Observations of total RONO$_2$ over the boreal forest: NO$_x$ sinks and HNO$_3$ sources

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Abstract. In contrast with the textbook view of remote chemistry where HNO$_3$ formation is the primary sink of nitrogen oxides, recent theoretical analyses show that formation of RONO$_2$ ($\Sigma$ ANs) from isoprene and other terpene precursors is the primary net chemical loss of nitrogen oxides over the remote continents where the concentration of nitrogen oxides is low. This then increases the prominence of questions concerning the chemical lifetime and ultimate fate of $\Sigma$ ANs. We present observations of nitrogen oxides and organic molecules collected over the Canadian boreal forest during the summer which show that $\Sigma$ ANs account for $\sim$ 20% of total oxidized nitrogen and that their instantaneous production rate is larger than that of HNO$_3$. This confirms the primary role of reactions producing $\Sigma$ ANs as a control over the lifetime of NO$_x$ ($\text{NO}_x = \text{NO} + \text{NO}_2$) in remote, continental environments. However, HNO$_3$ is generally present in larger concentrations than $\Sigma$ ANs indicating that the atmospheric lifetime of $\Sigma$ ANs is shorter than the HNO$_3$ lifetime. We investigate a range of proposed loss mechanisms that would explain the inferred lifetime of $\Sigma$ ANs finding that in combination with deposition, two processes are consistent with the observations: (1) rapid ozonolysis of isoprene nitrates where at least $\sim$ 40% of the ozonolysis products release NO$_x$ from the carbon backbone and/or (2) hydrolysis of particulate organic nitrates with HNO$_3$ as a product. Implications of these ideas for our understanding of NO$_x$ and NO$_2$ budget in remote and rural locations are discussed.

1 Introduction

In remote, continental regions, isoprene, terpenes and other biogenic volatile organic compounds (BVOCs) rival CH$_4$ and CO as controls over the free radical chemistry of the atmospheric boundary layer, affecting global distributions of oxidants (OH, O$_3$, NO$_3$) and oxidant precursors (e.g. NO$_x$,...
HCHO) (e.g. Fuentes et al., 2000). In turn, these oxidants control the burden of tropospheric ozone and of both short (e.g. isoprene) and long-lived (e.g. CH$_4$, CH$_3$Br) organic compounds thus impacting climate. Consequently, the oxidative chemistry of BVOC has been the subject of extensive research. Recent advances in laboratory and field measurements have focused on the products of BVOC oxidation and have inspired renewed examination of how the mechanisms of BVOC oxidation affect atmospheric composition. In particular, the impact of BVOC on the HO$_x$ budget has been highlighted (e.g. Thornton et al., 2002; Leelieveld et al., 2008; Hofzumahaus et al., 2009; Stavrakou et al., 2010; Stone et al., 2011; Whalley et al., 2011; Mao et al., 2012; Paulot et al., 2012; Taraborrelli et al., 2012).

Oxidation of BVOC by OH results in peroxy radicals, which may react with NO$_x$ (NO$_x$=NO + NO$_2$), with other peroxy radicals (RO$_2$ or HO$_2$), or – in some cases – may isomerize (potentially regenerating OH). The reaction of peroxy radicals with NO$_2$ results in the formation of peroxy nitrates (RO$_2$NO$_2$) – a class of molecules which generally act as temporary reservoirs of NO$_x$ and serve to transport NO$_x$ on regional and global scales. Reaction of peroxy radicals with NO generally acts to propagate the ozone production cycle (R1a); however, a minor channel of the Reaction (R1b) which proceeds with the efficiency $\alpha$ (also known as the branching ratio), results in the formation of organic nitrates (RONO$_2$).

$$\text{RO}_2 + \text{NO} \rightarrow \text{RO} + \text{NO}_2 \quad (1-\alpha) \quad (\text{R1a})$$

$$\text{RO}_2 + \text{NO} \rightarrow \text{RONO}_2 \quad (\alpha) \quad (\text{R1b})$$

Calculations with box and chemical transport models (CTMs) have shown that organic nitrates play a significant role in the NO$_x$ and O$_3$ budgets (e.g. Trainer et al., 1991; Chen et al., 1998; Horowitz et al., 1998, 2007; Liang et al., 1998; von Kuhlmann et al., 2004; Fiore et al., 2005; Wu et al., 2007; Paulot et al., 2012). In Browne and Cohen (2012) we have shown that at NO$_x$ concentrations typical of remote and rural environments; the formation of $\Sigma$ANs is the dominant instantaneous NO$_x$ sink even at modest concentrations of BVOC. However, the net impact on O$_3$ and NO$_x$ depends on the extent to which $\Sigma$ANs act as a permanent versus temporary NO$_x$ sink, as has been shown in numerous models (e.g. von Kuhlmann et al., 2004; Fiore et al., 2005, 2011; Horowitz et al., 2007; Ito et al., 2009; Paulot et al., 2012). The lifetime and fate of $\Sigma$ANs remains one of the outstanding questions about their chemistry; compared to other aspects of the NO$_x$, HO$_x$, and VOC chemistry, there has been limited research on products of $\Sigma$ANs oxidation. Even for those nitrates whose oxidation products and yields have been measured, these measurements have occurred under conditions where the resulting peroxy radicals react primarily with NO and not with HO$_2$ or RO$_2$ (which are the expected reactions in the low NO$_x$ conditions of the boreal forest). As recently pointed out by Elrod and co-workers (Darer et al., 2011; Hu et al., 2011), $\Sigma$ANs may also be removed via hydrolysis in aerosol with an assumed product of NO$_3^-$. This uncertainty in the fate of $\Sigma$ANs results in large uncertainties in global ozone budgets. For instance, recent modeling studies have found that the ozone response to increasing isoprene emissions (as predicted in a warmer climate) is highly sensitive to the fate of isoprene nitrates (Ito et al., 2009; Weaver et al., 2009).

Here, we use observations, collected aboard the NASA DC-8 aircraft, of a suite of nitrogen oxides, organic molecules, and oxidants (OH and O$_3$) from the July 2008 NASA ARCTAS (Arctic Research of the Composition of the Troposphere from Aircraft and Satellites) campaign over the Canadian boreal forest, to examine the extent to which the organic nitrate products of BVOC oxidation control the lifetime of NO$_x$ in the remote continental boundary layer. We find that the production of $\Sigma$ANs is dominated by biogenic molecules and is generally larger than the production of HNO$_3$. Using the concentration measurements in conjunction with the production rates, our measurements also provide a constraint on the ratio of the $\Sigma$ANs lifetime to the HNO$_3$ lifetime over the boreal forest. We examine the loss processes of $\Sigma$ANs find that both deposition and chemical loss processes (including oxidation of isoprene nitrates and hydrolysis of $\Sigma$ANs in aerosol) are important. We find that the ozonolysis of isoprene nitrates is the largest gas-phase sink and we find that the particle phase hydrolysis of $\Sigma$ANs, which produces HNO$_3$, may be both an important loss process for $\Sigma$ANs and a significant source of HNO$_3$. The branching of $\Sigma$ANs loss between the processes that return NO$_x$ to the pool of available free radicals (e.g. oxidation) and those that remove NO$_x$ from the atmosphere (e.g. deposition, hydrolysis) has important consequences for regional and global NO$_x$, O$_3$, and OH.

2 ARCTAS measurements

The NASA ARCTAS experiment was designed to study processes influencing Arctic chemistry and climate and has been described in detail previously by Jacob et al. (2010). In this analysis we use measurements from the summer portion of the campaign over the Canadian boreal forest (June–July 2008). These measurements were made aboard the NASA DC-8 aircraft which contained instrumentation for an extensive suite of gas and aerosol measurements.

NO$_2$, total peroxy nitrates ($\Sigma$PNs), and total organic nitrates ($\Sigma$ANs) were measured aboard the DC-8 using thermal dissociation-laser induced fluorescence (TD-LIF). The instrument has been described in detail elsewhere (Day et al., 2002; Wooldridge et al., 2010) and the specific configuration used during ARCTAS has been described in Browne et al. (2011). Briefly, a two-cell TD-LIF with supersonic expansion (Thornton et al., 2000; Clevely et al., 2002; Day et al., 2002; Wooldridge et al., 2010) was deployed for ARCTAS. We use a 7 kHz, Q-switched, frequency doubled Nd:YAG
laser to pump a tunable dye laser (pyrromethene 597 in isopropanol) tuned to a 585 nm absorption in the NO₂ spectrum. We reject prompt scatter using time gated detection and eliminate scattered light at < 700 nm using bandpass filters. Fluorescence is imaged onto a red sensitive photomultiplier tube and counts are recorded at 4 Hz. The dye laser is tuned on and off an isolated rovibronic feature in the NO₂ spectrum, spending 9 s on the peak of the NO₂ absorbance and 3 s in an off-line position in the continuum of the NO₂ absorption. The difference between the two signals is directly proportional to the NO₂ concentration. We calibrate at least every two hours during a level flight leg using a 4.5 ppm NO₂ reference standard diluted to ∼ 2–8 ppbv in zero air.

