Nemitz et al. (2000a) needed to adopt a dynamic approach for $\Gamma_{\text{litter}}$. By contrast, using a constant $\Gamma_{\text{litter}}$ resulted in an overestimation of $\chi_{\text{litter}}$ at
the start and an underestimation of $\chi_{\text{litter}}$ towards the end of the modelled period. This reflected the dynamics of the litter $\text{NH}_4^+$ pool, which could be shown in a simple dynamic model to be controlled by (a) mineralization and nitrification rates according to Dawson (1977) and (b) the response of the leaf water content to relative humidity (RH), as proposed by van Hove and Adema (1996).

**Vertical Distribution of Sources and Sinks Within and Above Ecosystems**

The magnitude and direction (or sign) of the net vegetation/atmosphere NH$_3$ flux are controlled by the difference between the ambient NH$_3$ concentration ($\chi_a$) and the canopy compensation point, denoted $\chi_c$ and introduced by Sutton et al. (1995b). The $\chi_c$ modelling concept (further developed in Section “Ammonia Exchange Models and Parameterizations from the Leaf to the Globe: State-of-the-Art”) reflects the fact that both NH$_3$ emission and deposition may co-occur at different levels within a canopy or plant-soil system, with for example emissions by a leaf litter on the soil surface and by sunlit stomates in the upper part of the canopy, concurrent with deposition to wet non-stomatal leaf surfaces and also possibly uptake by cooler, shaded stomates in the lower part of the canopy (Sutton et al. 1995a; Nemitz et al. 2000a, b, 2001a; Personne et al. 2009). Given this multi-layered vertical distribution of sources and sinks and internal canopy cycling of NH$_3$, $\chi_c$ defines the net bulk, canopy-scale potential for emission or deposition when set against the atmospheric NH$_3$ concentration $\chi_a$ occurring overhead.

Micrometeorological NH$_3$ flux measurements made above ecosystems provide estimates of the net exchange between the whole soil/litter/canopy system (including the within-canopy air space) and the free atmosphere. Such ecosystem-scale measurements by themselves do not provide the sink and source contributions of the different canopy components (soil, litter, stomates, green leaves, senescing leaves, stems, inflorescences, non-stomatal (cuticular) surfaces, etc.) to the net exchange. Measurements using dynamic chamber may be used to isolate certain terms, such as individual leaves, soil or litter, but other terms such as the partitioning between stomatal and non-stomatal fluxes (Sutton et al. 1995a), or the air column sink/source term from gas-particle interconversion (GPIC) (Brost et al. 1988; Nemitz et al. 1996), can only be apprehended by using models. The ability to model the different canopy component flux terms quantitatively is crucial to determine the net canopy-scale flux (for e.g. regional-scale modelling), but it also provides insights into the NH$_3$ canopy cycling and reveals potential feedbacks between total N inputs and the net NH$_3$ flux (Sutton et al. 1995a).

The NH$_3$ exchange literature shows many examples of vertical stratification of sources and sinks within soil-plants systems, and of widely varying NH$_3$ emission potentials for canopy components. This is exemplified by the different $\Gamma$ ratios (Fig. 3) in grassland, ranging over 4–5 orders of magnitude (Sutton et al. 2009b), and by a similar picture in maize (Walker et al. 2013), which also included a $\Gamma$ term for leaf surface wetness (dew, guttation).
Within-Canopy Vertical NH₃ Concentration Profiles

The vertical distribution of—and relationships between—the various NH₃ sources and sinks are influenced by canopy structure, leaf area index (LAI) and leaf area density profile, which control within-canopy turbulence as well as vertical profiles of wind speed, NH₃, temperature and RH. Ammonia profiles within cereal canopies have often shown the largest concentration at mid-canopy, at the height of the greatest leaf density (e.g. Meixner et al. 1996), which was consistent with the widely held assumption that, above cereal crops, NH₃ emissions mostly originate from stomata (e.g. Farquhar et al. 1980). By contrast, in canopies of grass-clover pasture as well as soybeans, oilseed rape and quackgrass, within-canopy profiles showed the highest concentrations at ground level (Denmead et al. 1976; Lemon and van Houtte 1980; Sutton et al. 1993b; Nemitz et al. 2000a, 2009a; Bash et al. 2010), which is generally attributed to leaf litter decomposition and NH₃ emission from the soil. In the light of the latter studies, and especially given the much larger emission potentials associated with the soil and leaf litter than with the apoplast (Fig. 3), the role of stomatal emissions as a major control of the net canopy-scale flux must be re-examined.

Although the apoplast may, under certain circumstances, act as an NH₃ source, this very much depends on the vertical position of leaves, which is correlated with their age, temperature, and their proximity to the free atmosphere or to the soil/litter layer.

Recapture of Soil/Litter-Emitted NH₃ by the Overlying Canopy

For agricultural crops during the growing season, soil emissions might be expected to be largely recaptured by the overlying canopy, either by stomatal absorption or by surface wetness uptake (Nemitz et al. 2000a; Meyers et al. 2006). In practice, the fraction of NH₃ estimated to be recaptured is very variable between studies.

The ability of plant canopies to recapture substantial amounts of NH₃ released from fertilizer or plant residues at the ground is an important issue in agricultural air quality that is still a matter of debate (Denmead et al. 2008). For example, management options to reduce NH₃ volatilization losses from urea include to delay its field application (Denmead et al. 2008), or to use urease inhibitors (Walker et al. 2013). In the second of these, it is envisaged that a developed canopy would attenuate canopy wind speeds, leading to lower transport rates in the canopy air space, increased NH₃ concentrations, and greater uptake by the canopy foliage (Denmead et al. 2008).

By combining vertical in-canopy NH₃ profile measurements with ILT modelling, Nemitz et al. (2000a) calculated that all NH₃ emitted from the ground level was recaptured within the lowest half of an oilseed rape canopy, except during windy nighttime conditions, and that the net ecosystem daytime emission (measured by the flux gradient technique above the canopy) originated from the top half of the canopy. The N loss from the plant’s top leaves and siliques (seed
cases) to the atmosphere as gaseous NH$_3$ was more than balanced by the lower leaves uptake from NH$_3$ emitted by decomposing leaf litter. Similarly, in a fully developed grassland canopy (before cutting), Nemitz et al. (2009a) measured in-canopy profiles of NH$_3$, which again were consistent with a large ground-level source, presumably from senescent plant parts, which was entirely recaptured by the overlying canopy. This ground-level source was believed to be responsible for the sustained NH$_3$ emissions observed after grass cutting, as indicated by independent bioassay and chamber measurements (David et al. 2009). The GRassland AMMonia Interactions Across Europe (GRAMINAE) grassland experiment, summarised by Sutton et al. (2009a, b), demonstrated that overall, net above-canopy fluxes were mostly determined by stomatal and cuticular uptake before the cut, by leaf litter emissions after the cut, and by fertilizer and litter emissions after fertilization.

A range of other experiments in crops have shown only partial canopy recapture of soil emissions. In maize, Bash et al. (2010) calculated, using an analytical first-order closure inverse source/sink model, that the fraction of soil-emitted NH$_3$ that was recaptured by the overlying canopy was 73 % for fertilizer applied to the soil surface (see also Walker et al. 2013). In another maize canopy, over which dairy waste effluent was spread, Harper et al. (2000) found that 17 % of the soil NH$_3$ emission was recaptured by the canopy during one ILT modelling run in mid-afternoon. However, overall only 21 % of the net emissions came from the soil, while 79 % came from the foliage. This occurred because the fertilizer was sprayed from above the canopy, so that much of the NH$_3$ was emitted from leaf surfaces even before the fertilizer hit the ground. This shows that the fertilizer application method alters the soil-canopy source and sink relationship and should be accounted for in CTMs as a way to more accurately simulate the impact of agricultural management practices on fertilizer NH$_3$ emissions.

In a sugarcane crop, Denmead et al. (2008) estimated that the percentage of canopy recapture of NH$_3$ volatilized from urea fertilizer applied to the ground was of the order of 20 % for a LAI of 2, but they indicated that this fraction would increase with LAI, and that the efficiency of NH$_3$ recapture would be different in denser canopies or crops with different canopy structure. By extension, in dry climates, and for young and/or sparse or recently cut vegetation (grassland), the soil source strength potential is likely to be more fully expressed (as net emission to the atmosphere), since the canopy recapture fraction is likely to be small. In such systems, if the soil layer $r$ ratio is large, then the net canopy-scale flux is likely to be largely independent of stomatal and leaf surface exchange if LAI is small (Nemitz et al. 2001a).

**Gas-Particle Interconversion (GPIC) Within and Above the Canopy**

Air column chemistry within and above the canopy, and particularly the reversible thermodynamic equilibria of the NH$_3$–HNO$_3$–NH$_4$NO$_3$ and NH$_3$–HCl–NH$_4$Cl gas-aerosol triads, is known to affect NH$_3$ surface-atmosphere exchange rates
(Brost et al. 1988). There are three ways in which gas-particle conversion and aerosol evaporation affect NH$_3$ fluxes and local N$_r$ budgets (Nemitz et al. 2009b):

1. Vertical flux divergence and error in flux measurement. The presence of additional sources or sinks in the air below the flux measurement height means that the measured flux differs from the true surface exchange. Thus, fluxes measured by micrometeorological techniques that operate at a single measurement height ($z_m$), such as EC and relaxed eddy accumulation (REA), may need to be corrected for this effect. While these single height approaches still derive the correct local flux at the measurement height, the situation is more complex for gradient flux measurements. In that case, the vertical NH$_3$ gradient is modified by the chemistry, so that the aerodynamic gradient technique (AGM) may need to be modified to derive the correct NH$_3$ flux, including the chemical production or depletion term within the canopy space in addition to foliar exchange (Nemitz et al. 2004; van Oss et al. 1988).

2. Error in inferential estimates and deposition modelling. Deposition and emission are often derived from the air concentration in an inferential approach, using resistance models of a range of complexity. This approach does not usually consider chemical conversion within the resistance analogue (Kramm and Dlugi 1994). In addition, changes in the gas/particle partitioning modify air concentrations compared with the simulation of an atmospheric transport model that ignores chemical reactions. For example, the NH$_3$ air concentration is lowered by transfer to the particle phase, further stimulating stomatal emission, which is governed by the difference between stomatal compensation point and atmospheric concentration. A multi-layer modelling framework that simulates the coupled exchange, transport and chemistry inside the canopy is needed to resolve this effect (Nemitz et al. 2012; Ellis et al. 2011).

3. Modification of the local N$_r$ budget. Gas-to-particle conversion usually occurs in situations of strong NH$_3$ emission. In this case a fraction of the emitted NH$_3$ is converted into slowly depositing NH$_4$NO$_3$ aerosol, “increasing” the potential for local N deposition and lowering the air concentration of NH$_3$ near the surface, thus stimulating further emissions from NH$_3$ compensation points. At the same time, fast depositing HNO$_3$ is converted into slowly depositing NH$_4$NO$_3$ aerosol, “decreasing” net N deposition. Similarly, NH$_4$NO$_3$ evaporation may occur near the surface, due to elevated canopy temperatures and reduced concentration of NH$_3$ and HNO$_3$ (driven by deposition), usually over semi-natural vegetation, which provides an efficient sink for NH$_3$. This process converts slowly depositing aerosol NH$_4$NO$_3$ into fast depositing HNO$_3$ and NH$_3$ gas, thus increasing total N deposition. The net effect of gas-to-particle conversion on the local N budget will depend on the relative magnitudes and exchange rates of the different compounds involved.

The potential degree of vertical flux divergence depends on the comparative chemical timescales for the evaporation or formation of NH$_4$NO$_3$ and NH$_4$Cl and the timescales for turbulent transport, which are different within and above the canopy; it also depends on the relative mixing ratios of NH$_3$ compared with the other
chemically interactive species (gaseous HNO$_3$ and HCl and aerosol-phase NH$_4^+$, NO$^-$ and Cl$^-$). Thus Nemitz et al. (2000c), for example, found ample evidence that there was the potential for NH$_4$Cl formation (i.e. an NH$_3$ sink) within an oilseed rape canopy in S. Scotland, where the in-canopy turbulence was low and residence times long. By contrast, above the canopy they predicted that there was potential for NH$_4$Cl evaporation (i.e. an NH$_3$ source). The small aerosol concentrations measured at their site resulted in chemical timescales for the evaporation or formation of NH$_4$NO$_3$ and NH$_4$Cl that were much longer than those for diffusive transport above the canopy. This meant that gas-particle interactions were unlikely to have affected above-canopy flux-gradient measurements of NH$_3$, and indicated that the aerodynamic gradient method is applicable to NH$_3$ flux measurements in environments with low particle concentrations (relative to NH$_3$) without the need to correct for the effects of GPIC. However, the relative effect of these interactions on the fluxes of HNO$_3$ and NH$_4$NO$_3$ may be considerable (cf. Nemitz et al. 2012). During the GRAMINAE Braunschweig experiment, gas-particle interactions were also believed to have had a minor effect on measured ammonia fluxes, though the relative effect on calculated aerosol deposition rates was significant (Sutton et al. 2009b; Nemitz et al. 2009b).

