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Understanding in situ ozone production in the summertime through radical observations and modelling studies during the Clean air for London project (ClearfLo)

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Abstract. Measurements of OH, HO₂, RO₂*i* (alkene and aromatic related RO₂) and total RO₂ radicals taken during the ClearfLo campaign in central London in the summer of 2012 are presented. A photostationary steady-state calculation of OH which considered measured OH reactivity as the OH sink term and the measured OH sources (of which HO₂+NO reaction and HONO photolysis dominated) compared well with the observed levels of OH. Comparison with calculations from a detailed box model utilising the Master Chemical Mechanism v3.2, however, highlighted a substantial discrepancy between radical observations under lower NO_x conditions ([NO] < 1 ppbv) typically experienced during the afternoon hours, and indicated that the model was missing a significant peroxy radical sink; the model over-predicted HO₂ by up to a factor of 10 at these times. Known radical termination steps, such as HO₂ uptake on aerosols, were not sufficient to reconcile the model measurement discrepancies alone suggesting other missing termination processes. This missing sink was most evident when the air reaching the site had previously passed over central London to the east and when elevated temperatures were experienced and, hence, contained higher concentrations of VOC. Uncertainties in the degradation mechanism at low NO_x of complex biogenic and diesel related VOC species which were particularly elevated and dominated OH reactivity under these easterly flows, may account for some of the model measurement disagreement. Under higher [NO] (>3 ppbv) the box model increasingly under-predicted total [RO₂]. The modelled and observed HO₂ were in agreement, however, under elevated NO concentrations ranging from 7 – 15 ppbv.

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The model uncertainty under low NO conditions leads to more ozone production predicted using modelled peroxy radical concentrations (~3 ppbv hr⁻¹) versus ozone production from peroxy radicals measured (~1 ppbv hr⁻¹). Conversely, ozone



production derived from the predicted peroxy radicals is up to an order of magnitude lower than from the observed peroxy radicals as [NO] increases beyond 7 ppbv due to the model under-prediction of RO₂ under these conditions.

1 Introduction

5 With greater than 50 % of the global population residing in urban conurbations, poor urban air quality has a demonstrable effect on human health. OH and HO₂ radicals, (collectively termed HO_x) together with RO₂ radicals, mediate virtually all of the oxidative chemistry in the atmosphere. The hydroxyl radical initiates the removal of primary emissions, including toxic gases such as CO and benzene, leading to the formation of peroxy radicals which, in the presence of NO, form secondary pollutants such as NO₂, O₃ and particulates. Public Health England (2014) reports that pollutants contribute to 29,000 deaths a year in the UK, with a reduction in life expectancy (by an average of 6 months) caused by the long-term exposure to pollutants, and the cost to society is estimated at up to £20 billion per year. In areas of London up to 1 in 12 deaths are at least partly attributable to air pollution, yet big uncertainties still remain relating to the chemistry, transformation and removal rate of primary emissions in large urban conurbations, meaning our ability to predict pollution episodes is compromised.

10 The EU air quality guidelines recommend that ozone concentrations do not exceed 60 ppbv for greater than an 8 hour period (<http://www.eea.europa.eu/themes/air/ozone>), with a 10 ppbv increment in long term exposure to ozone increasing the risk of death from respiratory causes by ~ 3 – 4% (Jerrett et al., 2009). Short-term exposure to elevated levels of tropospheric ozone have been associated with several adverse health effects including, for example, exacerbation of asthma in children (Thurston et al., 1997).

20 Despite successful reductions in many ozone precursors across Europe, ozone levels have increased at certain urban sites due to the long-term decrease in NO_x emissions. For example, Bigi and Harrison (2010) report a steady increase in ozone between 1996-2008 in North Kensington; an urban background site in London.

To implement efficient reduction strategies for ozone, a detailed understanding of the factors controlling free radicals is critical since the reaction of HO₂ and RO₂ radicals with NO, forming NO₂, followed by the subsequent photolysis of NO₂ represents the only net formation pathway to tropospheric ozone:





5 Measurements of radicals have been made at various urban and sub-urban locations worldwide, both during the summer and winter (Stone et al. (2012) and references therein). Observations of OH and HO₂ in the urban atmosphere have primarily been made using fluorescence assay by gas expansion (FAGE), and comparisons with predicted radical concentrations using chemistry box models constrained with co-located radical precursor measurements have revealed varying levels of success in replicating observations. Radical concentrations have been reported to be under-predicted by models (Ren et al., 2003; Martinez et al., 2003; Emmerson et al., 2005a; Chen et al., 2010; Lu et al., 2012; Lu et al., 2013), over-predicted (George et al., 1999; Konrad et al., 2003; Dusanter et al., 2009) and, at times, models and measurements have been reported to be in reasonable agreement, to within 40%, (Shirley et al., 2006; Emmerson et al., 2007; Kanaya et al., 2007; Sheehy et al., 2010; Elshorbany et al., 2012; Ren et al., 2013; Griffith et al., 2016). Often the level of agreement observed was found to be dependent on time of day (Brune et al., 2016); with poorest agreement between modelled and measured OH concentrations generally observed during the night. Griffith et al. (2016) found that the level of agreement between modelled and measured HO₂ was dependent on whether it was a weekday or weekend; the model under-predicted HO₂ by a factor of 3.4 at weekends but agreed well on weekdays (observed to modelled HO₂ = 1.3). In a number of studies, the model-measurement discrepancy was noted to increase as NO_x levels increased beyond ~1 ppbv (Martinez et al., 2003; Ren et al., 2013; Brune et al., 2016). This increasing under-prediction of the free-radicals (particularly for HO₂) with increasing NO_x concentrations observed may reflect inaccuracies in the radical propagation steps in the model, which cycle HO₂ to OH. In light of the recently reported RO₂ interference suffered by FAGE (Fuchs et al., 2011; Whalley et al., 2013) when detecting HO₂, however, it is possible that the measured HO₂ under these conditions may have been increasingly influenced by the presence of RO₂ species. The extent of this interference will be dependent upon the level of interference suffered by the specific FAGE instrument utilised and the concentration of those RO₂ species that interfere (principally aromatic, alkene and >C₃ alkane-derived RO₂ species) that were present in a particular environment. Similarly, two FAGE groups have reported interferences in their OH measurements made using wavelength modulation in the presence of ambient levels of ozone and alkenes (Mao et al., 2012; Novelli et al., 2014), whilst, in contrast, good agreement between OH measurements made using FAGE and differential optical absorption spectroscopy (DOAS) during chamber measurements suggests minimal interferences in the presence of ozone and alkenes for a third FAGE instrument (Fuchs et al., 2013). This lack of interference was further corroborated in recent laboratory tests (Fuchs et al., 2016) although an artefact signal under dark conditions (deriving from NO₃ in the presence of H₂O) was identified. These potential artefacts make it difficult to identify trends in earlier model-measurement comparisons and to assess how well the models are performing under a range of chemical conditions. Some of the more recently published radical measurements at urban sites include corrections for OH interferences, e.g. (Ren et al., 2013; Brune et al., 2016; Griffith et al.,



2016) and radical measurements from the MEGAPOLI project which took place at a suburban site close to Paris employed the chemical ionisation mass spectrometry (CIMS) technique to make observations of OH and the sum of HO₂ and RO₂ species rather than FAGE (Michoud et al., 2012). HO₂* (= [HO₂] + ∑_iα_i [RO_{2i}]), and α_i is the mean fractional contribution of the RO₂ species that interfere (RO_{2i}) model-measurement comparisons are now often reported (rather than HO₂) to take into account
5 contributions from RO₂ species (Lu et al., 2013; Griffith et al., 2016) and very recently Tan et al. (2017) presented interference-free HO₂ observations alongside RO₂ observations which were made using the FAGE technique coupled to a flow reactor (Fuchs et al., 2008) at a rural site in Wangdu, China. In contrast to some of the earlier HO₂ model-measurement comparisons which diverged at NO concentrations > 1ppbv with models increasingly under-predicting the levels of HO₂ observed, the predicted levels of HO₂ were in good agreement with HO₂ observations made in Wangdu over the whole range (0.1 – 4 ppbv)
10 of NO encountered (Tan et al., 2017). However, the authors did report an increasing model under-prediction of the observed RO₂ with increasing NO (Tan et al., 2017).

Despite uncertainties in some of the earlier radical observations and discrepancies between observed and predicted radical concentrations, detailed modelling studies have demonstrated a number of common themes relevant to urban photochemistry:

- 15 1. The primary source of OH from the photolysis of ozone and subsequent reaction of the excited state oxygen atom with H₂O, which is often considered the dominant radical source in many other environments (e.g in the remote marine atmosphere (Whalley et al., 2010)) tends only to play a minor role in urban centres, with this source accounting for < 6% of the total radical sources during MCMA-2006 (Dusanter et al., 2009) which took place in Mexico City.
- 20 2. Owing to the prevalence of carbonyl and dicarbonyl species in the urban atmosphere, a number of studies have highlighted the role that the photolysis of these species play as key radical precursors (and, hence, ozone precursors) in the summertime: During the SHARP-2009 project that took place in Houston, Texas, the photolysis of formaldehyde accounted for 14% of radical production, with the photolysis of other OVOCs contributing a further 15% (Ren et al., 2013). During the CAREBeijing2006 HCHO was estimated to contribute ~30% to the overall radical production (Lu et al., 2013).
- 25 3. Ozonolysis reactions have been reported as important primary radical sources during a number of studies, for example these reactions accounting for 67% of the OH initiation in Birmingham during the PUMA campaign in winter (Emmerson et al., 2005b) whilst in Tokyo during the IMPACT campaign ozonolysis reactions were the dominant radical source during the night-time in winter (Kanaya et al., 2007).
- 30 4. The photolysis of HONO, which takes place at longer wavelengths than ozone photolysis, has been demonstrated to act as an important OH source in the morning (Kleffmann, 2007). At urban sites (including London), significant concentrations of HONO (often several hundred pptv) have been reported to persist throughout the day (Lee et al., 2016), and, as such, HONO should be considered an important OH source throughout sunlit hours, not just at sunrise, in these environments. Dusanter et al. (2009) found that HONO photolysis contributed 35% of daytime HO_x



production in Mexico City during MCMA 2006, whilst Tan et al. (2017) found that HONO photolysis was the most important primary radical source in Wangdu in the North China Plain.