The sample flow was split in thirds with one third directed to detection cell 1, where ambient NO₂ was continuously measured. The remaining flow was equally split between the measurement of total peroxy nitrates (ΣPNs) and total organic nitrates (ΣANs) which are detected by thermal conversion to NO₂ in heated quartz tubes. ΣPNs were converted to NO₂ at ∼ 200 °C and ΣANs at ∼ 375 °C, which is sufficient to dissociate ΣANs as well as any semivolatile aerosol phase organic nitrates (Rollins et al., 2010b). We do not detect non-volatile nitrates (i.e. NaNO₃). The resulting ΣO₂ of both heated channels (NO₂+ΣPNs or NO₂+ΣPNs+ΣANs) was measured in cell 2. The duty cycle of cell 2 was evenly split between the measurement of ΣPNs and of ΣANs and alternated between the two every 12 s or every 24 s. The 9 s average from each on-line block was reported to the data archive which is publically available at http://www-air.larc.nasa.gov/missions/arctas/arctas.html.

ΣPNs are calculated from the difference in signal between the ambient temperature and 200 °C channel and likewise, ΣANs are calculated from the difference in signal between the 375 °C (NO₂+ΣPNs+ΣANs) and the 200 °C (NO₂+ΣPNs). The detection limit (defined as signal to noise of 2 for the 9 s average) of the ΣANs signal is directly related to the magnitude of the NO₂ + ΣPNs (NP) signal and during ARCTAS was on average < 20 pptv for a 200 pptv NP signal. The ΣANs signal also requires interpolation of the NP signal which we calculate using a weighted sum of a linear interpolation of the NP signal (weight ~ 1/3) and an interpolation of the ratio of NP to NO₂ signal scaled to the measured NO₂. The uncertainty in the ΣANs measurement depends both on the magnitude and the variability of the NP signal. On average, the NP signal varied by less than 20 % on the timescale of the ΣANs measurements. An example time series of the ΣANs and ΣPNs data is shown in Fig. 1.

In the analysis below we use measurements only between 10 and 18 local solar time which enables us to neglect the possible interference from ClNO₂ (Thaler et al., 2011) since ClNO₂ is rapidly photolyzed during daylight hours.

In addition to the core measurement of ΣANs, described above, we use the measurements listed in Table 1 in our analysis. We use the merged data set from flights over the boreal forest of Canada that took place 29 June–13 July 2008 averaged to the 60 s time base (version 13).

3 ΣANs concentration and production

In the continental boundary layer over the boreal forest (between 50° and 67.5° N), we observed that ΣANs were 22 % (median) of NOy (Fig. 2) in background conditions which were sampled on flights 17, 19, 20, and 23. Periods of boundary layer sampling were determined by visually inspecting the potential temperature and ratio of potential temperature to IR surface temperature. The boundary layer heights determined by this method (∼ 1.5–2.4 km a.g.l.) are consistent with boundary layer heights measured over Northern Saskatchewan in July 2002 (Shashkov et al., 2007). The minimum altitude of sampling was just under 500 m. We see no evidence of a significant vertical gradient in the contribution of ΣANs to NOy, and thus believe the use of the median values to be appropriate. The background conditions were defined to exclude recent anthropogenic and biomass burning influences by only using conditions where CO was less than 180 ppbv and NOx was less than 200 pptv. Remaining biomass burning influences were removed by visually inspecting the HCN and CH₂CN concentration time series and excluding plumes. The mean concentrations of CO, CH₂CN, and HCN used in our analysis are lower than the means of the background ARCTAS measurements described in Simpson et al. (2011). In this analysis we define NOₓ as the sum of the measured individual components of NOₓ (NO, NO₂, ΣPNs, ΣANs, gas phase nitric acid, and submicron aerosol nitrate). The observation that ΣANs are on the order of 20 % of NOy is consistent with almost all past measurements of

![Fig. 1](http://www-atmos-chem-phys.net/13/4543/2013/)
Table 1. Species and measurement techniques used in this paper in addition to the core measurement of Σ ANs and NO2.

| Species                          | Method                                               | Reference                      |
|----------------------------------|------------------------------------------------------|--------------------------------|
| VOCs*                            | Whole air sampling                                   | Blake et al. (2003)            |
| VOCs*                            | Trace Organic Gas Analyzer (TOGA)                   | Apel et al. (2003)             |
| VOCs*                            | Proton transfer reaction mass spectrometry          | Wisthaler et al. (2002)        |
| CH4                              | Tunable diode laser absorption spectroscopy          | Sachse et al. (1987)           |
| NO, O3                           | Chemiluminescence                                   | Weinheimer et al. (1994)       |
| OH,HO2                           | Laser Induced Fluorescence Chemical ionization mass spectrometry | Falloona et al. (2004); Cantrell et al. (2003); Mauldin III et al. (2003) |
| HNO3                             | Chemical ionization mass spectrometry                | Crounse et al. (2006)          |
| Submicron aerosol nitrate, sulphate, ammonium, and organic aerosol | Aerosol mass spectrometry             | Cubison et al. (2011)          |

* A full list of the VOCs from each measurement technique can be found in Appendix A.

Fig. 2. NOy composition in the boundary layer over the remote boreal forest for background conditions (see text). HNO3 (particle) refers to submicron particulate NO3 as measured by the AMS and may include a contribution from particulate Σ ANs (see text).

Σ ANs from TD-LIF in continental locations (Day et al., 2003; Rosen et al., 2004; Cleary et al., 2005; Perring et al., 2009, 2010; Farmer et al., 2011); however, in this data set we find that the instantaneous production rate of Σ ANs is larger than the HNO3 production rate – a situation that has not been reported previously.

Using the measured VOCs, OH, HO2, and NO concentrations (Table 1), we calculate the instantaneous production rate of Σ ANs (P(Σ ANs) Eq. 1) via OH oxidation of VOCs by assuming that peroxy radicals are in steady-state (Eqs. 2–3) which results in Eq. (4):

\[ P(\Sigma \text{ANs}) = \alpha_i \sum_j k_{\text{RO}_2, \text{NO}\text{[RO}_2\text{]}\text{[NO]}, \quad (1) \]

\[ \frac{d[\text{RO}_2]}{dt} = k_{\text{OH}+\text{VOC}_i}[\text{OH}][\text{VOC}_i] - k_{\text{RO}_2, \text{NO}[\text{RO}_2\text{]}\text{[NO]} \]

\[ - k_{\text{RO}_2, \text{HO}_2[\text{RO}_2\text{]}\text{[HO}_2]} \]

\[ - \sum_j k_{\text{RO}_2, \text{RO}_2j[\text{RO}_2\text{]}j - k_{\text{isom}}[\text{RO}_2]} \approx 0, \quad (2) \]

\[ [\text{RO}_2] \approx \frac{k_{\text{OH}+\text{VOC}_i}[\text{OH}][\text{VOC}_i]}{k_{\text{RO}_2, \text{NO}[\text{RO}_2\text{]}\text{[NO]} + k_{\text{RO}_2, \text{HO}_2[\text{RO}_2\text{]}\text{[HO}_2]} + \sum_j k_{\text{RO}_2, \text{RO}_2j[\text{RO}_2\text{]}j + k_{\text{isom}}}}, \quad (3) \]

\[ P(\Sigma \text{ANs}) \approx \sum_i \gamma_i \alpha_i k_{\text{OH}+\text{VOC}_i}[\text{OH}][\text{VOC}_i], \quad (4) \]

where

\[ \gamma_i = \frac{k_{\text{RO}_2, \text{NO}[\text{NO}]} + k_{\text{RO}_2, \text{HO}_2[\text{HO}_2]} + \sum_j k_{\text{RO}_2, \text{RO}_2j[\text{RO}_2\text{]}j + k_{\text{isom}}}}{k_{\text{RO}_2, \text{NO}[\text{NO}]} + k_{\text{RO}_2, \text{HO}_2[\text{HO}_2]} + \sum_j k_{\text{RO}_2, \text{RO}_2j[\text{RO}_2\text{]}j + k_{\text{isom}}}}, \quad (5) \]

Here, \( k_{\text{isom}} \) refers to the rate of a unimolecular isomerization reaction of RO2. This class of reactions has recently been shown to be important when the lifetime of RO2 is long, such as in low NOx conditions (e.g. Peeters et al., 2009; Peeters and Müller, 2010; Crounse et al., 2011). \( \gamma \) (Eq. 5) represents the fraction of RO2 that reacts with NO and depends on the identity of the RO2. We calculate specific \( \gamma \) values for peroxy...
radicals derived from monoterpenes (\(\alpha\)- and \(\beta\)-pinene), isoprene, methacrolein, and methyl vinyl ketone. All other peroxy radicals (which, as shown below, account for only 3\% of the \(\Sigma ANs\) production) are assumed to behave like methyl vinyl ketone peroxy radicals. Each of these \(\gamma\) values are calculated using the \(RO_2 + HO_2\) rate calculated from the parameterization used in the Master Chemical Mechanism (MCM) v3.2 (Jenkin et al., 1997; Saunders et al., 2003) available at http://mcm.leeds.ac.uk/MCM. We use measured isomerization rates for isoprene peroxy radicals (Crounse et al., 2011) and methacrolein peroxy radicals (Crounse et al., 2012). Although there are theoretical predictions that peroxy radicals derived from monoterpenes undergo a fast ring closure reaction followed by addition of \(O_2\), regenerating a peroxy radical (Vereecken and Peeters, 2012), there are no experimental measurements of \(\gamma\) values. We assume that the organic nitrate yield is the same as the parent and thus implicitly assume the isomerization reaction of monoterpenes-derived \(RO_2\) is unimportant for our calculation. We also assume that the isomerization reaction is negligible for the remaining \(RO_2\) species. All \(\gamma\) values use the same rate coefficients for \(RO_2 + NO\) (from MCM v3.2) and for \(RO_2 + RO_2\) (the IUPAC CH\(_3\)O\(_2\) + C\(_2\)H\(_5\)O\(_2\) reaction rate available at http://www.iupac-kinetic.ch.cam.ac.uk/; Atkinson et al., 2006). In these calculations, when the measured NO is less than 0 pptv, we assign it a value of 1 pptv. Due to the more complete data coverage, we use the LIF measurements of OH and HO\(_2\); however, the LIF and CIMS data agree well (Ren et al., 2012) and we see no significant difference when using the CIMS data (Appendix A2). Details regarding the VOCs, OH oxidation rates, branching ratios, and uncertainties regarding rate coefficients are described in Appendix A.