In more polluted environments, the impact of GPIC on NH$_3$ exchange can be significant. Over heathland in warm conditions in the Netherlands, Nemitz et al. (2004) established that there was near-surface evaporation of volatile NH$_4^+$ (i.e. an apparent NH$_3$ source) during the aerosol de-position process, which led to a substantial overestimation of the NH$_3$ flux (by the gradient method) of +20 ng m$^{-2}$ s$^{-1}$ during the day. They concluded that NH$_4^+$ evaporation may lead to a significant underestimation of NH$_3$ deposition to semi-natural vegetation during daytime by current measurements and models, in which such processes are not explicitly accounted for. This is particularly true if flux measurements are carried out in areas where large aerosol concentrations lead to short chemical timescales and where large concentration of volatile NH$_4$NO$_3$ or (less likely) NH$_4$Cl are present. These conditions are fulfilled above semi-natural vegetation in the vicinity of high NH$_3$ emission densities, common in the Netherlands and other areas with high livestock densities.

Model simulations by van Oss et al. (1988) successfully simulated observations of NH$_3$-aerosol deposition faster than permitted by turbulence above the Dutch forest Speulderbos. They showed that NH$_3$ emission fluxes obtained at Speulderbos may not originate from the foliage but could at least partly be explained by the evaporation of NH$_4$NO$_3$ close to or within the canopy. However, evaporation of NH$_4$NO$_3$ from leaf surfaces may have a similar effect. The complex topic of air column chemistry and gas-particle interconversion and its relevance to NH$_3$ exchange is addressed more fully by Nemitz et al. (2012).

The stratification and interactions of processes controlling surface/atmosphere NH$_3$ exchange reviewed in this section are illustrated in Fig. 4, which was originally drawn to summarise the scientific objectives and tasks within the GRAMINAE Braunschweig experiment (Sutton et al. 2009a). This project focused on processes in fertilized and cut grassland, but Fig. 4 can essentially serve as a
blueprint for any integrated project aiming at a full understanding of component-scale and canopy-scale NH$_3$ fluxes in other vegetation types (for semi-natural ecosystems, the management and fertilization issues can simply be ignored). The figure illustrates intuitively that NH$_3$ pools exist, expand or shrink over time, and interact at all levels of the ecosystem: soil (aggregates, cation exchange sites, water-filled porosity, open porosity); soil surface, fertilizer residues and litter; plant (xylem, phloem, apoplast, cytoplasm, vacuole, organelles); plant surfaces (water films, cuticle, deliquescent aerosols); and even in the air space within and above the canopy. Surface/exchange models should therefore, in theory, seek to simulate the temporal as well as the vertical variability in these pools, in order to simulate the dynamics of canopy-scale fluxes.

**Ammonia Exchange Models and Parameterizations from the Leaf to the Globe: State-of-the-Art**

A large number of models have been developed to simulate NH$_3$ exchange fluxes for the different ecosystem components or processes (soil, litter, leaf, plant, heterogeneous-phase chemistry), either separately or integrated into canopy-scale 1-dimensional (1-D) soil-vegetation-atmosphere (SVAT) frameworks.
Landscape-scale, regional-scale and global-scale models are 2-D or 3-D, and they typically include simplified versions of canopy-scale models to simulate the 1-D surface exchange as part of the wider modelling context of emission, dispersion, transport, chemistry and deposition. The level of complexity of 1-D NH₃ exchange models depends on the different purposes and temporal scales as well as spatial scales, at which they are put to use. Modelling approaches range from the fully empirical to the primarily mechanistic. This section provides an overview of existing models, and their current parameterizations, ranging from the component (or substrate) scale to the global scale. The review is by no means exhaustive, but instead focuses on state-of-the-art models, and those models which represent potential options for implementation into integrated canopy, or larger scale, models. At each level, the model’s scope, advances, challenges, and degree of validation are discussed. Model names are highlighted in bold characters on first mention, and a summary of models is provided in Table 1.

### Table 1

<table>
<thead>
<tr>
<th>Full model name</th>
<th>Acronym/short name</th>
<th>Reference</th>
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<td><strong>Process-based soil, manure, fertilizer, or agro/ecosystem emission</strong></td>
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<td>AGRIN</td>
<td>AGRIN</td>
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<td>Godwin et al. (1984)</td>
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<td>Crop environment REsource synthesis—EGC (INRA)</td>
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<td>DeNitrification DeComposition</td>
<td>DNDC</td>
<td>Li et al. (1992), Li (2000)</td>
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<td>Environmental Policy Integrated Climate</td>
<td>EPIC</td>
<td>Williams et al. (2008)</td>
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<td>Generation of emissions from Uric Acid Nitrogen Outputs</td>
<td>GUANO</td>
<td>Blackall et al. (2007), Riddick (2012)</td>
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<td>Volt’Air</td>
<td>Volt’Air</td>
<td>Genermont and Cellier (1997)</td>
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<td><strong>Leaf/plant-scale stomatal exchange</strong></td>
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<td>Pasture Simulation</td>
<td>PaSim</td>
<td>Riedo et al. (1998, 2002)</td>
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<td>STomatal AMmonia compensation Point</td>
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<td>Massad et al. (2010a)</td>
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<td>Erisman et al. (1994)</td>
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<td>Dynamic pollutant Exchange with Water films on vegetation Surfaces</td>
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<td>Multi-Layer Model</td>
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<td>SPRUCEDEP</td>
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<td>SURFace ATMosphere NH$_3$</td>
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<td>American Meteorological Society/Environmental Protection Agency Regulatory Model</td>
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<td>Perry et al. (2004)</td>
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<td>Atmospheric Dispersion Modelling System</td>
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<td>OPS-Pro 4.1</td>
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<td>Danish Ammonia MOdelling System</td>
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<td>UK Met. Office Global Three-Dimensional Lagrangian Model</td>
<td>STOCHEM</td>
<td>Collins et al. (1997), Bouwman et al. (2002)</td>
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**Process/Component Scale Models: Soil, Manure, Fertilizer, Leaf Litter, Leaf, Cuticle, Air Column Chemistry**

**Ammonia Emissions from Slurry and Fertilizer Applied to Soils ($\Gamma_{\text{soil Emission Potential}}$)**

Various modeling concepts have been developed to account for the physico-chemical processes controlling NH$_3$ emission from mineral or organic manures upon field application to bare soil, and to simulate the peak emissions and diurnal trends of NH$_3$ emissions following slurry application (e.g. Van der Molen 1990; Sommer et al. 2003; Montes et al. 2009). Genermont and Cellier (1997) developed a mechanistic model (Volt’Air) that simulates the controls by soil, meteorology and slurry characteristics on NH$_3$ volatilisation from field-applied slurry, accounting for the transfers and equilibria in the topsoil and between the soil and the atmosphere. The model included energy balance and advection submodels, which made it suitable for field scale applications using simple meteorological data. Sensitivity analysis showed that soil pH has a large influence on volatilization. The model is also sensitive to soil adsorption capacity and some hydraulic characteristics (saturation water conductivity, water content at field capacity) (Garcia et al. 2011). Volt’Air has also been extended to simulate emissions by mineral fertilizers (Laguell-Hamaoui 2012).

The process-based AGRIN model, developed by Beuning et al. (2008), combined model theory of soil biological processes such as SOM decomposition, nitrification and denitrification (DNDC, Li et al. 1992; Li 2000), with Volt’Air-type models of NH$_3$ volatilization (Genermont and Cellier 1997; Van der Molen et al. 1990). New processes were also introduced to improve model performance, such as a separate slurry layer. In such models a key challenge is to simulate the pH of the emitting layer, which may be rather different from, or independent of, the background pH value for the underlying topsoil, e.g. in cases where infiltration is limited. Also, for implementation in CTMs, regional soil pH maps need to account for the effects of liming practices.

Empirical/statistical regression approaches for slurry emissions include the Ammonia Loss from Field-applied Animal Manure (ALFAM) model (Segaard et al. 2002), whereby volatilisation is described mathematically by a Michaelis-Menten-type equation, with the loss rates as the response variable, and soil water content, air temperature, wind speed, slurry type, dry matter content of slurry, total ammoniacal nitrogen content of slurry ($\text{TAN} = [\text{NH}_4] = [\text{NH}_3] + [\text{NH}_4^+]$), application method and rate, mode of slurry incorporation and measuring technique are the explanatory variables. Similarly, using regression analysis, Menzi et al. (1998) used the results of field and wind tunnel experiments to derive a simple empirical model to estimate ammonia emissions after the application of liquid cattle manure on grassland. Their model takes into account the mean saturation deficit of the air, the TAN content of the manure and the application rate. Lim et al. (2007) proposed an artificial neural network (ANN) approach for predicting ammonia
emission from field-applied manure, which combined principal component analysis (PCA)-based preprocessing and weight partitioning method (WPM)-based post-processing. Their so-called PWA (standing for PCA-WPM-ANN) approach is expected to account for the complex nonlinear effects between the \( \text{NH}_3 \) emission variables such as soil and manure states, climate and agronomic factors.

For soils amended with commercial fertilizers, such as anhydrous \( \text{NH}_3 \), urea, ammonium nitrate, or mixtures of these forms, soil \( \text{NH}_3 \) emission is modelled in the Community Multiscale Air Quality (CMAQ) model (Foley et al. 2010) by a simplified version of the US Department of Agriculture’s Environmental Policy Integrated Climate (EPIC) model (Williams et al. 2008; Cooter et al. 2010), which includes simulation of nitrification through a combination of a first-order kinetic rate equation (Reddy et al. 1979) and elements of the Crop Environment REsource Synthesis (CERES) crop model (Godwin et al. 1984). The rate of \( \text{N} \) transformation is computed as a function of soil pH, temperature, and soil moisture effects on nitrification and subsequent volatilization. In EPIC, volatilization is simply a fixed fraction of nitrification, while the CMAQ-EPIC coupling application makes use of the bi-directional flux paradigm to characterize the emission. One basic hypothesis of the simplified EPIC processes included in CMAQ is that characterization of the nitrification process alone will adequately simulate the concentration of \( \text{NH}_4^+ \) and \( \text{H}^+ \) in agricultural soils. The upper 15–45 cm of the soil layer reflects the impact of specific tillage practices on biogeochemical process rates. The EPIC/CMAQ method requires knowledge of physical properties of the ambient soil profile, meteorology, and regional crop management practices and uses a crop growth model to estimate tillage and fertilizer application timing and amount. This information is provided to CMAQ by a full EPIC management simulation. The EPIC model also can perform detailed dynamic slurry or solid form manure simulations, but this information is not yet implemented in the current coupling with CMAQ. For \( \text{NH}_3 \) transfer to the surface, the EPIC/CMAQ model formally develops and evaluates refinements to the Nemitz et al. (2001a) model for \( \text{NH}_3 \) flux over a managed agricultural soil, that includes a soil resistance term (see Section “Canopy/Ecosystem Scale Models”). A similar approach was also developed using the Volt’Air \( \text{NH}_3 \) emission module and the CERES-EGC crop growth model (Gabrielle et al. 1995; Theobald et al. 2005).

Although strictly speaking not pertaining to the manure or fertilizer categories, \( \text{NH}_3 \) emissions from seabird excreta (guano) on the ground of land-based colonies present similarities and their study and modelling proves relevant in this context. Agricultural sources of \( \text{NH}_3 \) are complicated by different management practices across the globe, whereas seabird emissions represent a model system for studying climate dependence (Riddick et al. 2012). Seabird colonies are the largest point sources of ammonia globally (up to ~6 Gg \( \text{NH}_3 \) colony\(^{-1}\) year\(^{-1}\), on average; Blackall et al. 2007). Riddick et al. (2012) present an \( \text{NH}_3 \) emission mid estimate with an overall uncertainty range of 270 [97–442] Gg \( \text{NH}_3 \) per year for seabird colonies globally. In the Generation of emissions from Uric Acid Nitrogen Outputs (GUANO) model (Blackall et al. 2007; Riddick 2012), the emission of \( \text{NH}_3 \) from seabird excreted \( \text{N} \) is described in four steps: (i) Excretion of nitrogen rich guano, in the form of uric acid based on a seabird energetics model (Wilson et al. 2004);
Advances in Understanding, Models and Parameterizations …

(ii) conversion of uric acid total ammoniacal nitrogen (TAN), with a climate- and surface pH-dependent rate; (iii) TAN partition between \( \text{NH}_4^+ \) and \( \text{NH}_3 \) on the surface; and (iv) \( \text{NH}_3 \) volatilization to the atmosphere, controlled by the wind speed, aerodynamic resistance (\( R_a \) and \( R_b \)) and the fraction of \( \text{NH}_3 \) re-absorbed by the substrate and re-captured by any overlying vegetation.

The review by Sintermann et al. (2012) of published \( \text{NH}_3 \) emission factors for field applied slurry showed that (i) very substantial differences between EF estimates from field-scale (both AGM and EC) measurements and the ALFAM and Menzi et al. (1998) simple empirical models, for Swiss datasets (e.g. Spirig et al. 2010; Sintermann et al. 2011), with estimates TAN losses in the range 5–30 % by measurements versus 20–70 % by these two models; and (ii) that EF estimates by measurements depended on the spatial scale at which they were carried out (chamber, small or medium plot, field), suggesting strong potential methodological biases. This provides a very clear indication that the current level of validation for models of \( \text{NH}_3 \) volatilisation from field applied manures is rather poor. The authors concluded that new series of measurements are urgently needed in order to (i) provide systematic comparisons of measurements from medium-scale plots and field-scale measurements under identical conditions, and using a range of different measurement techniques, and (ii) pursue the characterisation of \( \text{NH}_3 \) EFs in terms of the influence of slurry composition and application method, soil properties and meteorology. Such future experiments should ideally cover the detailed temporal dynamics (hourly or better over the full course of emission) to help understand the environmental interactions, and must report on the parameters required to perform a plausibility check and to apply and develop process-oriented models.

**Litter Emissions (\( \Gamma_{\text{litter Emission Potential}} \))**

The model developed by Nemitz et al. (2000a) to simulate the dynamics of the litter \( \text{NH}_3 \) emission potential, based on measurements of \( [\text{NH}_4^+]/[\text{H}^+] \) ratio in bulk tissue extracts and on mineralization and nitrification rates, is one of very few available methods at present and appears to be relatively easy to implement. A more detailed mechanistic treatment is provided by EPICv.0509 (see Appendix A in Cooter et al. 2012; Williams et al. 2008), in which soil organic C and N are split into three compartments: microbial biomass, slow humus and passive humus, and organic residues added to the soil surface or belowground are split into metabolic and structural litter compartments as a function of C and N content. Following the CENTURY (Parton et al. 1994) approach, EPIC includes linear partition coefficients and soil water content to calculate movement as modified by sorption, which are used to move organic materials from surface litter to subsurface layers; temperature and water controls affecting transformation rates are calculated internally in EPIC; the surface litter fraction in EPIC has a slow compartment in addition to metabolic and structural litter components; while lignin concentration is simulated as an empirical sigmoidal function of plant age.
Although the NH$_3$ emission potential of the litter ($\Gamma_{\text{litter}}$) is very high, especially in fertilized agricultural systems (Fig. 3), this component has been very much understudied compared with, say, apoplastic $\Gamma_s$. Within the European Union-funded collaborative project ECLAIRE ("Effects of Climate Change on Air Pollution and Response Strategies for European Ecosystems"; http://www.eclaire-fp7.eu), work is on-going to characterise NH$_3$ emission potentials in a range of litter samples from selected ECLAIRE monitoring sites across Europe. The incubation of litter samples in a two-factorial design of different soil moistures (20–80 % water-filled pore space) and temperatures (5–20 °C) should provide a better understanding of litter emission dynamics.

**Leaf/Plant-Scale Stomatal Exchange ($R_s$ Emission Potential)**

Substantial progress has been achieved over the last 10 years in modelling the cell and plant physiological mechanisms that determine the apoplastic $\Gamma_s$ ratio and its temporal variations. In particular, the Pasture Simulation (PaSim) ecosystem model for the simulation of dry matter production and C, N, H$_2$O and energy fluxes (Riedo et al. 1998), accounts for the effects of nitrification, denitrification and grazing, and was extended by Riedo et al. (2002) to couple NH$_3$ exchange with ecosystem functioning. For this purpose, the above-ground plant substrate N pool in previous versions of PaSim was sub-divided into apoplastic and symplastic components. The apoplastic substrate N pool was linked to the stomatal NH$_3$ exchange, while soil ammoniacal N (NH$_x$) was partitioned between the soil surface and several soil layers, with the soil surface NH$_3$ exchange being driven by the NH$_4^+$ content in a soil surface layer (set at 0–3 mm depth). This was the first attempt by any model to account for plant N nutrition and development stage in predicting $R_s$. One significant drawback identified by the authors was that PaSim did not consider the form of N taken up by the roots (reduced or oxidised), which may be significant since plants absorbing NH$_4^+$ have higher NH$_3$ emissions compared with plants absorbing NO$^-$ (see Section “Plant Physiological Controls”). Riedo et al. (2002) offered this as an explanation for the lack of late summer emissions in their simulations, in contrast to observed fluxes in a Scottish pasture.

Another significant development is the stomatal ammonia compensation point (STAMP) leaf-scale model for C3 plants by Massad et al. (2010a), in which $\Gamma_s$ is likewise related to plant N and C metabolism. Here, five compartments are considered explicitly: xylem, cytoplasm, apoplas, vacuole and sub-stomatal cavity, while the main processes accounted for are (i) transport of NH$_4^+$, NH$_3$ and NO$^-$ between the five compartments; (ii) NH$_4^+$ production through photorespiration and NO$^-$ reduction; (iii) NH$_4^+$ assimilation by the GS/GOGAT cycle; (iv) chemical and thermodynamic equilibriums in all the compartments; and (v) stomatal transfer of NH$_3$ (Fig. 5). In contrast to PaSim, STAMP accounts for either NH$_4^+$-based nutrition, NO$^-$-based nutrition, or a combination of both. However, STAMP only represents a leaf (single-layer canopy) in a vegetative stage of growth, in which apoplast and cytoplasm are relatively uncoupled with respect to NH$_x$; STAMP
does not account for the effects of senescence on NH$_3$ metabolism, restricting model applicability in the case of plants having senescent leaves and in multilayered canopies. STAMP was validated against measured $\chi_s$ values and both apoplastic and intracellular NH$_x$ concentrations, using flux chamber measurements with 7–9 oilseed rape plants at 5 weeks of age (Massad et al. 2009). The model has yet to be scaled up to the crop canopy level, integrating soil and plant processes, which will also require the model to be thoroughly tested against field data.

Unlike PaSim and STAMP, the stomatal compensation point model integrated by Wu et al. (2009) in the MultiLayer BioChemical (MLBC) dry deposition model of Wu et al. (2003) is not driven by ecosystem, plant and leaf biochemistry and

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**Fig. 5** Components and flow diagram of the STAMP (stomatal ammonia compensation point) model by Massad et al. (2010a). **Oneway arrows** represent active transport, **two-way arrows** represent passive diffusion, **dotted arrows** represent equilibria and **red arrows** represent forcing variables.
metabolism, but it does explore from a theoretical viewpoint the issue of potential feedbacks between emission, deposition and leaf temperature on the dynamics of apoplastic $I_s$. Simulations show that modeled apoplastic $[\text{NH}_4^+]$ and $[\text{H}^+]$ display significant diurnal variation when the buffer effect of the underlying metabolic processes generating or consuming $\text{NH}_4^+$ are ignored, and that the model predictive capability for canopy-scale exchange fluxes over fertilized soybean (measurements by Walker et al. 2006) is slightly improved by incorporating the feedback of $\text{NH}_3$ flux on apoplastic $[\text{NH}_4^+]$ (vs. a constant $I_s$ approach). Ignoring entirely the apoplastic buffer effects associated with xylem supply and cytoplasmic exchange appears to be an unrealistic oversimplification, but the dynamic stomatal compensation point MLBC runs by Wu et al. (2009) do raise the issue of the significance for modelling of diurnal $I_s$ variations, which have been observed elsewhere (e.g. Herrmann et al. 2009), albeit of a smaller magnitude.

**Leaf Surface Aqueous Chemistry ($I_d$ Emission Potential)**

Water droplets resting on leaf surfaces have long been known to act as sinks for soluble atmospheric trace gases including $\text{SO}_2$ (Brimblecombe 1978; Fowler and Unsworth 1979) and $\text{NH}_3$ (Sutton et al. 1992). Although leaf wetness is usually assumed to increase surface affinity (i.e. reduce surface resistance) for $\text{NH}_3$ uptake, Sutton et al. (1995c, 1998a) recognized that exchange with leaf surface water could be reversible and they developed the first capacitance-based model to simulate $\text{NH}_3$ desorption from the drying out cuticle of a wheat canopy. One underlying assumption was that part of the previously deposited $\text{NH}_3$ was not fixed by reaction to form low vapour pressure salts (e.g. $(\text{NH}_4)_2\text{SO}_4$) and thus may be released back to the atmosphere upon evaporation of surface wetness, with this leading to an increase in $[\text{NH}_4^+]$ in the leaf surface water pool, and the associated values termed $T_d$ and $x_d$. The water film thickness ($M_{\text{H}_2\text{O}}$), which scaled by LAI determines the bulk canopy leaf surface water storage ($M_{\text{cH}_2\text{O}}$), was estimated on the basis of relative humidity at the surface (Sutton et al. 1998a; van Hove et al. 1989; Burkhardt and Eiden 1994). The treatment of leaf surface wetness as a dynamic pool of $\text{NH}_4^+$, with periods of pool contraction (evaporation) followed by periods of expansion (dewfall, rainfall), meant that the bi-directional cuticular $\text{NH}_3$ flux (into or out of the adsorption capacitor $C_d$) was dependent on previous fluxes (hysteresis). The charging resistance ($R_d$) was calculated as $R_d$ (s m$^{-1}$) = 5000/$C_d$, equivalent to an 83 min time constant, and the $\text{NH}_3$ surface reaction rate ($K_r$) and surface solution pH (needed to calculate $I_d$) were both prescribed.

The Sutton et al. (1998) $C_d/R_d$ simple dynamic approach was subsequently adopted by Neirynck and Ceulemans (2008) for Scots Pine forest; here, however, water film thickness was calculated as a function of the normalized output of a leaf wetness sensor (LW), while parameterizations of both $K_r$ and surface pH were obtained by optimizing the model results to minimize bias and maximize the $R^2$ between observed and modelled fluxes.
A significant development of the capacitance model was provided by Flechard et al. (1999), here termed DEWS (Dynamic pollutant Exchange with Water films on vegetation Surfaces), originally developed in moorland vegetation. This model has since been applied for managed grassland to the Braunschweig flux dataset (Burkhardt et al. 2009). By contrast to the Sutton et al. (1998a) and Neirynck and Ceulemans (2008) implementations of the $C_d R_d$ model, in which leaf surface solution pH was prescribed or statistically optimized, the dynamic chemistry model of Flechard et al. (1999) simulated solution chemistry, pH and $F_d$ mechanically, where Henry and dissociation equilibria were forced by measured ambient concentrations of the trace gases NH$_3$, SO$_2$, CO$_2$, HNO$_2$, HNO$_3$ and HCl. The oxidation of SO$_2$ to SO$_4^{2-}$ by O$_3$, O$_2$ and H$_2$O$_2$ and the exchange of base cations and NH$_4^+$ between the leaf surface and plant interior were also accounted for. The cuticular adsorption resistance ($R_d$) was parameterized as an exponential function of the ionic strength of the solution. Activity coefficients were included in the numerical calculations of the equilibrium pH and solute concentrations for solutions with ionic strengths up to 0.3 M.