The instantaneous production of HNO\(_3\) is calculated using the measured OH and NO\(_2\). We use the rate constant from Mollner et al. (2010) with the temperature dependence from Henderson et al. (2012).

The total calculated \(P(\Sigma ANs)\), shown in Fig. 3, is similar to or greater than the calculated nitric acid production. Biogenic species account for the majority (97\%) of \(P(\Sigma ANs)\) (Fig. 4) with isoprene (64\%), methyl vinyl ketone (9\%), and \(\alpha\) and \(\beta\)-pinene (25\%) contributing the most production. Due to the rapid isomerization of the methacrolein peroxy radical, very few methacrolein nitrates are formed (< 1\% of \(P(\Sigma ANs)\)). The conclusion that the \(P(\Sigma ANs)\) rate is faster than the \(P(HNO_3)\) rate holds for both the isoprene nitrate branching ratio of 11.7\% from Pauly et al. (2009) as shown in Fig. 3 and the lower value of 7\% from Lockwood et al. (2010) (details in Appendix A). The time series of \(\Sigma ANs\) and isoprene shown in Fig. 1 illustrates how increases in \(\Sigma ANs\) roughly correspond to increases in the precursors (e.g. isoprene).

Since only \(\alpha\) and \(\beta\)-pinene were measured aboard the DC-8 aircraft, it is likely that the concentration of monoterpenes is underestimated. Enclosure measurements of black spruce trees (an important constituent of the Canadian boreal forest) indicate that emissions of camphene and 3-carnene are larger than those of \(\alpha\)-pinene (Fulton et al., 1998) and extensive measurements of VOCs in the boreal forest of Finland have shown that \(\alpha\) and \(\beta\)-pinene represent only a fraction of the monoterpenes (e.g. Räisänen et al., 2009; Hakola et al., 2012). Vertical profile measurements from the surface to \(\sim 800\) m in the boreal forest of Finland also indicate steep vertical gradients in monoterpenes and isoprene (Spirig et al., 2004), indicating that production of \(\Sigma ANs\) is likely much faster at altitudes lower than those sampled by the DC-8 aircraft (minimum of \(\sim 500\) m). Since the composition of monoterpenes is dependent on the ecosystem, we do not attempt to scale the monoterpenes measurement. Rather, we note that if the monoterpenes concentration is doubled, the monoterpenes contribution to \(\Sigma ANs\) production increases to 39\% reducing the isoprene contribution to 51\%. The median of the ratio of \(P(\Sigma ANs)\) to \(P(HNO_3)\) also increases from 1.96 to 2.6.

Despite this larger production rate of \(\Sigma ANs\) than of HNO\(_3\), the median concentration of \(\Sigma ANs\) (108 pptv) is less than the median concentration of the sum of gas phase HNO\(_3\) and particulate NO\(_3\) (180 pptv). One possible explanation of this apparent discrepancy is that entrainment may have a significant effect on the concentrations. The observed concentration differences during flight segments where the DC-8 crossed the boundary layer indicate that entrainment will...
dilute both HNO₃ and Σ ANs. Σ ANs have a slightly faster dilution: the median concentration difference above and within the boundary layer is $1.0 \times 10^9$ molecules cm$^{-3}$ for Σ ANs and $7.0 \times 10^8$ molecules cm$^{-3}$ for HNO₃ (gas+particle). As an upper limit estimate we assume that the average boundary layer height is 1.5 km and is growing at 10 cm s$^{-1}$. Even with this dilution correction the production rate of Σ ANs is greater than that of HNO₃ in 50% of the boundary layer data. In contrast, in 90% of the data the concentration of Σ ANs is less than that of HNO₃ (gas+particle). Since we use an upper limit estimate of the effect of entrainment and considering the production rate of Σ ANs is likely larger than calculated here due to the presence of unmeasured BVOCs (particularly within the forest canopy), we conclude that factors other than entrainment are responsible for the production rate-concentration discrepancy between HNO₃ and Σ ANs.

It is also possible that the particulate phase NO$_3^-$ as measured by the aerosol mass spectrometer (AMS) includes a contribution from particle phase Σ ANs (e.g. Farmer et al., 2010; Rollins et al., 2010a). For 77% of the one minute data for which there are both gas phase HNO₃ and Σ ANs measurements, the concentration of Σ ANs is less than the concentration of gas phase HNO₃. Therefore, the possible contribution from Σ ANs to the AMS NO$_3^-$ signal does not affect our conclusions that HNO₃ is generally present in higher concentrations than Σ ANs. In the remainder of the manuscript HNO₃ will refer to the sum of gas phase HNO₃ and particulate NO$_3^-$ unless stated otherwise.

We conclude that the larger production rate yet smaller concentration of Σ ANs than of HNO₃ implies a shorter lifetime of Σ ANs than of HNO₃. We note that the lifetime of Σ ANs represents the loss of the nitrate functionality and thus will be longer than the average lifetime of individual nitrates because oxidation of some nitrates results in products that are more highly functionalized ANs.

4 Lifetime of HNO₃

The lifetime of HNO₃ in the boundary layer is primarily determined by deposition that, for gas phase HNO₃, is generally assumed to occur with unit efficiency at a mass transfer rate set by turbulence. Assuming an approximate boundary layer height of $\sim 2$ km (we observed boundary layer heights that ranged from 1.5 km to 2.6 km) and a deposition velocity of $4$ cm s$^{-1}$, we calculate a lifetime of $\sim 14$ h (loss rate of $2 \times 10^{-3}$ s$^{-1}$) for HNO₃ for midday conditions. The deposition velocity of HNO₃ over forests has been reported to range from $2$ cm s$^{-1}$ to $10$ cm s$^{-1}$ (Horii et al., 2005 and references therein), with a strong variation associated with time of day and season. Given the uncertainty and time of day dependence also associated with the boundary layer height, we use this lifetime as a guide for thinking about the daytime lifetime of Σ ANs, which our measurements indicate is shorter than that of HNO₃, and do not focus on the exact number. The depositional loss of aerosol phase NO$_3^-$ is generally on the order of days, however, due to its low contribution to total HNO₃ (Fig. 2), we consider only the gas-phase loss. Other losses, photolysis and oxidation by OH, are quite slow with median lifetimes of several weeks.

5 Lifetime of Σ ANs

Using the ARCTAS data we are unable to constrain the exact Σ ANs lifetime since to do so would require knowledge of the photochemical age of the air mass, the history of Σ ANs production (which is likely to have significant vertical gradients), and the exact chemical speciation of the Σ ANs. However, with the constraint imposed by the HNO₃ data and with some reasonable assumptions we can identify the most likely Σ ANs loss processes.

5.1 Deposition

Deposition is likely a significant term in the Σ ANs budget, however, the deposition velocity of Σ ANs will be less than that of HNO₃. The measured Henry’s law coefficients of some of the more soluble individual hydroxy nitrates ($\sim 10^3$–$10^5$ Matm$^{-1}$, Shepson et al., 1996; Treves et al., 2000) are orders of magnitude lower than that of HNO₃ (1 $\times$ 10$^{14}$ Matm$^{-1}$ at pH $\sim 6.5$, Seinfeld and Pandis, 2006). Still, these measured Henry’s law coefficients of hydroxy nitrates indicate that wet deposition is a significant loss process and a recent study indicates that foliar uptake of organic nitrates is possible (Lockwood et al., 2008). The only direct simultaneous measurements of Σ ANs and HNO$_3$ deposition are those of Farmer and Cohen (2008) who estimated a Σ ANs dry deposition velocity of 2.7 cm s$^{-1}$ compared to
3.4 cm$^{-1}$ for HNO$_3$ above a ponderosa pine forest. Similar results have been obtained more recently at the same forest (K.-E. Min, personal communication, 2012). Although the exact magnitude of the depositional loss likely depends on the specific composition of $\Sigma$ANs, as well as the partitioning between gas and aerosol, we assume that a similar result exists for the boreal forest since recent measurements of speciated organic nitrates using chemical ionization mass spectrometry at the ponderosa pine forest (Beaver et al., 2012) indicate a similar composition of $\Sigma$ANs as assumed here from the instantaneous production rate. Therefore, although the deposition of $\Sigma$ANs is important, it is slower than the deposition of HNO$_3$, thus implying the existence of other sinks of $\Sigma$ANs. In other words, chemistry must be an important sink of $\Sigma$ANs.