Although mechanistically satisfying, and successful in field-scale studies, these dynamic chemistry models to simulate surface-wetness-related NH$_3$ fluxes are computationally intensive, requiring short time steps (seconds to minutes), and thus they have not been implemented until now in large-scale models such as CTMs. Most models use unidirectional, steady-state cuticular resistance approaches for leaf surface wetness, in which no $\Gamma_d$ is assumed. Instead, the non-stomatal resistance to deposition, associated with the epifoliar NH$_3$ sink and termed $R_w$ here (or $R_{ext}$, or $R_{ns}$, or $R_{cut}$, in different models; e.g. Flechard et al. 2011), typically decreases with increasing RH (or increases with VPD), to reflect the larger sink strength of wet surfaces. The effect of pH on NH$_3$ uptake rates is reflected, in some models or parameterizations, in the dependence of $R_w$ on the atmospheric molar ratio of SO$_2$/NH$_3$ or Total Acids/NH$_3$ (e.g. Erisman et al. 1994; Nemitz et al. 2001a; Massad et al. 2010b; Simpson et al. 2012), or simply on the NH$_3$ concentration itself (Jones et al. 2007). Figure 6 shows the exponential decay curve fitted to a compilation of published $R_w$ values (at 95 % RH) as a function of the Total Acids/NH$_3$ ratio, at a range of NH$_3$ flux measurement sites, for four major ecosystem types (Massad et al. 2010b). Despite a substantial scatter, there is no question that, at sites where the pollution climate is dominated by NH$_3$, non-stomatal uptake is severely restricted by a high pH and high surface [NH$_4^+$] (e.g. high $\Gamma_d$).

A “hybrid” non-stomatal NH$_3$ exchange modelling concept, half-way between capacitance ($\Gamma_d > 0$, bi-directional) and resistance ($\Gamma_d = 0$, deposition-only) models, was developed within the DEPosition of Acidifying Compounds (DE-PAC 3.11) model by Wichink-Kruit et al. (2010) and Van Zanten et al. (2010). Their model recognized the existence of a non-zero $\Gamma_d$ emission potential (which they termed $\Gamma_w$), which increased with ambient NH$_3$ concentration at a given site. However, the parameterization of the external leaf surface pathway was not truly bi-directional, since the equivalent $\chi_d$ (or $\chi_w$) was approximately parameterized as a fraction of the ambient air concentration ($\chi_a$), and thus $\chi_d$ never exceeded $\chi_a$. 
Nonetheless, the parameterization accounted for saturation effects at high air concentrations, in a similar fashion to e.g. the NH$_3$-dependent $R_w$ of Jones et al. (2007), with the difference that non-zero values of $\Gamma_d$ and $\chi_d$ were mechanistically more realistic. In making this modification, much of the uncertainty in the dependence of the cuticular exchange on the pollution climate and ecosystem was transferred from $R_w$ to $\chi_w$. While the exact partitioning between the two terms remained uncertain, the hybrid approach had the advantage of accounting, in theory, for the bi-directional and concentration-dependent exchange with the leaf cuticle, while avoiding the requirement for more complex time-dependent dynamic modelling solutions.

Air Column Chemistry

Nemitz (2012) present a comprehensive review of models dealing with acid gases, aerosols and their interactions with NH$_3$, and thus only a brief overview is given here. Several numerical models have been developed for the implementation of modified gradient techniques to infer the surface flux of NH$_3$ and chemically reactive species from profile measurements and accounting for GPIC effects on vertical flux divergence (Brost et al. 1988; Kramm and Dlugi 1994; Nemitz et al. 1996; van Oss et al. 1988; Nemitz and Sutton 2004; Ryder 2010). Modelling results showed that reactions could theoretically change NH$_3$ fluxes by as much as 40 % (Kramm and Dlugi 1994) or even lead to flux reversal (van Oss et al. 1988).
For the chemical source/sink term associated with the NH$_3$–HNO$_3$–NH$_4$NO$_3$ triad, the kinetics of the chemical inter-conversion can either be described by the use of chemical timescales, reaction rate coefficients, or by using a full model of size-resolved chemistry and microphysics. Brost et al. (1988) were the first to model the effect of the NH$_3$–HNO$_3$–NH$_4$NO$_3$ equilibrium on surface exchange fluxes of NH$_3$, and described the reaction as a first-order relaxation towards equilibrium with a characteristic time $\tau_c$. The later model by van Oss et al. (1988) also described the shift towards equilibrium by a relaxation-type equation for the flux divergence. The first-order relaxation approach received criticism from Kramm and Dlugi (1994), who proposed an alternative model, favouring a reaction rate formulation using rate coefficients for condensation ($k_1$) and evaporation ($k_2$), and coupled with an inferential resistance model for the estimation of surface exchange fluxes from single-point concentration data. Nemitz (1998) argued that both first-order relaxation and reaction rate approaches were actually equally valid, but there are large uncertainties in the reaction rate coefficients (Kramm and Dlugi 1994) and in chemical timescales (Wexler and Seinfeld 1990).

For the calculation of the concentration and flux profiles modified by chemical reactions, additional information linking the flux ($F_\chi$) to atmospheric turbulence is required to solve the vertical flux divergence, i.e. the $\delta F_\chi/\delta z$ differential, which constitutes a so-called closure problem (Nemitz 1998). Second-order closure (SOC) approaches use information from the budget equations of the turbulent fluxes, which include second-moment terms. By contrast, first-order closure (FOC), also called K-closure models, use information provided by the concentrations themselves, implying that K-theory is used for the flux-gradient relationship. SOC tends to be regarded as a reference and should be accurate, but there are difficulties in applying the method to all atmospheric stabilities. By contrast, FOC is much easier to apply in all stabilities, but there are limitations of the applicability of inert K-theory to reactive species. Thus efforts have been made to estimate the magnitude of the error induced by FOC compared with SOC, and to develop modified K-theories and correction procedures (Nemitz 1998).

The effects of ground NH$_3$ emissions on NH$_4$NO$_3$ formation, the extension of existing FOC approaches by the NH$_3$–HCl–NH$_4$Cl triad, and the inclusion of vertical gradients of temperature, relative humidity and aerosol composition were innovative aspects developed by Nemitz et al. (1996) and Nemitz (1998). The numerical model presented by Nemitz and Sutton (2004) took the approach further and developed a modified gradient technique, which explicitly calculated the particle size distribution of the NH$_4^+$ aerosol as a function of height, in addition to the concentration and flux profiles of the bulk aerosol species. From the change of the size distribution with height ($z$), apparent aerosol deposition velocities could be inferred, which may be compared with values derived from eddy-covariance (EC) measurements, e.g. using optical particle counters. With the knowledge of the size distribution it became also possible to calculate the chemical timescale ($\tau_c$) of the equilibration process (Wexler and Seinfeld 1990) as a function of the size distribution at each height. Ryder (2010) took this approach another major step forward, by modeling the evolution of a mixed, size-distributed aerosol in a fully
coupled model treating transport, emission/deposition, chemistry, phase transition and aerosol microphysics in a multi-layer approach, which also resolved chemical interactions within the canopy. All previous approaches were based on single-layer (big-leaf) exchange models.

The advances in GPIC/flux interaction modelling over the last 15 year have therefore been very substantial, but models have not yet been applied on a routine basis at spatial scales larger than the field. Also, despite the increasing availability of multiple gas and aerosol species concentrations and fluxes over a range of ecosystems (e.g. Douglas fir forest, van Oss et al. 1988; oilseed rape, Nemitz et al. 2000b; heathland, Nemitz et al. 2004, Nemitz and Sutton 2004; tropical pasture, Trebs et al. 2004; grassland, Nemitz 1998, Nemitz et al. 2009b, Wolff et al. 2010a, b), model results have only rarely been compared with measurements. Significant future model improvements could be anticipated from a systematic processing of all existing datasets and from conducting model sensitivity analyses of the minimum complexity required to reproduce measurements adequately. It should be noted that, in general, the relative effect of GPIC on fluxes of acids and aerosols is larger than that on NH3 (Nemitz et al. 2012).

Canopy/Ecosystem Scale Models

Canopy-scale models integrate component processes and their interactions within SVAT frameworks, with the objective of predicting the net ecosystem NH3 flux from the inputs of: (i) ambient NH3 and other concentrations (χa); (ii) meteorology (global and net radiation, temperature, relative humidity or VPD, wind speed, and friction velocity, sensible, latent and ground heat fluxes if available); and (iii) ecosystem characteristics such as LAI, canopy height (hc). Model concepts range from simple, steady-state, “Big-Leaf” canopy resistance (Rc)/deposition velocity (Vd) approaches, to complex, dynamic, multiple-layer canopy compensation point schemes. Most models are based on the resistance analogy, in which the flux (Fχ) between two potentials A and B is equal to the potential difference (χA−χB) divided by the resistance (RA,B), with the soil-canopy-atmosphere system being represented as a network of potentials connected by resistances in series (for different layers) and in parallel (for different pathways) (e.g. Monteith and Unsworth 1990).

Canopy Resistance (Rc) Models

Canopy resistance/deposition velocity (Rc/Vd) models (e.g. Baldocchi et al. 1987; Wesely 1989; Erisman et al. 1994; see review by Wesely and Hicks 2000) simulate NH3 dry deposition to the surface, whereby Rc is the total resistance to deposition resulting from component terms such as stomatal (Rs), mesophyll (Rm),
non-stomatal/external/cuticular ($R_w$ or $R_{\text{Soil}}$ or $R_{\text{ext}}$ or $R_{\text{cut}}$), or soil ($R_{\text{Soil}}$ or $R_g$) resistances (Fig. 7a). $R_a/V_d$ models assume a zero NH$_3$ emission potential in the canopy, and thus the exchange is uni-directional (deposition-only). The deposition velocity is calculated as the inverse sum of $R_c$ in series with the aerodynamic ($R_a$) and viscous sub-layer ($R_b$) resistances above the canopy, and the flux $F_X$ as the product of NH$_3$ concentration ($\chi_a$) and $V_d$:

$$V_d(z) = (R_a(z) + R_b + R_c)^{-1}$$  \hspace{1cm} (1)

$$F_X = V_d(z) \times \chi_a(z)$$  \hspace{1cm} (2)

where $V_d$, $R_a$ and $\chi_a$ are all expressed at the same reference height ($z$) above d, the displacement height. The resistances $R_a$ and $R_b$ are relatively well characterised and readily calculated from micrometeorological measurements (e.g. Monteith and Unsworth 1990; Garland 1977). Stomatal resistance to gaseous transfer is typically derived in the different models using a generic light-response function within

Fig. 7 Typical surface/atmosphere schemes for the modelling of net canopy-scale NH$_3$ fluxes. 

a Generic example of canopy resistance ($R_c$) model; b the 1-layer $\chi_s/R_w$ canopy compensation point model by Sutton et al. (1995b); c the 2-layer $\chi_s/\chi_g/R_w$ canopy compensation point model by Nemitz et al. (2001a); d the 3-layer (soil, foliage, silique/inflorescence) canopy compensation point model by Nemitz et al. (2000b); e the 1-layer $\chi_s/\chi_d/R_d$ capacitANCE canopy compensation point model by Sutton et al. (1998a); and f the 2-layer $\chi_s/\chi_g/\chi_d/R_d$ dynamic chemistry canopy compensation point model by Burkhardt et al. (2009).
a multiplicative algorithm also accounting for $\Gamma$, VPD and SWC stress factors (Jarvis 1976; Emberson et al. 2000a, b).

Some models split PAR into its direct and diffuse fractions and compute the sunlit and shaded components of LAI, such that total (or bulk) stomatal resistance is calculated from sunlit and shaded resistances weighted by their respective LAI fractions (Baldocchi et al. 1987). By contrast the much simpler $R_s$ routine by Wesely (1989) only requires global radiation and surface temperature as input, and may be used when land use and vegetation characteristics are not well known.

Canopy resistance models often use a Big-Leaf approach, i.e. they do not distinguish several layers vertically in the canopy, nor do they simulate in-canopy turbulent transfer, and vegetation is thus assumed to behave as one single leaf. Such models can nonetheless include an in-canopy aerodynamic resistance term ($R_{ac}$) in series with $R_{soil}$ (e.g. Wesely 1989; DEPAC, Erisman et al. 1994; European Monitoring and Evaluation Programme (EMEP), Simpson et al. 2012; A Unified Regional Air-quality Modelling System (AURAMS), Zhang et al. 2003). Most of the existing $R_c$ model variants, alongside specific innovations, actually borrowed model parts and parameterizations from other models, e.g. PLant ATmosphere INteractions (PLATIN, Grunhage and Haenel 1997), drawing on Wesely (1989), Sutton et al. (1995b) and DEPAC; or SPRUCE forest DEPosition (SPRUCEDEP, Zimmermann et al. 2006), drawing on PLATIN, Wesely (1989), DEPAC, EMEP and AURAMS.