5.2 Photolysis

The OH oxidation of both isoprene and monoterpenes produces hydroxy nitrates as first generation products. These molecules account for at least 89 % of the instantaneous production rate of $\Sigma$ANs (Fig. 4) for the conditions considered here. Although there are no direct measurements of the photolysis rates of these specific molecules, by analogy to other compounds we estimate that photolysis is a negligible sink for them. Roberts and Fajer (1989) report that the cross section of nitrooxy ethanol is approximately a factor of three smaller than methyl nitrate. Similarly, photolysis rates of alkyl nitrates are on the order of several days (e.g. Roberts and Fajer, 1989; Talukdar et al., 1997) and are thus too slow to be important. In contrast, $\alpha$-nitrooxy ketones have been shown to have a cross section approximately five times larger than alkyl nitrates (Roberts and Fajer, 1989; Barnes et al., 1993). Our calculations suggest these are too small a fraction of the total to affect the overall lifetime. To estimate an upper limit, we use the fastest reported photolysis rate from Suarez-Bertoa et al. (2012), which is for 3-methyl-3-nitrooxy-2-butanol. This rate was calculated assuming solar conditions appropriate for 1 July at noon on 40 °N. To achieve a rate appropriate for the ARCTAS conditions we use the median rates of methyl and ethyl nitrate photolysis measured during ARCTAS and scale these to the rate of 3-methyl-3-nitrooxy-2-butanol using the measurements of Roberts and Fajer (1989) and Suarez-Bertoa et al. (2012). We take the average of the rate calculated from methyl nitrate and from ethyl nitrate and assume that 9 % of the nitrates (the methyl vinyl ketone contribution in Fig. 4) are $\alpha$-nitrooxy ketones. This results in an overall photolysis rate for $\Sigma$ANs of 2.5 × 10$^{-6}$ s$^{-1}$ (lifetime of ~110 h), a rate that even when combined with deposition is too slow to account for the inferred $\Sigma$ANs loss.

5.3 Oxidation

The overall gas-phase chemical removal rate of $\Sigma$ANs can be represented as

$$k_{\text{ox-loss}} = \sum k_{\text{AN}_{i}+\text{OX}} \left[\text{AN}_{i}\right] / \left[\Sigma\text{ANs}\right] \chi_{i},$$

(6)

where $k_{\text{AN}_{i}+\text{OX}}$ the rate constant of that oxidant with the specific nitrate, $\left[\text{OX}\right]$ represents the concentration of oxidant (OH, O$_3$, or NO$_3$), $\left[\text{AN}_{i}\right]$ represents the concentration of a specific nitrate, and $\chi_{i}$ the fraction of the reaction that results in loss of the nitrate functionality (referred to as NO$_3$ recycling). To simplify our calculation, we neglect the possibility that the oxidation of nitrates results in the formation of dinitrates which would result in a small positive term in Eq. (6). We also ignore oxidation by NO$_3$ since we only use daytime measurements above the forest canopy.

We estimate the composition of $\Sigma$ANs as a mixture of the small, long-lived, alkyl nitrates measured in the whole air samples (which account for a median of 30 % of the $\Sigma$ANs measured by TD-LIF) and molecules that can be estimated from the instantaneous production rate of $\Sigma$ANs (Fig. 4). The small nitrates have very long lifetimes and are a negligible term in the overall loss rate. We use the OH oxidation rates of isoprene-derived nitrates (assuming 60 % $\delta$-hydroxy isoprene nitrates and 40 % $\beta$-hydroxy isoprene nitrates) and methyl vinyl ketone-derived nitrates from Paulot et al. (2009). Recently, Lockwood et al. (2010) have measured the ozone oxidation rate of three of the eight possible isoprene nitrate isomers. The three isomers include one $\delta$-hydroxy isomer and two $\beta$-hydroxy isomers. We assume that the $\delta$-hydroxy isoprene nitrate rate constant from Lockwood et al. (2010) is representative of all $\delta$-hydroxy isomers. The rate constants for two $\beta$-hydroxy isomers differ by approximately a factor of three and we bound the possible range of reaction rates using these two rates. This results in an ozone loss rate ranging from 7.4 × 10$^{-17}$ cm$^3$ molecules$^{-1}$ s$^{-1}$ to 1.7 × 10$^{-16}$ cm$^3$ molecules$^{-1}$ s$^{-1}$. Results using the branching ratio between the $\delta$ and $\beta$-hydroxy nitrate channels as determined by Lockwood et al. (2010) (and updated by Pratt et al., 2012) are included in Appendix B.

We are unaware of any experimental constraints on the oxidation rate of monoterpene nitrates by OH and we estimate an OH oxidation rate constant of 4.8 × 10$^{-12}$ cm$^3$ molecules$^{-1}$ s$^{-1}$ based on a weighting of the MCM v3.2 rates for $\alpha$-pinene and $\beta$-pinene nitrates as described in Browne et al. (2013). The monoterpene nitrates in our calculations are based on the production from the observed concentrations of $\alpha$- and $\beta$-pinene, the only two monoterpenes measured aboard the aircraft. These nitrates will predominantly be saturated molecules and thus ozonolysis of these nitrates should be too slow to be important. As discussed in Sect. 3, it is likely that the contribution of monoterpene nitrate is underestimated. It is therefore possible that some of the monoterpene-derived nitrates may be
unsaturated molecules. We discuss the impact of this possibility in Appendix B and conclude that since the release of NO\textsubscript{2} from these molecules following oxidation is likely low, the effect on the oxidation rate is minimal.

The NO\textsubscript{x} recycling (\(\chi\)) following OH oxidation depends on the fate of the resulting nitroxy peroxy radical (R(NO\textsubscript{3})O\textsubscript{2}) which may react with NO, HO\textsubscript{2}, or other RO\textsubscript{2}. We assume that reactions with HO\textsubscript{2} generate a more highly functionalized nitrate and that the NO\textsubscript{x} recycling (the loss of the nitrate functionality) occurs with the same efficiency through both the R(NO\textsubscript{3})O\textsubscript{2} + NO and R(NO\textsubscript{3})O\textsubscript{2} + RO\textsubscript{2} channels. We use the same assumptions for the R(NO\textsubscript{3})O\textsubscript{2} + HO\textsubscript{2} rate as in the calculation of \(\gamma\) in Sect. 3, however, we assume that no isomerization reactions occur. We find that RO\textsubscript{2} + RO\textsubscript{2} reactions account for at most 1% of the RO\textsubscript{2} reactions. Uncertainties regarding these estimations are discussed in Appendix B. NO\textsubscript{x} recycling from the RO\textsubscript{2} + NO reaction have been constrained by laboratory experiments to be \(\sim 55\%\) for isoprene nitrates and 100% for MVK nitrates (Pautot et al., 2009). We are unaware of any measurements of NO\textsubscript{x} recycling from monoterpane nitrates and assume a value of 100% as an upper limit. Although the molecular structure of monoterpane nitrates implies that the NO\textsubscript{x} recycling is likely much lower than 100%, the contribution (as calculated below) from monoterpane nitrates to NO\textsubscript{x} recycling is negligible making a more accurate estimate unnecessary.

NO\textsubscript{x} recycling following ozonolysis of unsaturated nitrates (isoprene nitrates) depends on the initial branching of the ozonide to the two possible pairs of a carbonyl molecule and an energy-rich Criegee biradical and the subsequent fate of the Criegee biradical (stabilization or decomposition). To our knowledge, no experimental constraints on this process exist for any unsaturated organic nitrate. The MCM v3.2 assumes equal branching between the two possible carbonyl/Criegee biradical pairs; we calculate NO\textsubscript{x} recycling (40%) using the MCM v3.2 products of the ozonolysis of isoprene nitrates, the assumption that a stabilized Criegee biradical reacts only with water, and the relative abundances of the different isoprene nitrate isomers from Pautot et al. (2009) (ignoring the minor 3,4 and 2,1 isomers). Using the relative abundances of the different isoprene nitrate isomers from Lockwood et al. (2010) (updated with the numbers from Pratt et al., 2012) results in a NO\textsubscript{x} recycling of 38%.

Our calculation of the \(\Sigma ANs\) loss rate can be summarized by expanding Eq. (6) to

\[
k_{\text{ox-loss}} = \sum_{i} k_{\text{AN}i + \text{OH}[\text{OH}]} \frac{[\text{AN}i]}{[\Sigma \text{AN}s]} \beta_i (1 - F_{\text{RO2+HO2}}) + \sum_{i} k_{\text{AN}i + \text{O3}[\text{O3}]} \frac{[\text{AN}i]}{[\Sigma \text{AN}s]} \delta_i,
\]

Here, \(\beta\) represents the fraction of NO\textsubscript{x} recycled following the reaction of the peroxy radical with RO\textsubscript{2} or NO, \(F_{\text{RO2+HO2}}\) (Eq. 8) represents the fraction of the time that the peroxy radical reacts with HO\textsubscript{2} (and thus does not recycle NO\textsubscript{x}), and \(\delta\) represents the NO\textsubscript{x} recycling from ozonolysis. Uncertainties regarding this calculation are described in Appendix B.

Using the assumptions above, we calculate a chemical \(\Sigma ANs\) lifetime of \(\sim 9\text{–}18\text{ h}\) (Table 2) which ranges from slightly shorter to slightly longer than our estimated HNO\textsubscript{3} lifetime (\(\sim 14\text{ h}\)). In combination with deposition (\(\sim 17.5\text{ h}\) for a 2 km boundary layer), a detailed representation of oxidative \(\Sigma ANs\) loss results in a calculated \(\Sigma ANs\) lifetime in the range of the assumed lifetime of HNO\textsubscript{3}. In these calculations, the majority of \(\Sigma ANs\) loss occurs via isoprene nitrate ozonolysis, which has recently been reported to be much faster than previously assumed (Lockwood et al., 2010). Additional measurements of this rate and the products are important to constraining our understanding of \(\Sigma ANs\) and their role in the NO\textsubscript{x} budget.
5.4 Hydrolysis of particulate organic nitrates

5.4.1 Loss of ΣANs

Although we calculate a ΣANs loss rate due to oxidation and deposition that is similar to the assumed loss rate of HNO₃, the recent suggestion that organic nitrates may undergo hydrolysis in aerosols to produce HNO₃ as a product (Sato, 2008; Darer et al., 2011; Hu et al., 2011) is also a viable hypothesis to explain the measurements. Evidence for organic nitrate losses in ambient (Day et al., 2010) and chamber generated particles (Liu et al., 2012) analyzed with IR spectroscopy is consistent with this mechanism. This chemistry results in the depletion of ΣANs and an enhancement in HNO₃; both effects would contribute to the ratio of ΣANs to HNO₃ production and concentration that we report here.