In contrast to big leaf $R_c$ models, the MLBC dry deposition model proposed by Wu et al. (2003), based on the Multi-Layer Model (MLM) by Meyers et al. (1998), described gaseous exchange between the soil, plants, and the atmosphere. A biochemical stomatal resistance model based on the Berry-Farquhar approach (Berry and Farquhar 1978) described photosynthesis and respiration and their coupling with stomatal resistance for sunlit and shaded leaves separately. Various aspects of the photosynthetic process in both C3 and C4 plants were considered in the model. The source/sink term $S(z)$ was parameterized using terms to account for fluxes through the stomata of sun-lit and shaded leaves, and for fluxes through the cuticles of the leaves. The canopy was divided into $N = 20$ equally spaced levels, and $S(z)$ was evaluated at each height, and summed with appropriate normalization. Vertical leaf area density LAI($z$) was assumed to be described by a beta distribution (Massman 1982), which was chosen for compatibility with the roughness length and displacement height model of Massman (1997). Plant canopy structures were fit by one of six typical vertical profiles.

**Canopy Compensation ($\chi_c$) Point Models**

The recognition that there is a non-zero NH$_3$ emission potential ($\Gamma'$) in most vegetation types, as well as in different parts of the canopy (Section “Processes Controlling NH$_3$ Emission and Uptake in the Soil/Plant/Atmosphere Continuum”), has led to the development of a range of canopy compensation point ($\chi_c$) models, in which the net bi-directional flux to or from the atmosphere is provided generically from the difference between $\chi_c$ and
For the formulation of $\chi_c$ itself, various canopy architectures have been put forward. The first $\chi_c$ model was developed by Sutton et al. (1995b, 1998a) and is often referred to as the “two-leg” $\chi_c$ model (Fig. 7b), featuring bidirectional exchange with stomata and deposition to non-stomatal surfaces. Here the $R_w$ term accounted for all non-stomatal canopy sink terms, including leaf cuticle waxes and water, and allowed both deposition from the atmosphere as well as re-capture of NH$_3$ emitted by stomata. The canopy compensation point was calculated as (Sutton et al. 1995b):

$$\chi_c = \frac{\chi_a[z]}{R_a[z] + R_b} + \frac{\chi_s}{R_s} + \frac{\chi_g}{R_g}.$$

(4)

This 1-layer framework has been successfully applied for situations in which the canopy was closed and/or where soil NH$_3$ emission was negligible. However, where soil or litter NH$_3$ emission took place and dominated the canopy-scale flux, very large and unrealistic apoplastic $\Gamma_s$ ratios (compared with independent estimates by apoplastic bioassays) were required to simulate the observed net emissions (Milford 2004). The 2-layer model by Nemitz et al. (2001a) was thus the logical extension of the 1-layer $\chi_s/R_w$ model, introducing, in addition to stomatal $\chi_s$ and non-stomatal $R_w$, a soil + ground surface emission potential (termed $\chi_g$ in Fig. 7c), mediated by in-canopy $R_{ac}$ and by a further ground surface viscous sublayer term ($R_{bg}$). This $\chi_s/\chi_g/R_w$ model has been extensively tested and applied in diverse contexts, and was proposed as the optimum compromise between simplicity and accuracy, capable of describing bi-directional NH$_3$ exchange in atmospheric transport models over a very wide range of vegetation types (Nemitz et al. 2001a; Massad et al. 2010b; Cooter et al. 2010). As with the 1-layer $\chi_c$ model, the central term in solving the resistance model is $\chi_c$, the resolution of which provides (Nemitz et al. 2001a):

$$\chi_c = [\chi_a(R_aR_b)^{-1} + \chi_s((R_aR_s)^{-1} + (R_bR_s)^{-1} + (R_gR_s)^{-1}) + \chi_g(R_bR_g)^{-1}] \\
\times (R_aR_b)^{-1} + (R_gR_s)^{-1} + (R_gR_w)^{-1} + (R_bR_g)^{-1}.$$

(5)

A three-layer model was also developed by Nemitz et al. (2000b), to account for a third potential NH$_3$ emission/uptake layer in the inflorescences or siliques at the top of an oilseed rape canopy, in addition to foliar and ground exchange. Here, two terms were defined for $R_{ac}$ ($R_{ac1}$ from siliques to foliage, $R_{ac2}$ from foliage to ground), as were two $R_b$ terms and two $R_w$ terms for the siliques and foliage layers (Fig. 7d). The authors concluded that the leaf stomata were an effective NH$_3$ sink, whereas the leaf litter dominated nighttime emissions with the siliques layer thought to dominate daytime emissions.
As modelled fluxes are highly sensitive to soil and plant surface temperatures (Section “Thermodynamic and Chemical Controls”), an accurate description of in-canopy vertical profiles of temperature is highly desirable, such that each potential through the profile (Section “Vertical Distribution of Sources and Sinks Withinand Above Ecosystems”) is expressed with the proper temperature scaling. Thus the Surface Atmosphere (SURFATM)-NH3 SVAT model of Personne et al. (2009) coupled an energy budget model (Choudhury and Monteith 1988) with a pollutant exchange model, which was based on the $\chi_s/\chi_g/R_w$ model of Nemitz et al. (2001a), and additionally included a diffusive resistance term from the topsoil layer to the soil surface. In a 3-week simulation for the Braunschweig grassland, Personne et al. (2009) demonstrated that the energy balance model was suitably adapted for modelling the latent and sensible heat fluxes as the grass was cut then fertilized, based on prescribed (measured) values LAI and $h_c$. The model reproduced the temperatures of leaf and ground surfaces satisfactorily, except for a few days during which the cut grass lay on the ground prior to lifting. The model was later successfully validated against a two-month flux measurement period over a triticale canopy, where is was found that a very small cuticular resistance ($R_w < 1 \text{ s m}^{-1}$ at RH $> 75 \%$, $R_w = 32 \text{ s m}^{-1}$ at RH $= 50 \%$) was required to explain the observed fluxes (Loubet et al. 2012). In a similar fashion to SURFATM-NH3, in the Wu et al. (2009) NH3 stomatal compensation point version of the Wu et al. (2003) MLBC model (see above), the scheme was re-parameterized in order to derive leaf temperature from the energy balance at each level ($z$) in the canopy.

Elsewhere, earlier $R_c$ models have also been modified to include a surface NH$_3$ compensation point, such as: the surface exchange scheme within AURAMS (Zhang et al. 2003, 2010) with a 2-layer $\chi_s/\chi_g/R_w$ structure; the revision of the DEPAC model (Erisman et al. 1994) into DEPAC3.11 with a 1-layer $\chi_s/\chi_w/R_w$ structure (van Zanten et al. 2010; Wichink-Kruit et al. 2010); a revised $\chi_s/R_w$ version of PLATIN (Griinhage and Haenel 2008); or the inclusion of the 2-layer $\chi_s/\chi_g/R_w$ by Nemitz et al. (2001a) into CMAQ for managed agricultural soils (Cooter et al. 2010, 2012; Bash et al. 2013) (see parameterization details below).

A further degree of complexity has been added by leaf surface NH$_x$ capacitance approaches, as an alternative to the steady-state, uni-directional $R_w$ pathway in the $\chi_c$ models described above (Fig. 7b–d). Dynamic numerical solutions for the variable non-stomatal leaf surface NH$_x$ pool have been grafted onto 1-layer (Fig. 7e; Sutton et al. 1998a; Flechard et al. 1999; Neirynck and Ceulemans 2008) and 2-layer (Fig. 7f; Burkhardt et al. 2009) $\chi_c$ models. For individual sites, such models tend to improve the overall model predictive capability only marginally, compared with steady state $R_w$-based $\chi_c$ models that have been optimised with site-specific parameterizations, i.e. an $R_w$ function fitted to reproduce local flux data. Nonetheless, the added value of dynamic chemistry approaches for the leaf surface is three-fold: (i) to better explain the temporal dynamics of emissions; (ii) to allow bi-directional cuticular exchange and NH$_3$ desorption, especially for the morning peak; and (iii) in theory, to predict the leaf surface sink/source strength in a generic and mechanistic fashion, mostly driven by the local pollution climate and atmospheric acid/base mixing ratios, without the need for site-specific, empirical parameterizations for $R_w$ (Flechard et al. 1999). This means that such
an approach is more suitable for regional-scale and global applications where the site-specific optimised parametrizations are not generally and systematically applicable.

Parameterization Schemes for $\chi_c$ Models

The canopy compensation point models presented above proposed generic frameworks, which for individual ecosystems or flux measurement sites require an optimisation with locally fitted parameters or functions (e.g. $\Gamma_s$, $\Gamma_g$, $R_w$). Loubet et al. (2012) argue that one drawback of model/flux comparisons at given measurement sites is the non-uniqueness of parameter vectors that best fit the NH$_3$ fluxes: it is for example often difficult to establish whether soil or stomata are the main sources.

To achieve this, it is typically necessary to add additional site evidence, such as bioassay estimates of $\Gamma$ for different ecosystem compartments (e.g. Fig. 3) and to carefully analyze the time course of differences between measurements and the estimates provided by different model approaches.

For generalisation and application of models at larger scales, typically within regional CTMs, several parameterization schemes have been proposed recently. The new parameterizations for the 1-layer ($\chi_s/\chi_w/R_w$) DEPAC 3.11 scheme by Wichink-Kruit et al. (2010) and Van Zanten et al. (2010) were based on a combination of the results of three years of ammonia flux measurements over a Dutch grassland (Lolium perenne/Poa trivialis) canopy and of existing parameterizations from the literature. Values of $\chi_w$ were derived from actual nighttime flux measurements and accounted for the pollution climate of the site, while their derived $R_w$ function mostly reflected surface humidity effects. The observed seasonal variations in $\Gamma_s$ at their grassland site (typically >5000 from autumn until early spring, decreasing to ~1000 in summer, see Fig. 2), presumably reflecting photosynthetic activity and GS/GOGAT activity, and were parameterized as a function of temperature with an exponential decay fit. (Note that Loubet et al. (2012) found a similar exponential decay for $\Gamma_c$ in a triticale canopy in spring). The spatial variations of $\Gamma_s$ were linearly linked to atmospheric pollution levels through the long-term NH$_3$ concentration for given sites, based on a review of literature values. Two linear regressions were proposed, either based on literature $\Gamma_s$ values derived from micrometeorological flux measurements, to be used in 1-layer $\chi_s/R_w$ or $\chi_s/\chi_g/R_w$ models, or based on $\Gamma_s$ values from apoplastic extraction, to be used in 2-or multilayer (e.g. $\chi_s/\chi_g/R_w$) models (see Fig. 7). This distinction was based on the recognition that bioassay-derived $\Gamma_s$ values were typically a factor of 3 lower than micrometeorologically derived values (e.g. Fig. 1d), presumably due to additional contributions by litter and soil emissions to the latter estimates.

Zhang et al. (2010) proposed parameterizations for their 2-layer $\chi_s/\chi_g/R_w$ model within AURAMS based on an extensive literature review. Their approach was to compile a large database of published $\chi_s$ and $\chi_g$ values, and to create a model lookup table (cf Table 5 in Zhang et al. 2010) for both parameters. For each of their
26 land-use classes (LUC), they derived representative model input values based on statistics of literature data. For LUC classes with fertilized vegetation, a much larger value was used (typically factor 10–100) for both $\chi_s$ and $\chi_g$ than for semi-natural ecosystems. For the former (fertilized), one single value was used throughout, while for the latter (semi-natural), both $\Gamma_s$ and $\Gamma_g$ can take either one of two default values, either “high” or “low”, depending on the background atmospheric N input by wet and dry deposition. The parameterization for $R_w$ (leaf cuticle) was unchanged from Zhang et al. (2003) and based on canopy wetness, leaf area, and meteorological conditions (relative humidity, friction velocity), but did not account for differences in pollution climate. Initial model runs showed that typical summer daytime $\chi_c$ values (at a temperature of 25 °C), assuming a low N status, were less than 2 μg m$^{-3}$ over forests and other semi-natural canopies, below 5 μg m$^{-3}$ over grasslands, and between 5 and 10 μg m$^{-3}$ over agricultural crops. In the winter, these values decreased to almost zero over the forests and to below 3 μg m$^{-3}$ over the crops. The application of this new bi-directional air-surface exchange model in replacement of the original dry deposition model will reduce the dry deposition fluxes simulated in the regional scale air-quality model for which it was designed, especially during the daytime and for canopies with high-N status. The reductions in simulated dry deposition fluxes will also be larger at higher temperatures, stronger wind speeds, and drier conditions (Zhang et al. 2010).