Bulk solution studies of hydrolysis of organic nitrates indicate that primary and secondary nitrates are stable at atmospherically relevant pH, but that the lifetime of tertiary hydroxy organic nitrates is surprisingly short (0.019–0.67 h), even in neutral solutions (Darer et al., 2011; Hu et al., 2011). Since these are bulk solution studies, there are some difficulties associated with extending the rates to aerosol processes. Namely, the question arises as to whether the nitrates are present in the organic or aqueous phase of the aerosol and if the availability of liquid water is sufficient for the reaction. Some of these issues have been recently discussed by Liu et al. (2012) who, using a smog chamber without seed aerosol, constrained the hydrolysis of particulate organic nitrates derived from the photo oxidation of 1,2,4-trimethylbenzene. Using their measurements of the organic aerosol composition, they calculated a lifetime of ~6 h for particulate organic nitrates when the relative humidity was greater than 20%.

Since the vapor pressures of first generation isoprene nitrates are generally too high to partition into aerosol (Rollins et al., 2009), we begin the estimation of the hydrolysis rate by assuming that only monoterpene nitrates are present in organic aerosol. Although Henry’s law coefficients of small (≤5 carbons) hydroxy nitrates have been measured to be quite large, approximately ~10³–10⁵Matm⁻¹ (Shepson et al., 1996; Treves et al., 2000), it is reasonable to assume that as a ten carbon compound, a monoterpene nitrate may have a lower Henry’s law coefficient. We therefore assume that these nitrates partition only into organic aerosol and that the organic aerosol contains sufficient liquid water for this reaction to occur (median RH of 63 % and minimum of 34 %).

We use absorptive partitioning theory to determine the fraction of the monoterpene nitrate in the particle phase (Pankow, 1994; Donahue et al., 2006):

\[ C_i^* = \frac{C_i^g \cdot C_{OA}}{C_i^a} = \frac{MW_i \cdot 10^6 \cdot \zeta_i \cdot p_i}{760 \cdot R \cdot T}. \]  

(9)

Here \( C_i^* \) represents the effective saturation concentration (µg m⁻³) of the organic nitrate, \( C_i^g \) is the concentration of the organic nitrate in the condensed phase (µg m⁻³), \( C_i^a \) the concentration of the organic nitrate in the gas phase (µg m⁻³), and \( C_{OA} \) is the concentration of organic aerosol (µg m⁻³). In the second equality \( R \) is the universal gas constant (8.206 × 10⁻⁵atm m³ K⁻¹ mol⁻¹), \( MW_i \) is the molecular weight of the organic nitrate (assumed here to be a hydroxy monoterpene nitrate – 215 g mol⁻¹), \( \zeta_i \) is the molality based activity coefficient (assumed to be 1), \( p_i \) is the vapor pressure of the organic nitrate (Torr), and 760 and 10⁶ are unit conversion factors. We calculate an estimated bound on the partitioning of monoterpene nitrates to the aerosol using vapor pressures of \( 4 \times 10^{-5} \)Torr (\( C_i^g = \) of 48 µg m⁻³ at 286 K – the median temperature during ARCTAS) derived from chamber measurements of nitrate products of the NO₂ + β-pinene reaction (Fry et al., 2009) and of \( 5.8 \times 10^{-2} \)Torr (\( C_i^g = 7 \) µg m⁻³) from chamber measurements of the NO₂ + limonene reaction (Fry et al., 2011). The organic aerosol loading is from the AMS measurement and can be subdivide into two distinct regimes: one with a median loading of ~1 µg m⁻³ (at ambient temperature and pressure) and one with a median loading of ~6.6 µg m⁻³. The enhanced loading regime (60% of the data) was associated with higher concentrations of acetone, a known oxidation product of monoterpens, suggesting that monoterpens are an important source of SOA. This is consistent with measurements in southern Ontario reporting high concentrations of biogenic SOA (Slowik et al., 2010). The concentration of the biogenic species (β-pinene, α-pinene, isoprene, MVK, and MACR) were all higher in the regime of enhanced organic aerosol loading than in the lower loading regime. The isoprene oxidation products showed higher enhancements (e.g. 181 % equivalent to 278 ppv for MVK) than did isoprene (18%–53 ppv). The concentration enhancement of acetone (117%–1.23 ppbv) was also larger than that of the monoterpenes (105%–122 pptv), however, the long lifetime and multiple sources of acetone make a direct attribution to monoterpene oxidation impossible. Nevertheless, it is clear that the enhanced loading regime represents a larger biogenic influence and is more aged than the lower loading regime.

The fraction of the monoterpene nitrate in the aerosol (\( F_{aero} \)) is calculated using Eq. (10).

\[ F_{aero} = \frac{C_i^g}{C_i^g + C_i^a} = \left(1 + \frac{C_i^g}{C_{OA}}\right)^{-1}. \]  

(10)

We calculate the loss rate of ΣANs through hydrolysis (\( k_{hyd-loss} \)) using Eq. (11):

\[ k_{hyd-loss} = \sum_i k_{hyd} F_{aero,i} F_{t_{tertiary,i}} \frac{[AN_i]}{[ΣANs]}. \]  

(11)

where \( F_{t_{tertiary}} \) represents the fraction that is tertiary nitrate and \( k_{hyd} \) represents the hydrolysis rate constant. We set \( F_{t_{tertiary}} \) at 75 %, midway between the 63 % for α-pinene nitrates and 92 % for β-pinene nitrates from MCM v3.2. We
Table 3. Median calculated loss rate of ΣANs due to hydrolysis in the particle phase assuming that only monoterpene nitrates may partition into the aerosol and hydrolyze. We consider cases that span different vapor pressures, hydrolysis rates, and organic aerosol loadings. Here, $C^*$ represents the effective saturation concentration, $\tau_{\text{hyd}}$ is the lifetime to hydrolysis for a tertiary nitrate in the particle phase, $k_{\text{hyd-loss}}$ is the calculated loss rate of ΣANs via hydrolysis (see text for details), and the last column is the median of the ratio of this HNO$_3$ source to the source from the reaction of OH with NO$_2$. After correction for the small alkyl nitrates, monoterpene nitrates accounted for $\sim 10\%$ ($\sim 19\%$) (median value) of the ΣANs in the low (high) aerosol loading periods. We assume that only a fraction ($75\%$) of the monoterpene nitrates undergo hydrolysis and thus the fraction of the ΣANs that are in the particle phase and undergoing hydrolysis is 2–7% for the high loadings and < 1% for the low loadings.

| Organic aerosol loading (µg$^{-3}$) | $C^*$ (µg$^{-3}$) | $\tau_{\text{hyd}}$ (h) | $k_{\text{hyd-loss}}$ (s$^{-1}$) | $k_{\text{hyd-loss}} [\Sigma \text{ANs}] / ([\text{OH+NO}_3][\text{OH}][\text{NO}_2])$ |
|------------------------------------|-----------------|-----------------|-----------------|--------------------------------------------------|
| 1                                 | 7               | 6               | $3.8 \times 10^{-7}$ | 0.03                                             |
| 6.6                               | 7               | 6               | $3.2 \times 10^{-6}$ | 0.53                                             |
| 1                                 | 48              | 6               | $6.1 \times 10^{-8}$ | 0.00                                             |
| 6.6                               | 48              | 6               | $8.0 \times 10^{-7}$ | 0.13                                             |
| 1                                 | 48              | 0.67            | $5.5 \times 10^{-7}$ | 0.04                                             |
| 6.6                               | 48              | 0.67            | $7.2 \times 10^{-6}$ | 1.16                                             |

note that the fraction of ΣANs predicted to be derived from monoterpene based on the instantaneous production rate changes insignificantly between the low and enhanced loadings and we use the value from Fig. 4. However, in the low loading regime the small alkyl nitrates represent a larger fraction of ΣANs (61%) than in the enhanced loading regime (23%). Thus, the absolute fraction of ΣANs from monoterpene nitrates is higher in the enhanced loading regime.

Based on the work by Elrod and co-workers (Darer et al., 2011; Hu et al., 2011) showing an order of magnitude variation in the tertiary nitrate hydrolysis lifetime, it appears that the identity of the organic nitrate influences the hydrolysis rate. Although these bulk solution rates may not be strictly applicable to aerosol processes, it is also likely that the lifetime reported by Liu et al. (2012) for 1,2,4-trimethylbenzene-derived organic nitrates may not apply to biogenic systems. Therefore, we calculate the overall ΣANs hydrolysis rate ($k_{\text{hyd-loss}}$) for three different combinations of hydrolysis rates ($k_{\text{hyd}}$) and $C^*$ values as shown in Table 3.