Massad et al. (2010b) also made a very comprehensive review of the NH$_3$ flux literature, in order to derive a generalised parameterization scheme for the 2-layer $\chi_s/\chi_g/R_w$ model by Nemitz et al. (2001a). Although their parameterizations were intended for application in any CTM, their scheme was to some extent tailored to fit the LUC of the EMEP model (Table 6 in Massad et al. 2010b; Simpson et al. 2012). The meta-analysis confirmed that nitrogen input was the main driver of apoplastic [NH$_3^+$] and bulk tissue [NH$_4^+$]. For managed ecosystems, the parameterizations derived for fertilization were reflected in peak value of $\Gamma_s$ and $\Gamma_g$ a few days following application, followed by a gradual return to background values. Fertilizer amounts determined the magnitude of the $\Gamma_s$ response, regardless of fertilizer form (mineral, organic, grazing), and also the scale of the $\Gamma_g$ response for mineral fertilizer. The initial $\Gamma_g$ response to slurry application was equal to the $\Gamma_g$slurry value, while animal grazing resulted in an initial $\Gamma_g$ value of 4000. The sharp temporal decrease in $\Gamma_s$ and $\Gamma_g$ following the initial fertilization or grazing peak was parameterized by an exponential decay function with an e-folding time constant ($\Gamma$) of 2.88 days. For unmanaged ecosystems, as well as managed agrosystems in background conditions, $\Gamma_s$ was parameterized as a power law function of total N input (N$_{in}$) to the ecosystem, i.e. atmospheric N deposition (N$_{dep}$) plus annual fertilizer application (N$_{app}$) if applicable. Although the meta-analysis had demonstrated that the relationship of $\Gamma_s$ to bulk tissue [NH$_3^+$] was more robust than to N$_{in}$ across a wide range of plant species (see also Mattsson et al. 2009a), the use of N$_{in}$ as a proxy for $\Gamma_s$ was deemed more convenient than bulk tissue [NH$_4^+$], which by contrast would not be easily available as spatial input fields for CTMs. The parameterization derived by Massad et al. (2010b) for the leaf surface resistance $R_w$ is discussed above in Section “Process/Component Scale Models: Soil, Manure, Fertilizer, Leaf Litter,
Leaf, Cuticle, Air Column Chemistry” and Fig. 6. One of the major advantages of the Massad et al. (2010b) scheme, compared to the parameterization by Zhang et al. (2010), is the mechanistic linkage of $\Gamma_s$ and $\Gamma_g$ to atmospheric N deposition and to agricultural practices, allowing ecosystems to respond dynamically to changes in emissions and deposition patterns and to land management events.

Cooter et al. (2010) presented an upgrade of the earlier $R_c$-based NH3 dry deposition approach of Wesely (1989) that had been used within CMAQ (Byun and Schere 2006), into a bi-directional $\chi_c$ model based on the $\chi_s/\chi_g/R_w$ approach by Nemitz et al. (2001a). The work was motivated by the realisation that the CMAQ representation of the regional nitrogen budget was limited by its treatment of NH3 soil emission from, and deposition to, underlying surfaces as independent, rather than tightly coupled, processes. At the same time, it was recognized that NH3 emission estimates from fertilized agricultural crops needed to respond to variable meteorology and ambient chemical conditions. These objectives were met by the integration of the $\chi_s/\chi_g/R_w$ approach together with elements of the EPIC model (see Section “Process/Component Scale Models: Soil, Manure, Fertilizer, Leaf Litter, Leaf, Cuticle, Air Column Chemistry”), which was calibrated using data collected during an intensive 2007 maize field study in Lillington, North Carolina (Bash et al. 2010; Walker et al. 2013). More recently, regional simulations of CMAQ coupled with EPIC have provided dynamic continental (US) scale NH3 emission estimates from fertilizer applications with a tight coupling between emissions, deposition and agricultural cropping practices (Cooter et al. 2012; Bash et al. 2013) (see Section “Ammonia Exchange in Chemical TransportModels (CTMs) at Regional Scales”).

**Landscape Scale Models**

The specificity of the landscape scale, especially in agricultural areas, with respect to surface/atmosphere NH3 exchange modelling is characterised by the close proximity of large agricultural point sources, or “hotspots” (Loubet et al. 2009a) and of semi-natural NH3 sink areas such as forests, moorlands and wetlands. Hotspots induce large horizontal NH3 concentration gradients downwind from sources, typically an exponential decay with distance (Walker et al. 2008), and a large spatial heterogeneity in NH3 concentrations (e.g. Dragosits et al. 2002; van Pul et al. 2008) and exchange fluxes (Sommer et al. 2009). This fine-scale variability occurs at spatial scales (typically 100 m to 1 km) much smaller than, and therefore not “seen” by, regional CTMs (resolution typically 5 × 5 km$^2$ to 50 × 50 km$^2$); from a regional modelling viewpoint the (unresolved) landscape scale generally falls under the header “sub-grid issues” (Dragosits et al. 2002). Modelling studies have been applied to determine the fraction of emitted NH3, which is recaptured locally downwind from the source (Fowler et al. 1998; Asman et al. 1998). The results vary widely, showing recapture fractions within the first 2 km between 2 % and up to 60 %, but in most cases in the range between 10 and 40 % (Loubet et al. 2006, 2009a).

The variability is in part due to variations in vegetation types, roughness and LAI over the patchwork of land uses, but also due to the nitrogen enrichment
associated with large NH$_3$ deposition rates close to sources (animal houses, manure storage facilities, fertilized fields) (Pitcairn et al. 2006). Given an otherwise homogeneous, large field (a few hectares) cropped with, say, wheat or maize, and located just outside a large point animal production facility, one may expect a 10- or 20-fold higher NH$_3$ deposition at a distance of 20 m from the source than 200 m further downwind (Loubet et al. 2009a). One may thus also expect much higher bulk tissue N or [NH$_4^+$] and higher $\Gamma_s$ close to the farm buildings, as well as higher NH$_x$ concentrations in soil ($\Gamma_g$) and especially on leaf surfaces ($\Gamma_d$), together with higher pH, which theoretically lead to less efficient NH$_3$ removal by vegetation (per unit ambient NH$_3$ concentration) (Jones et al. 2007). Such feedbacks of cuticular saturation and apoplastic NH$_4^+$ enrichment on NH$_3$ deposition rates (Walker et al. 2008) can potentially affect spatial NH$_3$ deposition budgets very significantly at the scale of the landscape, but uncertainties are very large, datasets are few, and parameterizations to account for N enrichment feedbacks for landscape-scale models have yet to emerge.

These processes and their coupled emission/dispersion/deposition modelling have recently been thoroughly reviewed by Loubet et al. (2009a), and earlier by Hertel et al. (2006) and Asman (1998, 2002), and thus only a brief overview is presented here. Loubet et al. (2009a) provided a technical comparison of 7 existing local atmospheric transport and deposition models for NH$_3$: DDR (Asman et al. 1989); TREND/OPS (Asman and van Jaarsveld 1992), LADD (Hill 1998), DEPO1 (Asman 1998), FIDES-2D (Loubet et al. 2001), MODDAAS-2D (Loubet et al. 2006), and OML-DEP (Olesen et al. 2007). All models except MODDAAS-2-D (multi-layer, see Loubet et al. 2006) used a 1-layer (big leaf) surface exchange architecture, and most models used a uni-directional dry deposition $R_c/V_d$ scheme by default. However, both MODDAAS-2-D and FIDES-2D (Loubet et al. 2001) allowed bi-directional exchange with stomata, though they did not account for any potential soil emissions.

Theobald et al. (2012) presented the first intercomparison of 4 short-range atmospheric dispersion models (ADMS, Carruthers et al. 1999; AERMOD, Perry et al. 2004; LADD; and OPS-st, van Jaarsveld 2004), which they applied to the case of ammonia emitted from agricultural sources. The intercomparison focused on atmospheric NH$_3$ concentration prediction in two case study farms in Denmark and the USA. Wet deposition processes were not included in the simulations because dry deposition is likely the dominant deposition mechanism near sources (Loubet et al. 2009a; Pitcairn et al. 2006). Similarly, chemical processing of NH$_3$ in the atmosphere were also assumed to be negligible for short-range dispersion. Thus the only NH$_3$ removal mechanism involved was surface dry deposition, with all models using $R_c/V_d$ schemes. The performance of all of the models for concentration prediction was judged to be “acceptable” according to a set of objective criteria, although there were large differences between models, depending on which source scenarios (area or volume source, elevation above ground, exit velocity) were tested. The findings highlight that the rate of removal by dry deposition near such a source leads to a rather small effect on simulated near-source NH$_3$ concentrations, which largely depended on sound treatment of source characteristics and dispersion rates.
Ammonia Exchange in Chemical Transport Models (CTMs) at Regional Scales

Despite unequivocal evidence and widespread consensus that \( \text{NH}_3 \) exchange is bi-directional in most climates and ecosystem types, including unfertilized vegetation, most CTMs operating at national, regional and continental scales still use \( R_c/V_d \) deposition-only schemes for \( \text{NH}_3 \) (see model review by van Pul et al. 2009): e.g. unified EMEP MSC-W model (Simpson et al. 2012) and EMEP4UK 5 × 5 km (Vieno et al. 2010); a Wesely (1989) approach is used in CHIMERE (Vautard et al. 2001; LMD 2011); DEPAC is used in OPS-Pro 4.1 (van Jaarsveld 2004); EMEP \( R_c/V_d \) approach is used in the coupled Danish Ammonia Modelling System DAMOS (DEHM/OML-DEP) (Geels et al. 2012); combined DEPAC and EMEP parameterizations in MATCH (Klein et al. 2002); and LUC-specific values of \( R_c \) are used in FRAME (Singles et al. 1998). Nevertheless, a few instances of \( \chi_c \) model implementation in CTMs have recently been reported, using new \( \chi_c \) parameterization schemes (see Section “Canopy/Ecosystem Scale Models”): e.g. the LOTOS-EUROS model (using revised DEPAC 3.11) (Wichink-Kruit et al. 2012); the coupled CMAQ-EPIC model (Cooter et al. 2010, 2012; Bash et al. 2013); and AURAMS (Zhang et al. 2010). Other CTMs have meanwhile focused on improving the treatment of sub-grid variability (DAMOS; Geels et al. 2012) or the spatial and temporal distribution of \( \text{NH}_3 \) emissions by field-applied mineral fertilizers (CHIMERE/Volt’Air, Hamaoui-Laguel et al. 2012).

Canopy Compensation Point Implementations in Regional CTMs

The first test implementation of a \( \chi_c \) approach within a CTM was made by Sorteberg and Hov (1996) using an early version of the EMEP model and the \( \chi_s/R_w \) model by Sutton et al. (1995b, 1998a), but the parameterizations were very crude, with only 2 fixed \( \Gamma_s \) values, 946 and 315 for grassland/cropland and other vegetation types, respectively.

In their LOTOS-EUROS model runs at the European scale (25 × 25 km\(^2\) resolution), Wichink-Kruit et al. (2012) found that by using the bi-directional \( \text{NH}_3 \) exchange scheme by Wichink-Kruit et al. (2010), the modeled ammonia concentrations increased almost everywhere (compared with the \( R_c \)-based model), in particular in agricultural source areas. This was largely due to increased \( \text{NH}_3 \) lifetime and transport distance. As a consequence, \( \text{NH}_3 \) deposition decreased in source areas, while it increased in large nature areas and remote regions (e.g. S. Scandinavia). The inclusion of a compensation point for sea water restricted dry deposition over sea and better reproduced the observed marine background concentrations at coastal locations. Over the land area, the model predictive capability improved slightly, compared with \( \text{NH}_3 \) network data, but concentrations in nature areas were slightly overestimated, while concentrations in agricultural source areas were still underestimated. The authors also discuss the issue of model...
validation using measured NH₃ concentration, related to the representativeness of a single measurement point within a heterogeneous landscape, compared with the modelled grid square average NH₃.

As in most other CTMs, the treatment of the atmospheric NHₓ budget in CMAQ v4.7 traditionally relied on: (i) a unidirectional $R_c$ approach, and (ii) estimates of fertilizer NH₃ emission that were independent of the physical and chemical variables and components of the CTM that simulate atmospheric transport, transformation and loss processes. The coupling of CMAQ v5.0 with EPIC and the Nemitz et al. (2001a) $\chi_s/\chi_g/R_w$ model to simulate the bi-directional exchange of NH₃ (Bash et al. 2013) allowed for the direct estimation of NH₃ emissions, transport and deposition from agricultural practices, with dynamic interactions between weather, soil, vegetation and atmospheric chemistry (Fig. 8). The CMAQ-EPIC coupled model thus shifted the NH₃ emissions modeling paradigm for fertilizer application from static or seasonal emission factors to a more dynamic, process-based approach. Some parameterizations were borrowed from Massad et al. (2010b), but unlike their exponential decay function to adjust $\Gamma_g$ as a function of time after fertilization, the soil NH₄⁺ budget in CMAQ v5.0 was simulated as being dynamically coupled to hourly soil NH₄⁺ losses due to evasion and nitrification, and increases in soil NH₃ due to deposition. Values of $\Gamma_s$ for crops and of $\Gamma_g$ for

non-agricultural soils were modeled as a function of land cover type and ranged from 10 to 160, which were at the low end of published values (e.g. Massad et al. 2010b; Zhang et al. 2010). The new coupled approach improved the predictive capability of CMAQ for NH₃ wet deposition and for ambient nitrate aerosol concentrations. The largest improvements in the aerosol simulations were during the spring and fall, when the US EPA’s national emission inventory estimates at these times are particularly uncertain. In Cooter et al. (2012), the EPIC agro-ecosystem and CMAQ models were used to assess agro-ecosystem management and changes in biogeochemical processes, providing more robust model assessments of future land use, agricultural, energy and climate change scenario analyses.

**Improved Treatment of Sub-grid Variability and Spatial and Temporal NH₃ Emissions**

High spatial resolution deposition modelling is crucial to determine the frequency of occurrence and magnitude of N critical loads and levels exceedances, since many sensitive nature areas and sites of special scientific interest (e.g. wetlands, heathlands, etc.) are very small, say a few hectares, and often located close to agricultural NH₃ sources (Dragosits et al. 2002). As noted above (Section “Ammonia Exchange Models and Parameterizations from the Leaf to the Globe: State-of-the-Art”), this is a landscape scale issue, but it is also a CTM issue, because (i) failing to reproduce local NH₃ budgets affects the predictive capability of regional modelling, and (ii) CTMs must be used to derive critical loads exceedance maps at national and regional scales in support of environmental policy development. Improving the performance of high-resolution local-scale models requires high quality emission inventories with sufficiently high spatial resolution (Skjeth et al. 2011). In addition, a high temporal resolution for emissions is also crucial for the performance of CTMs, and dynamic calculations of NH₃ emissions are needed for a better prediction of high particulate matter episodes (Menut and Bessagnet 2010; Henze et al. 2009). This is especially relevant as NH₃ emissions in winter will lead to a higher contribution to particulate matter than NH₃ emissions in summer.

Data requirements for such models are access to detailed information about activity data and the spatial distribution in emissions on annual basis. Such requirements are met in very few countries, e.g. in Denmark and the Netherlands, where the ammonia emission inventory relies on highly detailed national agricultural registers, containing the exact location of farm houses, storages, and associated fields, as well as data on type and number of livestock, and information about applied production methods (Skjeth et al. 2004). In many other countries agricultural activity and NH₃ emission data are either very crude, based on e.g. default emission factors, and/or confidential at resolutions finer than typically 10 × 10 km².

To address both spatial and temporal issues, the Danish Ammonia Modelling System (DAMOS) has been established as a coupled system consisting of the Danish 3-D Eulerian Hemispheric Model (DEHM) CTM covering the Northern Hemisphere (6 × 6 km² resolution) and of the local-scale (up to ca. 20 km)
Gaussian plume dispersion and deposition model OML-DEP (400 × 400 m² resolution) (Geels et al. 2012). The model may be coupled to a code (Skjeth et al. 2011) for calculating ammonia emission on the European scale, accounting for local climate and local management, in which a modular approach is applied for deriving data as input to the temporally varying ammonia emission model. Comparisons between computed and measured ambient NH₃ concentrations demonstrated considerable improvements in model performance over Denmark when the high spatial and temporal resolution emission inventory was applied, instead of the conventional (static) seasonal variations approach (Skjeth et al. 2004). Further, Geels et al. (2012) showed that the coupled DEHM/OML-DEP model system captured the measured NH₃ time series in Denmark better than the regional-scale model alone, and that about 50 % of the modelled concentration level at a given location originated from non-local emission sources. However, the coupled DAMOS model still overestimated observed local ammonia concentrations across Denmark, which might in part be explained by underestimated national emissions, by underestimated rates of conversion to NH₄⁺ and of dry deposition, and, as in the LOTOS-EUROS case (Wichink-Kruit et al. 2012), by the model grid square size.

Laguel-Hamaoui (2012) coupled the 1-D Volt’Air model (Genermont and Cellier 1997), originally developed for field-applied slurry and adapted here for mineral fertilizers, to the CHIMERE CTM (Vautard et al. 2001; LMD 2011), in order to assess the impact of fertilizer NH₃ emissions on PM10 and NH₄NO₃ aerosol at the national scale. Ammonia emissions were computed from mineral fertilizer spread over agricultural soils, using datasets of crop management practices, soil properties and meteorology. Considerable effort went first into collecting management practices data at the national level, together with data processing to derive their spatial distribution. Three sets of CHIMERE runs were made, using as NH₃ emission inputs to the CTM either (i) the official EMEP data under the CLRTAP convention, (ii) the French national emissions inventory (INS) data, or (iii) a combination of the coupled Volt’Air emissions for mineral fertilizers and INS data for other sources. The three options for NH₃ emission inputs had different impacts on aerosol concentrations, depending on HNO₃ concentrations. The comparison of modelled PM10 and NH₄NO₃ aerosol with observations showed that the new ammonia emission method lent a marginal improvement to the spatial and temporal correlations in several regions and a slight reduction of the negative bias (1–2 μg m⁻³ on average).

Global Scale

Uncertainties in the global NH₃/NHₓ cycle are very large, not least because the NH₃ emission factors typically used for global emission upscaling, and the parameterizations for surface exchange modelling, are heavily biased towards NW European and N American conditions. Some sources are rather well studied, such as livestock agriculture in temperate Europe, while others are based on very few...
atmospheric NH$_3$ flux measurements. The uncertainties are particularly large for natural emissions from terrestrial sources and oceans (Dentener and Crutzen 1994; Bouwman et al. 1997), biomass burning (Andreae and Merlet 2001) and for livestock sources and forests in tropical regions. There is a major lack of knowledge on agricultural management practices in many parts of the world and on the effect of the many climates and soils of the world on emission processes, especially the interplay of temperature and moisture. With 37 % of the world’s population between them, China and India’s collective NH$_3$ emissions account for around 13.5 Tg NH$_3$–N year$^{-1}$ (Huang et al. 2012; Aneja et al. 2012), i.e. about one-third of the EDGAR (2011) global emission estimate of 40.6 Tg NH$_3$–N year$^{-1}$, but subject to huge uncertainty. Aneja et al. (2012) estimate that NH$_3$ emissions from livestock could be a factor of 2–3 higher than their best estimate, while emissions from fertilizer application could be up to 40 % lower than they estimated.

In global atmospheric CTMs, which are coupled to general circulation models (GCMs) or driven by analyzed meteorological fields, and by prescribed emissions of NH$_3$ (e.g. Bouwman et al. 1997) and of other trace gases, ammonia exchange over terrestrial vegetation is generally modelled using $R_c/V_d$ resistance schemes, often following Wesely (1989) (e.g. TM5 model, Huijnen et al. 2010; Ganzeveld and Lelieveld 1995; STOCHEM, Collins et al. 1997; Bouwman et al. 2002; GEOS-Chem Bey et al. 2001; Wang et al. 1998). However, in the MOGUNTIA model at 10 × 10° resolution, Dentener and Crutzen (1994)—who were the first to reconcile by modelling the consistency on a global scale of upscaled NH$_3$ emission inventories and atmospheric NH$_3$/NH$_4^+$ concentrations and deposition—did use a canopy compensation point to calculate NH$_3$ emissions from natural continental ecosystems. Their approach did not distinguish stomatal from non-stomatal (soil, leaf surfaces) contributions, as they applied one set value (equivalent to $\Gamma = 290$) for the canopy, corresponding to [NH$_4^+$] = 46 μmolL$^{-1}$ and pH = 6.8 in the mesophyll, based on measurements over pine forest by Langford and Fehsenfeld (1992). To account for the short atmospheric lifetime and the sub-grid local deposition of NH$_3$, Dentener and Crutzen (1994) directly removed 25 % of all anthropogenic emissions over land, such that these emissions never entered the transport and chemistry calculations. Bouwman et al. (2002) similarly reduced their grid square emissions for the same reason; the fraction of the total emission deposited within a few kilometers from the source depended on many factors, including the height of the source and the surface roughness (Asman 1998), and the compensation point concentration of vegetation.

Dentener et al. (2006) reported a multi-model evaluation (23 global CTMs) of current and future (2030) deposition of reactive nitrogen (NO$_y$, NH$_x$) as well as sulfate (SO$_x$) to land and ocean surfaces. Models predicted that NH$_3$ dry deposition represents between 30 and 70 % of total deposition. Present-day deposition using nearly all information on wet deposition available worldwide showed a good agreement with observations in Europe and North America, where 60–70 % of the model-calculated wet deposition rates agreed to within ±50 % of quality-controlled measurements. However, models systematically overestimated NH$_3$ deposition in South Asia compared with available bulk wet deposition measurements.
There were substantial differences among models for the removal mechanisms of \( \text{NH}_x \), as well as for \( \text{NO}_x \) and \( \text{SO}_x \), leading to \( \pm 1 \sigma \) variance in total deposition fluxes of about 30% in the anthropogenic emissions regions, and up to a factor of 2 outside.

The evaluation/validation of global CTMs for \( \text{NH}_3 \) dry deposition (or surface exchange) is even more difficult than for regional CTMs, with scarce or no \( \text{NH}_3 \) concentration and wet \( \text{NH}_x \) deposition data in many parts of the world, and, where there are data, point measurements being largely de-coupled from the very large grid square modelled averages (typically 1° × 1° to 10° × 10°). Satellite data providing atmospheric column integrated \( \text{NH}_3 \) concentrations have recently offered a very welcome addition (Clarisse et al. 2009; Shephard et al. 2011; R’Honi et al. 2013), but their interpretation can prove complex in a modelling context. Despite a good qualitative agreement between satellite (IASI/MetOp) measurements and simulations by the TM5 global CTM, Clarisse et al. (2009) found that the satellite data yielded substantially higher \( \text{NH}_3 \) concentrations north of 30°N compared with model projections, and lower concentrations than the model south of 30°N. They concluded that ammonia emissions could have been significantly underestimated in TM5 in the Northern Hemisphere, but there were also issues with IASI’s detection limit, limited thermal contrast, and an unrepresentative morning orbit time.

Similarly, Shephard et al. (2011) compared the output of global high-spectral resolution nadir measurements from the Tropospheric Emissions Spectrometer (TES) on NASA’s Aura with GEOS-Chem model runs; initial comparisons showed that TES/Aura values were higher overall. These authors also invoked the possible underestimation of \( \text{NH}_3 \) emissions in the GEOS-Chem inputs, but also possibly the over-representation of \( \text{NH}_3 \) values at the 2° × 2.5° resolution coming from TES sampling \( \text{NH}_3 \) hotspots at the subgrid level. They argued that the better agreement between TES/Aura and GEOS-Chem seasonality over biomass burning regions, compared with agricultural source regions, suggested that the latter may be a more likely source of uncertainty in models.

**Synthesis and Conclusions**

The basic processes controlling surface/atmosphere \( \text{NH}_3 \) exchange are relatively well understood, at least qualitatively. A wide range of factors are important, including: thermodynamics, meteorology, surface and air column heterogeneous chemistry, plant physiology and N uptake, ecosystem N cycling, compensation points, nitrogen inputs via fertilization and atmospheric deposition, leaf litter decomposition, SOM and soil microbial turnover, soil properties. Most of the fundamental process understanding was gained during the 1980s and 1990s, while many advances in modelling logically followed from the late 1990s onwards, spurred by the canopy compensation point concept of Sutton et al. (1995b, 1998a). There has been a gradual increase in the complexity of surface/atmosphere \( \text{NH}_3 \)
exchange models, from simple steady-state $R_c$ models to dynamic, multiple layer, multiple sink/source, multiple chemical species exchange models. This reflects both the improvement in process understanding and the increasing availability of flux datasets, which are needed to parameterize models.

Yet there remain substantial challenges at all spatial scales (leaf to globe). The predictive capability of existing models at the field scale is often poor when tested against new flux measurement or at new sites, and a local re-parameterization is often necessary to describe observations satisfactorily (even accounting for potentially large errors in flux measurements, as shown by intercomparison exercises). Semi-empirical parameterization schemes that are developed on the basis of a literature review and many flux datasets (Massad et al. 2010b; Zhang et al. 2010; Segaard et al. 2002) should in principle, statistically, reproduce large-scale features of NH$_3$ exchange, as least within the multi-dimensional climate/vegetation/soil/management matrix, from which they derive. However, if their degree of empiricism is too large, they may prove unsuitable for generalisation to other conditions and for scenario simulations (e.g. climate change). On the other hand, the more mechanistic process-oriented models should in theory be applicable in all conditions, but they typically require more input data (some of which may not be available), are more difficult to parameterize (a greater number of parameters with no established reference), and are more computationally intensive (and thus less likely candidates for large-scale models).