In the enhanced loading regime these rates range from 2% to 20% of the oxidative lifetime (assuming the faster ozonolysis rate). It should be emphasized that the hydrolysis loss rate calculated here is reflective of the hydrolysis loss rate averaged over all the individual organic nitrates; in other words, the loss rate of an individual nitrate might be faster or slower than this rate. In fact, the rate calculated here is the result of only $\sim 2\%$ ($C^* = 48\mu g\text{ m}^{-3}$) or $\sim 7\%$ ($C^* = 7\mu g\text{ m}^{-3}$) of the ΣANs undergoing hydrolysis in the enhanced loading regime and for < 1% of the ΣANs (regardless of $C^*$ value) undergoing hydrolysis in the low loading regime. Any changes to this fraction will result in proportional changes to the overall hydrolysis rate. Consequently, due to the chemical complexity of this process our range of rates should not be taken as upper and lower estimates of the impact of this channel. Rather, this range should be interpreted as evidence that the hydrolysis reaction may represent an important, previously unaccounted for ΣANs loss process as well as a potentially important source of HNO$_3$. This loss process of ΣANs is important in that unlike the oxidative pathway, hydrolysis represents a sink of ΣANs that removes NO$_x$ from the atmosphere.

5.4.2 Production of HNO$_3$

In addition to being a sink of ΣANs the hydrolysis reaction may also be an important source of HNO$_3$. As shown in Table 3, the ratio of this HNO$_3$ source to the known source from the reaction of OH with NO$_2$ and ranges from a median of 0.13 to greater than 1 in the enhanced loading regime. We believe that this upper limit is likely incompatible with the HNO$_3$ budget and is likely the result of extrapolating bulk solution rates to aerosol environments; however, we do find evidence of this HNO$_3$ source in the variation of the ratio of HNO$_3$ to NO$_2$ with NO$_x$. In the boundary layer when the lifetime of HNO$_3$ is short, HNO$_3$ is in photochemical steady-state and the ratio of HNO$_3$ to NO$_2$ should be proportional to the OH concentration (Day et al., 2008). We estimate the lifetime of HNO$_3$ to be $\sim 14\,$h, a value short enough that HNO$_3$ should be in diurnal steady-state. When there is a substantial concentration of ΣANs, the ratio of HNO$_3$ to NO$_2$ increases while NO$_x$ decreases. OH, however, exhibits the opposite trend and it decreases (Fig. 5a). For conditions of low ΣANs, the ratio of HNO$_3$ to NO$_2$ is more similar to OH. It is unlikely that variations in photochemical age are the dominant factor explaining the observed behavior of the HNO$_3$ to NO$_2$ ratio (Fig. 5a). The largest deviation in the expected behavior of the HNO$_3$ to NO$_2$ ratio as a function of NO$_x$ occurs at the lowest NO$_x$ concentrations – air masses which are likely to be more aged than those with higher NO$_x$ concentrations. However, the deviation only occurs in those air masses with a substantial concentration of both ΣANs (Fig. 5a) and monoterpene (not shown) and thus likely higher aerosol phase organic nitrates. Other than NO$_x$ concentration, other available chemical tracers for defining age with time zero at biogenic emissions were found to be unsuitable because of their direct correlations with ΣANs or because their sources were not unique.

This trend of increasing values as NO$_x$ decreases is also the same trend as the ratio of P(ΣANs) to P(HNO$_3$) (where $P(\text{HNO}_3) = k_{\text{OH+NO}_3}[\text{OH}][\text{NO}_2]$) as shown in Fig. 5b using results from the steady-state model in Browne and Cohen (2012). The similarity in magnitude between the HNO$_3$ to NO$_2$ ratio and the P(ΣANs) to P(HNO$_3$) ratio is expected if the hydrolysis of ΣANs constitutes the major loss process.
of ΣANs (i.e. that the ratio of hydrolysis to oxidation may be higher than our calculations here suggest). These results suggest that the ARCTAS HNO3 concentration is consistent with a source of HNO3 other than the reaction of OH with NO2 and that this source is likely the hydrolysis of ΣANs.

There are similar hints of this additional HNO3 source in a reinterpretation of data from previous experiments. Previous measurements of HNO3 have found evidence for a temperature dependent OH source (Day et al., 2008) and of an elevated within canopy OH concentration (Farmer and Cohen, 2008) in a ponderosa pine forest. However, these results are also consistent with a source of HNO3 from rapid ΣANs hydrolysis. For instance, the temperature dependent OH source may result from an increase in biogenic VOC emissions with temperature resulting in a larger ΣANs production and consequently a larger HNO3 source. Likewise, the rapid hydrolysis of ΣANs with low vapor pressures formed from sesquiterpenes and monoterpenes in the forest canopy would result in a within canopy source of HNO3. This reinterpretation of the HNO3 data as resulting from an additional production pathway (via hydrolysis of ΣANs) rather than through an elevated concentration of OH is also more consistent with OH measurements made in the same forest a few years later (Mao et al., 2012) that report a within-canopy OH gradient and temperature dependence smaller than that inferred from the previous studies. However, we note that these studies were conducted in different years and it is possible that the ecosystem and its within-canopy chemistry have changed in between those years.

It is interesting to consider the ultimate fate of the NO3− possibly produced by the organic nitrate hydrolysis. In 57% of the background measurements the molar ratio of sulfate to ammonium (as measured by the AMS) is greater than one-half, indicating that it is unfavorable for NO3− to be present in the aerosol and that ΣANs hydrolysis is possibly a source of gas phase HNO3. However, this is a simplistic approximation to an extremely complex problem. The thermodynamics of an aerosol that is an organic-inorganic mixture are much more complex (Zuend et al., 2011) than purely inorganic aerosols and are subject to uncertainties regarding the composition of aerosol and the interaction of ions with various functional groups present on organic species. Further studies on organic nitrate hydrolysis in aerosols are needed to better constrain the atmospheric impacts; however, it appears that the hydrolysis of organic nitrates may contribute (quite significantly) to HNO3 production.

These results suggest the need for research constraining the possible hydrolysis loss of ΣANs and the associated HNO3 production. In particular, we need measurements of how the hydrolysis of organic nitrates from biogenic species differs in aerosol versus bulk solution, the aerosol liquid water content necessary for this reaction, and specific rates for monoterpane nitrates.

6 Implications

As shown in Fig. 3, the calculated ΣANs production for most of the data is similar to the steady-state model results from Browne and Cohen (2012) if we assume a branching ratio somewhere between 5% and 10% for ΣANs formation from the entire VOC mixture. For the ARCTAS data we calculate that the biogenic VOCs account for ~53% of the VOC reactivity with respect to OH (median value not including CO and CH4). Assuming that the biogenic VOCs are the only sources of ΣANs with an average branching ratio of 11% (similar to isoprene), results in an overall branching ratio of ~6%. This suggests that the NOx lifetime and ozone production efficiency in the boreal forest are similar to those calculated in
Browne and Cohen (2012) and that the steady-state model provides a useful framework for understanding the NOx budget under low NOx conditions on the continents.

However, as discussed in Browne and Cohen (2012), the net regional and global impact of ΣANs on NOx lifetime and ozone production depends on the degree to which ΣANs serve as a permanent versus temporary NOx sink. Modeling studies have found that different assumptions regarding NOx recycling from isoprene nitrates result in large sensitivities in NOx and O3 (e.g. von Kuhlmann et al., 2004; Paulot et al., 2012; Fiore et al., 2005; Horowitz et al., 2007; Wu et al., 2007) and that these uncertainties affect predictions of ozone in a future climate (e.g. Ito et al., 2009; Weaver et al., 2009).

The analysis presented here suggests that ΣANs have a short atmospheric lifetime due to a combination of deposition and chemical loss, but we find the data is ambiguous about the relative fraction of the ΣANs chemical loss that acts to release NOx or to produce HNO3. Furthermore, the exact fate of ΣANs loss is likely ecosystem dependent; for instance, ΣANs may have a significantly different impact on the NOx budget in forests dominated by isoprene emissions versus in forests dominated by monoterpane emissions since first generation monoterpene nitrates have lower vapor pressures than first generation isoprene nitrates.

Due to the lumped treatment of ΣANs in most condensed chemical mechanisms, it is likely that these mechanisms will be unable to reproduce the ARCTAS results, and consequently are misrepresenting the NOx lifetime and ozone production. For instance, some condensed mechanisms instantaneously convert isoprene nitrates to HNO3, resulting in zero NOx recycling. The ozonolysis of isoprene nitrates is also ignored in many mechanisms; this is incompatible with our results that the majority of NOx recycling during ARCTAS results from ozonolysis. Lastly, many condensed mechanisms ignore monoterpene nitrates or lump them into a long-lived nitrate. Our results suggest that, at least in the boreal forest, monoterpane nitrates are an important NOx sink and that their particle phase hydrolysis may represent a source of HNO3.

Finally, it is interesting to note that since the loss of ΣANs through hydrolysis depends on the specific isomer of the nitrate, there are interesting implications for the loss of monoterpane nitrates formed from OH versus from NO3 chemistry. Based on the assumption that the assumption that radical reactions are more stable than primary radicals and thus have a higher nitrate yield, the generation of α- and β-pinene and limonene by NO3 is more likely to result in a primary nitrate and oxidation by OH is more likely to result in a tertiary nitrate. Thus, nitrates formed by OH oxidation may have a shorter atmospheric lifetime than those formed from NO3 chemistry.