The ideal surface/atmosphere NH$_3$ exchange model should treat all ecosystem NH$_x$-related processes, fluxes and pools dynamically (fertilizer volatilisation and recapture, soil biogeochemistry, plant biochemistry and physiology, air and surface chemistry, atmosphere exchange) within a multiple-layer canopy framework (in-canopy profiles of turbulence, radiation, temperature, humidity, green vs senescent leaves, soil layer). Such a coupling is possible and practicable at the field scale (e.g. coupled STAMP/CERES-EGC/VoIt’Air/SURFATM over crops), with a view to investigating certain aspects of the exchange, their dynamics and interactions, in parallel with detailed measurements of fluxes and pools. Clearly the task is more complex at the regional scale, although the CMAQ/EPIC example (Bash et al. 2013; Cooter et al. 2012; Fig. 8) demonstrates that it is feasible to a degree. The level of complexity of surface exchange schemes must be tailored to suit the modelling objectives, the scale and the availability of input data, while the availability of measurement data for validation assessment may prove a limiting factor in model development.

**Realistic NH$_3$ Exchange Frameworks for CTMs**

The current level of complexity of NH$_3$ surface exchange schemes in most regional and global CTMs is low relative to the advances that have been included in field scale models, i.e. static emissions from inventories and $R_c/V_d$ unidirectional deposition (with the exception of those few models mentioned in
Section “Ammonia Exchange in Chemical Transport Models (CTMs) at Regional Scales”), and clearly does not reflect the current level of process understanding. The following list highlights features that could realistically be implemented in $\chi_s/\chi_g/R_w$ two-layer schemes (Nemitz et al. 2001a; see Section “Canopy/Ecosystem Scale Models”; Fig. 7) within CTMs, at least at regional scales.

- Dynamic agricultural NH$_3$ emissions from field-applied manures and fertilizers. At present these emissions are typically prescribed from national or international inventories, and independent of meteorological conditions and crop development stage, but seasonal and diurnal distribution factors are applied. Dynamic emissions could be simulated using process-based models (Section “Process/Component Scale Models: Soil, Manure, Fertilizer, Leaf Litter, Leaf, Cuticle, Air Column Chemistry”), even if the treatment does not extend all the way to soil biogeochemistry, soil NH$_4^+$ pools and plant uptake.

- Soil/litter emission potential (outside fertilization events). This term is likely negligible in most temperate forests and semi-natural vegetation on acidic soils, but $\Gamma_g$ can be very large in grasslands and crops during the growing season, and might also be important in tropical forests due to large mineralisation rates and higher temperatures.

- Canopy re-capture of soil-based emissions. Emissions from fertilizers and other ground-based sources are partially re-captured by foliage (stomatal and non-stomatal pathways in a two-layer model, Fig. 7c). The degree of re-capture is controlled by canopy closure and leaf density (LAI profile), wind penetration, leaf wetness.

- Bi-directional stomatal exchange; N input-dependent $\Gamma_s$. The analysis by Massad et al. (2010b, their Fig. 5) shows consistent and convincing relationships between N inputs and $\Gamma_s$ for crops and grasslands, which could be implemented in CTMs. Because fertilization outweighs atmospheric deposition by a factor of 10 in such systems, the circularity issue (N inputs affect $\Gamma_s$, while $\Gamma_s$ controls NH$_3$ deposition) is less critical than in seminatural vegetation, though this represents a potentially important long-term negative feedback on deposition. Nevertheless, the relationship of $\Gamma_s$ to atmospheric N deposition remains rather uncertain.

- Photosynthesis-dependent stomatal resistance ($R_s$). The widely used multiplicative algorithm by Jarvis (1976), and other simplified empirical approaches (Wesely 1989), should be upgraded to a more mechanistic, photosynthesis-driven model (e.g. Ball et al. 1987), following the example of CTMs for O$_3$ (Anav et al. 2012).

- Pollution-climate dependent non-stomatal uptake ($R_w$). This feature is present in some CTMs via the (long-term) NH$_3$/SO$_2$ ratio, but likely most regional and especially global models do not account for the effects of surface chemical loadings on non-stomatal uptake rates. Accounting for NH$_3$ alone (Jones et al. 2007) is not sufficient away from the large agricultural point sources; rather, the ratio of Total Acids to NH$_3$ (Fig. 6; Massad et al. 2010b) should be used generically. Wind erosion of soil particles and leaf base cation leaching may raise leaf surface moisture pH significantly, but there are too few available data to account for this at present.
• Offline ecosystem and leaf surface chemistry modelling. Some CTM frameworks may not be able to accommodate coupled (online, interactive) ecosystem functioning together with the transport, chemistry and exchange calculations. However, soil/plant/ecosystem models (e.g. DNDC, STAMP, PaSim) could potentially be used offline to generate many values of $\Gamma_s$, $\Gamma_g$, $\Gamma_{\text{litter}}$, $\Gamma_{\text{soil}}$ in multiple simulations of ecosystems, seasons, soil and pollution climate conditions, representative of the region in which the CTM is applied. Such $\Gamma$ values should first be validated versus values published in the literature, and could then be called during CTM simulations from look-up tables or multiple regression functions. This might prove a viable compromise between constant default values (Zhang et al. 2010), or empirical functions (e.g. exponential decay with time, Massad et al. 2010b), and fully coupled CTM/ecosystem frameworks (Cooter et al. 2012). A similar concept could be applied for dynamic leaf surface chemistry (Flechard et al. 1999), whereby typical $T_d$ potentials could be simulated offline for a wide range of environmental conditions, and called up by the CTM in a $\chi_s/\chi_g/\chi_d/R_d$ scheme.

Further Needs for Flux Measurements, Model Input Data, and Validation Data

For regional and global representativeness, model development and parameterization rely heavily on new field-scale flux measurement datasets becoming available, but it is also clear that the availability of model input data and of spatially distributed validation data can be limiting factors for CTMs at regional and global scales. The most pressing data needs are summarised below.

• Flux measurements for under-represented ecosystems in temperate regions. The NH$_3$ flux literature is heavily dominated by grasslands, cereal crops, heathlands/moorlands and coniferous forests. There are few measurements over root crops, leguminous crops and legume-rich grasslands, deciduous forests, dry scrubland.

• Flux measurements in the tropics: data are needed for all ecosystem types including rain forests, savannah, tropical crops.

• Flux measurements near (<500 m) agricultural point sources in rural landscapes, together with a quantification of soil, apoplastic and epifoliar $\Gamma$ values as a function of distance from sources. Errors in measured fluxes arising from NH$_3$ advection must be accounted for (Loubet et al. 2009b).

• Seasonal and spatial variations in bulk leaf N content and apoplastic $\Gamma_s$ ratio for a range of ecosystems. Such measurements could be carried out at a large number of sites across a CTM modelling domain, without necessarily measuring NH$_3$ fluxes above ecosystems, and would be useful to explore temporal and spatial patterns of modelled NH$_3$ exchange and total N deposition.
- Measurements of $\Gamma$ values for the dominant crops, ecosystems and land uses in different climates and for different agricultural practices. These experimental $\Gamma$ estimates should be collected with a view to both (i) underpinning the development of empirical parameterizations for bi-directional models and (ii) validating process-based ecosystem model $\Gamma$ predictions. Long-term (e.g. annual, growing season) flux and $\Gamma$ datasets are needed to better represent background conditions, as campaign-based measurements over fertilised systems have traditionally tended to focus on emission events. Wherever possible, the determination of $\Gamma_s$ values should be attempted using different techniques (micrometeorological surface concentration extrapolation; controlled gas exchange chamber experiments; apoplastic extraction), as they tend to yield different results and the discrepancies between techniques are as yet poorly understood, given the current paucity of parallel measurements.

- Collection of critical ancillary data wherever NH$_3$ flux are measured in the field. In addition to classical (micro-) meteorological data, measured ancillary data must include variables that are likely to be useful later for model parameterization or validation. Efforts should be made to measure the following according to the issues being addressed: LAI and leaf density profile; leaf wetness profile; soil texture, porosity, wilting point, organic matter content, pH, [NH$_4^+$] and [NH$_3$]; slurry pH, TAN, dry matter content and application rate; bulk leaf N and NH$_4^+$ content; leaf litter pH and [NH$_4^+$]; leaf surface water (dew, rain) pH and [NH$_4^+$]. More difficult to measure, but equally important, would be apoplastic pH and [NH$_4^+$], such as by the vacuum infiltration technique (Husted and Schjoerring 1995); in-canopy vertical NH$_3$ profiles; ambient concentrations of SO$_2$, HNO$_3$, HNO$_2$ and HCl, and particulate NH$_4^+$ and NO$^-$. Studies quantifying base cation and other ion exchange with leaf surfaces are also needed.

- Fundamental analytical research is needed to provide guidance on the most appropriate soil NH$_4^+$ extraction method for the development of representative soil $\Gamma$ values. Many studies have demonstrated the variability of extracted/extractable NH$_4^+$ depending on the electrolyte used (e.g. KCl, CaCl$_2$) and its concentration in the extraction solution (see for example Fig. 1 in the Supplement on http://www.biogeosciences-discuss.net/10/C2954/2013/bgd-10-C2954-2013-supplement.pdf). Provided a better understanding of the relationships between extractable NH$_4^+$ and soil $\Gamma$, historical soil chemistry datasets from long-term ecological sites, agricultural experiment stations, soil surveys, etc., could be put to use within the context of soil/vegetation/atmosphere NH$_3$ modelling.

- Use of environmental microscopy (e.g. Burkhardt et al. 2012) as a powerful set of tools for improving our fundamental understanding of the chemical dynamics of leaf surface water during the transition from wet to dry conditions. Further testing and development of dynamic leaf surface chemistry models is currently hindered by the fact that the chemistry of microscale cuticular water layers present on leaves and needles during the day cannot be measured. In the absence of
suitable techniques for field measurements, such laboratory techniques should be encouraged.

- Development, testing, validation and deployment of low-cost instrumentation for long-term NH$_3$ flux estimates. Given the complexity and elevated costs associated with intensive and high-resolution NH$_3$ flux measurement campaigns, there have been endeavours to develop robust “low-cost, low-tech” methods for long-term flux estimates and parameterizations, such as the COTAG (COnditional Time-Averaged Gradient) system (Famulari et al. 2010). However, such systems have been successfully deployed at only a handful of sites to date, and further they lack consistent validation against established reference methods.

- Spatial fields of measured atmospheric NH$_3$ and NH$_4^+$ concentrations. Satellite-derived column NH$_3$ data offer much promise for CTM evaluation at regional and global scales, but there are still large uncertainties in the retrieved concentrations. Ground-based monitoring networks for both NH$_3$ and NH$_4^+$ by low-cost denuder/filter methods (Tang et al. 2009; Flechard et al. 2011; Adon et al. 2010) are available in only a handful of countries worldwide and should be encouraged, both for CTM evaluation and for ground truthing of satellite data. The vertical dimension of the concentration field in the atmospheric boundary layer should also be explored; aircraft measurements provide such information but are expensive; the extent to which low-cost measurement techniques could be deployed in profile configurations on tall towers should be investigated.

- Fine-resolution (~1 km$^2$) agricultural census data, and management practices. These model input data for CTMs are often only poorly known. The former are in many countries either classified information or not documented, and only available at much coarser resolution (>10 km $\times$ 10 km). Data on typical management practices with respect to manure and fertilizer application (timing, amounts, machinery) should be easier to obtain, but require extensive survey work.

- Development of methods for sub-grid assessments. The accuracy and evaluation of models close to sources is a source of uncertainty, since especially NH$_3$ deposition can occur at scales substantially smaller than the horizontal and vertical extent of CTMs (e.g. Section “Landscape Scale Models”, and Loubet et al. 2009a). Even where network data are available, the application and evaluation of CTMs for NH$_3$ concentrations is hindered by such local-scale gradients and variability (Wichink-Kruit et al. 2012). Use of plume or Lagrangian 1-D models close to the source (see Asman, 2001; Hertel et al. 2006, 2011) or coupling of sub-grid dispersion models to CTMs (e.g. Geels et al. 2012) should help bridge the gap between ground-based, single-point observations and spatially averaged CTM outputs, and could be used to help parameterize larger scale CTM models in future.

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