7 Conclusions

We present the first measurements of ΣANs over the remote boreal forest of Canada and show that ΣANs are present in significant concentrations. Using measurements of VOCs we calculate the instantaneous production rate of ΣANs and find that, as expected for a remote forested environment, biogenic species, specifically monoterpenes and isoprene, dominate the ΣANs production. If the observations of α- and β-pinene understate the total source of monoterpenes then monoterpenes play an even larger role, than the 25% we calculate. We also find that the instantaneous production rate of ΣANs is, in general, faster than that of gas phase HNO3 production, despite a lower overall concentration, implying that ΣANs have a shorter lifetime than HNO3. We estimate that depositional loss of ΣANs is important and that the combined loss to reaction with O3 and OH occurs at a rate similar to the assumed deposition rate of HNO3. Oxidation of isoprene nitrates, in particular by O3, is primarily responsible for the rapid loss rate. We emphasize that this oxidative loss rate represents the loss of the nitrate functionality and that oxidative reactions of individual nitrates are faster since some of their products are more highly functionalized nitrates.

We also provide evidence which suggests that particulate organic nitrates undergo rapid hydrolysis contributing to HNO3 production. Although, we are unable to constrain the magnitude of this source precisely, all reasonable assumptions imply that it is significant both as a loss of ΣANs and as a source of HNO3. Furthermore, there is evidence of its existence in the variation of the HNO3 to NO2 ratio as a function of NOx. We conclude that the rapid loss of ΣANs required to explain these observations is a balance between processes which recycle NOx (oxidation) and those which remove it (hydrolysis and deposition).

Appendix A

A1 Calculation of ΣANs production

In the calculation of the ΣANs production rate we use the VOCs, branching ratios, and OH reaction rates listed in Table A1. We do not attempt to estimate the concentration of any unmeasured VOCs or to fill in any missing data.

A2 Uncertainties in the calculation of ΣANs production

The calculated production of ΣANs is sensitive to the assumptions about reaction rates, organic nitrate branching ratios, the assumption that the VOC measurements are representative of the entire VOC mix, and possible errors in measurements. We have investigated several possibilities (outlined in Table A2) and find that our conclusion is robust. In Table A2 we show the median value of the ratio of the
Table A1. VOC parameters used in the calculation of the instantaneous production rate of $\Sigma$ANs.

| VOC                        | $\alpha$   | OH reaction rate | VOC                        | $\alpha$   | OH reaction rate |
|----------------------------|------------|------------------|----------------------------|------------|------------------|
| Alkanes                    |            |                  | Alkenes                    |            |                  |
| methane$^b$                 | 0.001$^c$  | 2.45E-12 $\times$ exp$(-1775/T)^k$ | ethane$^b$                 | 0.0086$^h$ | $k_o = 1.0E-28 \times (T/300)^{-4.5}$ |
| ethane$^b$                 | 0.009$^e$  | 7.66E-12 $\times$ exp$(-1020/T)^k$ | propene$^b$                | 0.015$^f$  | $k_o = 8E-27 \times (T/300)^{-3.5}$ |
| propane$^b$                | 0.036$^i$  | 8.7E-12 $\times$ exp$(-615/T)^k$ | 1-butene$^b$               | 0.039$^g$  | 6.6E-12 $\times$ exp$465/T)^l$ |
| n-butane$^c$               | 0.077$^l$  | 9.8E-12 $\times$ exp$(-425/T)^l$ | methylpropene$^b$          | 0.012$^e$  | 9.4E-12 $\times$ exp$505/T)^l$ |
| n-pentane$^c$              | 0.129$^f$  | 1.81E-11 $\times$ exp$(-452/T)^m$ | trans-2-butene$^b$         | 0.034$^b$  | 1.0E-11 $\times$ exp$553/T)^l$ |
| i-butane$^c$               | 0.027$^g$  | 7.0E-12 $\times$ exp$(-350/T)^m$ | cis-2-butene$^b$           | 0.034$^b$  | 1.1E-11 $\times$ exp$485/T)^l$ |
| i-pentane$^c$              | 0.075$^o$  | 1.01E-11 $\times$ exp$(-296/T)^m$ | butadiene$^b$              | 0.065$^e$  | 1.58E-11 $\times$ exp$436/T)^n$ |
| 2,3-dimethylbutane$^b$     | 0.061$^e$  | 1.25E-11 $\times$ exp$(-212/T)^m$ | benzene$^d$                | 0.029$^d$  | 2.3E-12 $\times$ exp$(-190/T)^l$ |
| 2-methylpentane and 3-methylpentane$^b$ | 0.11$^e$ | 1.77E-11 $\times$ exp$(-362/T)^m$ | propyl-benzene$^b$         | 0.093$^d$  | 5.8E-12$^p$ |
| hexane$^b$                 | 0.141$^g$  | 1.98E-11 $\times$ exp$(-394/T)^m$ | toluene$^d$                | 0.08$^e$  | 1.8E-12 $\times$ exp$340/T)^l$ |
| heptane$^b$                | 0.178$^8$  | 2.76E-11 $\times$ exp$(-430/T)^m$ | 2-ethyltoluene$^b$         | 0.106$^e$  | 1.19E-11$^p$ |
| Isopropanes and derivatives|            |                  |                            |            |                  |
| isoprene$^d$               | 0.117$^i$  | 2.7E-11 $\times$ exp$390/T)^l$ | 3-ethyltoluene$^b$         | 0.094$^e$  | 1.86E-11$^p$ |
| methacrolein$^c$           | 0.0705$^i$ | 8.0E-12 $\times$ exp$380/T)^l$ | 4-ethyltoluene$^b$         | 0.137$^e$  | 1.18E-11$^p$ |
| methyl vinyl ketone$^c$     | 0.11$^i$   | 2.6E-12 $\times$ exp$610/T)^l$ | o-xylene$^c$               | 0.081$^e$  | 1.36E-11$^p$ |
| Monoterpenes               |            |                  | m-xylene and p-xylene$^b$  | 0.085$^g$  | 2.31E-11$^p$ |
| $\alpha$-pinene$^b$        | 0.18$^j$   | 1.2E-11 $\times$ exp$440/T)^l$ | ethylbenzene$^b$           | 0.072$^e$  | 7.0E-12$^p$ |
| $\beta$-pinene$^b$         | 0.18$^j$   | 1.47E-11 $\times$ exp$467/T)^o$| 1,3,5-trimethylbenzene$^b$ | 0.127$^g$  | 5.67E-11$^p$ |
| $\gamma$-pinene$^b$        |            |                  | 1,2,4-trimethylbenzene$^b$ | 0.105$^g$  | 3.25E-11$^p$ |
| $\delta$-pinene$^b$        |            |                  | 1,2,3-trimethylbenzene$^b$ | 0.119$^e$  | 3.27E-11$^p$ |
| OVOCs                      |            |                  |                            |            |                  |
| butanone$^c$               | 0.015$^e$  | 1.5E-12 $\times$ exp$(-90/T)^l$ |                            |            |                  |
| butanol$^c$                | 0.013$^e$  | 6.0E-12 $\times$ exp$410/T)^l$  |                            |            |                  |

$^a$ Tunable diode laser absorption spectroscopy; $^b$ Whole air sampling; $^c$ Gas chromatography – mass spectrometry; $^d$ Proton transfer reaction mass spectrometry; $^e$ MCM v3.2, may involve weighting by isomers; $^f$ Atkinson et al. (1982); $^g$ Arye et al. (2001); $^h$ O’Brien et al. (1995); $i$ Paulot et al. (2009); $j$ Nozière et al. (1999); $k$ Sander et al. (2006); $^l$ Atkinson et al. (2006); $^m$ Calvert et al. (2008); $n$ Li et al. (2006); $^o$ Gill and Hites (2002); $p$ Calvert et al. (2002); $q$ Rate for 2-methylpentane; $r$ $k = k_{\infty}$ $k_o M/(k_o M + k_{\infty}) \times F^k_j, j = 1 + \log(k_o M/k_{\infty})/N)^2, \log = \log_{10}$

instantaneous production of $\Sigma$ANs to HNO$_3$ for ten different possibilities (including our base case that was presented in the text). In the unique RO$_2$ + RO$_2$ rate case we take the rate of RO$_2$ + RO$_2$ reactions from MCM v3.2 RO$_2$ + CH$_3$O$_2$ rates for methyl vinyl ketone, methacrolein, isoprene, and monoterpenes. We weight the methyl vinyl ketone and isoprene rates by the initial branching of the different peroxy radicals. The monoterpenes rate is calculated assuming an even split between $\alpha$- and $\beta$-pinene and weighting the different peroxy radicals. No significant difference is observed using these rates. If we increase the isomerization rate of the isoprene peroxy radical by an order of magnitude (Isomerization $\times$ 10 case), we also observe no significant difference.

Recent measurements of the isoprene nitrate branching ratio range from 7% to 12% (Paulot et al., 2009; Lockwood et al., 2010). In our base calculation we use the branching ratio of 11.7% reported by Paulot et al. (2009). In the 7% IN case below, we use the yield of 7% measured by Lockwood et al. (2010) and find that although the contribution from isoprene decreases, P($\Sigma$ANs) is still larger than P(HNO$_3$).

It is also likely that there are VOCs contributing to organic nitrate production that were not measured during ARC-TAS, and thus the base calculation is biased low. For instance, only the monoterpenes $\alpha$-pinene and $\beta$-pinene are measured. Measurements from the boreal forest in Finland indicate substantial contributions from other monoterpenes as well as contributions from sesquiterpenes (Spirig et al., 2004; Räisänen et al., 2009; Hakola et al., 2012). As expected, if we double the production from monoterpenes (2 $\times$ Monoterpenes) to account for unmeasured species, we see an increases in the ratio of P($\Sigma$ANs) to P(HNO$_3$).
Table A2. The median value of the \( P(\Sigma ANs) / P(HNO_3) \) ratio and the speciation of \( P(\Sigma ANs) \) for different assumptions regarding RO2 reaction rates, OH and HO2 concentrations, and VOC concentrations as described in Appendix A.

| Case                        | \( P(\Sigma ANs)/P(HNO_3) \) |
|-----------------------------|-------------------------------|
| Base                        | 1.96                          |
| Unique RO2 + RO2 rate       | 1.84                          |
| Isomerization \( \times 10 \) | 1.42                          |
| 7 % IN                      | 1.54                          |
| 2 \times Monoterpenes       | 2.57                          |
| HO2 CIMS OH                 | 1.60                          |
| [HO2] \( \times 0.6 \)      | 2.83                          |
| HO2 CIMS HO2                | 2.04                          |
| [RO2] \( \times 10 \)       | 1.82                          |
| Steady-state NO             | 3.00                          |

In our base calculation we use the LIF OH measurement. It has recently been shown that this measurement may have an interference in environments with high biogenic emissions (Mao et al., 2012). This should have a minor effect on our calculation since any change in OH will affect both \( P(\Sigma ANs) \) and \( P(HNO_3) \). Nevertheless, we test this possibility using the OH measurement from the chemical ionization mass spectrometry instrument (Cantrell et al., 2003) – the HOx CIMS OH case. These two different measurements agreed well during the campaign (Ren et al., 2012). We see a slight decrease in the ratio of \( P(\Sigma ANs) \) to \( P(HNO_3) \), however this can be attributed to the discrepancy in data coverage between the two instruments; if we restrict the LIF OH to the same points with CIMS OH coverage, we calculate the same median ratio.

Recently it has been reported that some LIF HO2 measurements may suffer from a positive interference from the conversion of RO2 to HO2 in the instrument (Fuchs et al., 2011). This should increase our production of \( \Sigma ANs \) relative to HNO3 due to an increase in the fraction of RO2 that reacts with NO. If we decrease the HO2 by 40 % ([HO2] \( \times 0.6 \) case), we find this to be true ([HO2] \( \times 0.6 \) case). Using the HOx CIMS HO2 measurement also results in an insignificant change to the median \( P(\Sigma ANs) \) to \( P(HNO_3) \).

In low NOx environments it is possible that RO2 is present in higher concentrations than HO2 which would decrease our \( \Sigma ANs \) production. However, we find that increasing the RO2 concentration by an order of magnitude (RO2 \( \times 10 \) case) has a negligible effect on our calculation. Even if this increase is coupled with a doubling of the RO2 + RO2 rate (not shown), there is no significant effect. Furthermore, the HOx CIMS measurements of RO2 do not show any evidence that the RO2 to HO2 ratio has any significant increase at low NOx.

Lastly, we investigate the sensitivity of the calculation to the NO concentration. Using the NO concentration calculated assuming a steady-state between NO and NO2 and the measured concentrations of NO2, HO2, O3, NO2 photolysis, and assuming that RO2 is equal to HO2 we find an increase in the median of the \( P(\Sigma ANs) \) to \( P(HNO_3) \) ratio (Steady-state NO case). Overall, we conclude that although there is uncertainty in the absolute numbers, the production of \( \Sigma ANs \) is, on average, faster than the production of HNO3.

Appendix B

Uncertainty in the calculated \( \Sigma ANs \) oxidation rate

The calculated oxidation rate of \( \Sigma ANs \) is sensitive to uncertainties and assumptions including: the assumption that the instantaneous production represents the composition, possible interferences in HOx measurements, reaction rate uncertainties, and assumptions regarding NOx recycling.

Two of the most likely deviations from our assumption that the production in Fig. 4 represents the composition are nitrates produced from unmeasured BVOCS (likely monoterpenes and sesquiterpenes) and the presence of higher generation isoprene and monoterpane nitrates. In order for these nitrates to increase the \( \Sigma ANs \) loss rate, their loss rate must, on a per molecule basis, be faster than the isoprene nitrate loss which implies that these nitrates are unsaturated. In Browne et al. (2013) we estimate the oxidation rates of unsaturated monoterpane nitrates \( 7.29 \times 10^{-11} \text{cm}^3 \text{molecules}^{-1} \text{s}^{-1} \) for OH oxidation and \( 1.67 \times 10^{-10} \text{cm}^3 \text{molecules}^{-1} \text{s}^{-1} \) for ozonolysis, similar to the isoprene nitrate oxidation rates. Thus, if the monoterpane nitrates had a larger NOx recycling than the isoprene nitrates, then they would increase the \( \Sigma ANs \) loss. NOx recycling from monoterpane nitrates is difficult to estimate given the number of different monoterpene structures and the variability of emission factors between species. Furthermore, since the ozonolysis of the nitrates will dominate the loss process, NOx recycling through this channel will be most important. To our knowledge, there are no measurement constraints on the NOx recycling from the ozonolysis of any organic nitrate.

To estimate the effect of this complex problem we use results from the WRF-Chem model run over the boreal forest of Canada for the ARCTAS time period. This model uses a chemical mechanism with a comprehensive treatment of \( \Sigma ANs \) including 11 isoprene-derived nitrates and two monterpene-derived nitrates (one unsaturated and one saturated) as described in Browne et al. (2013) and Browne (2012). Sampled along the flight track, the WRF-Chem model predicts a \( \Sigma ANs \) oxidative loss rate of \( 2.3 \times 10^{-5} \text{s}^{-1} \) (median), a number similar to our estimate here, which suggests that these effects have only a small influence on our calculation.

Since the ozonolysis of isoprene nitrates accounts for the majority of the \( \Sigma ANs \) loss rate, the possible interferences in the HOx measurements (OH and HO2) and uncertainties in the RO2 reaction rates have a negligible effect on our calculated loss. Consequently, the uncertainties regarding isoprene nitrate ozonolysis, particularly the yields of the various...
isoprene nitrate isomers, the ozonolysis rates, and the magnitude of the NOx recycling are non-negligible. If we use the split between the δ-hydroxy and β-hydroxy nitrates from Lockwood et al. (2010) (with updates from Pratt et al., 2012) (∼10% and ∼90%, respectively) and the distribution of ΣANs production calculated using the isoprene nitrate formation yield from Lockwood et al. (2010) (51% isoprene, 12% MVK, 33% α- and β-pinene), we calculate an overall ΣANs loss rate of 1.4 × 10^{-3} s^{-1} assuming the slower β-hydroxy rate and 3.8 × 10^{-3} s^{-1} if we assume the faster rate. These rates are similar to those in Table 2. We note that we have weighted the ozonolysis rates using the initial production yields of the β-hydroxy and δ-hydroxy nitrates. Given that these nitrates have (potentially) different atmospheric lifetimes (at 2°C) of 0.97–2.6 h using the OH rate constants from Paulot et al. (2009), it is likely that the reaction rate of the ΣANs we measure will favor the less reactive nitrates and our calculation may be high. Lastly, in our derivation of the NOx recycling we follow the assumptions of MCM v3.2, which include the assumption of equal branching between the two possible carboxyl/Criegee biradical pairs. However, the exact branching depends on nature and number of the substituents on the alkene (Calvert et al., 2000).

There is also uncertainty introduced via our assumption that when the nitrooxy peroxy radical formed via OH oxidation reacts with HO2 the nitrate functionality is preserved. Recent experimental work on the nitrooxy peroxy radicals derived from the reaction of isoprene with NO3 indicates that the reaction of this peroxy radical with HO2 likely has a large flux through the channel forming radical products (i.e. the alkoxyl radical and OH) (Kwan et al., 2012). If we assume that this channel occurs half of the time, which is within the range estimated by Kwan et al. (2012), we calculate that the oxidation rate increases by 66% when we assume the slower isoprene nitrate ozonolysis rate and 26% when the faster rate is assumed.

It is also possible that reaction of the nitrooxy peroxy radical with other RO2 (in particular, acyl peroxy radicals) may proceed at a faster rate. For instance, the reaction rate of CH3C(O)2O with CH3O2 at 285 K is approximately two orders of magnitude faster than the self reaction rate of CH3O2 (Atkinson et al., 2006). If we increase the RO2 rate constant by a factor of 50, an increase which is consistent with assuming that about half the peroxy radicals react with a rate of 2.3 × 10^{-11} cm^3 molecules^{-1} s^{-1} rather than 2.3 × 10^{-13} cm^3 molecules^{-1} s^{-1} (i.e. are more like acyl peroxy radicals), we calculate that the RO2 + RO2 reaction occurs ∼30% of the time. This increases the oxidation rate to 2.2–3.6 × 10^{-5} s^{-1}. We note that this likely overestimates the number of peroxy radicals. Furthermore, in our analysis we have assumed that the products of the nitrooxy peroxy radical reaction with other RO2 are the same as those when it reacts with NO (i.e. we assume that the channel forming RO is dominant). While this channel is likely favored when the reaction is with an acyl peroxy radical, molecular channels which will retain the nitrate will likely be more important for non-acyl peroxy radicals. For instance, Kwan et al. (2012) estimate that only 19–38% of the RO2 + RO2 reactions in their study result in the formation of alkoxyl radicals. This decrease in alkoxyl radical formation will also decrease the calculated oxidation rate.

Overall, these calculations suggest that ozonolysis of isoprene nitrates is the largest oxidation sink of organic nitrates. Further experimental constraints on the ozonolysis rates and products of the isoprene nitrates are needed to reduce the uncertainty concerning the fraction of NOx that is recycled back to the atmosphere. Additional experiments constraining the products of isoprene-derived nitrooxy peroxy radicals with HO2 and other peroxy radicals are also needed in order to understand the oxidation of these nitrates under low NOx conditions.

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