There have now been several observations of total OH reactivity (k_{OH}) in urban environments. With some of the highest reactivities of $>120 \text{ s}^{-1}$ recorded in the megacities such as Mexico City, London and Paris. In many of the large cities (Houston, New York City, Mexico City), OH reactivity has been found to be dominated by anthropogenic hydrocarbons, CO and NO_x . OVOCs have been highlighted as significant OH sinks in a number of urban studies, contributing between 11–24% during summertime at these urban centres (Mao et al., 2010b), whilst we recently reported that the oxidation products of biogenic emissions contributed a significant fraction to the total OH reactivity observed in London (Whalley et al., 2016). A measurement of OH reactivity can provide an additional model target, with model-measurement comparisons helping identify unmeasured primary emissions or unmeasured oxidised intermediates which may promote radical propagation. Furthermore, when coupled with OH (and HO_2) observations, the closure of OH production (P_{OH}) and OH loss ($D_{OH} = k_{OH}[\text{OH}]$) terms can be critically assessed independent of a model. In an urban atmosphere the dominant OH sources include recycling from $\text{HO}_2 + \text{NO}$, HONO photolysis, $\text{O}(^1\text{D})$ (from ozone photolysis) $+ \text{H}_2\text{O}$ and ozonolysis reactions. In the recent study in the Wangdu region of China, P_{OH} was found to equal D_{OH} within uncertainties throughout the day demonstrating consistency between the observed radical concentrations and observed OH reactivity. Several previous studies in urban regions, however, have found that P_{OH} is balanced by D_{OH} during the afternoon but not in the mornings, with measured P_{OH} approximately twice D_{OH} from sunrise to noon (Brune et al., 2016). This imbalance of P_{OH} and D_{OH} suggests either a negative bias of OH reactivity measurements, an error in the HO_2 measurement or uncertainties in the chemistry of other key OH sources, e.g. HONO (Brune et al., 2016).

Urban radical measurements can be used to estimate local ozone production (Kanaya et al., 2007; Ren et al., 2013; Brune et al., 2016) by approximating the rate of ozone production to the production rate of NO_2 from the reaction of NO with HO_2 and RO_2 radicals, and assuming instantaneous O_3 production following photolysis of NO_2 at wavelengths $<400 \text{ nm}$. Any loss of NO_2 which does not yield O_3 , for example the reaction of OH or RO_2 radicals with NO_2 , and also deposition, should also be considered:

$$P(\text{O}_3) = (k_{\text{HO}_2+\text{NO}}[\text{HO}_2][\text{NO}] + k_{\text{RO}_2+\text{NO}}[\text{RO}_2][\text{NO}]) - (k_{\text{OH}+\text{NO}_2+\text{M}}[\text{OH}][\text{NO}_2][\text{M}] + k_{\text{RO}_2+\text{NO}_2+\text{M}}[\text{RO}_2][\text{NO}_2][\text{M}] + \text{deposition}) \quad (1)$$

Given the short lifetime of the radicals, this estimate provides a method of gauging the extent to which the fast local chemistry influences the net ozone levels observed relative to O_3 generated during transport. A shortcoming of this approach in earlier studies is that often the RO_2 concentration used in E1 is estimated or modelled, as traditionally the FAGE technique measures



OH and HO₂ only. In the Wangdu study, however, Tan et al., (2017) using observed RO₂ demonstrated that models may under-predict ozone production at high NO due to an underestimation of the RO₂ radical concentration.

In the present paper we utilise observations of OH and HO₂ radicals made using the FAGE technique and RO₂ radicals using the RO_xLIF method (Whalley et al., 2013). The radical observations were made during the Clean air for London project (ClearfLo) during the summer of 2012 and are used to directly determine local ozone production. To assess the factors controlling the radical budget and in turn ozone production, we have employed a detailed box model based on the MCM v3.2. By comparing model predictions to radical observations the key reactions taking place in London that are ultimately controlling the air quality are identified and uncertainties in our current understanding of urban oxidation chemistry is highlighted.

10 2 Experimental

2.1 Site description

The ClearfLo Intensive Operation Period (IOP) ran from 22nd July to 18th August and overlapped with the London 2012 summer Olympics. An extensive suite of instrumentation was deployed and operated from the grounds of Sion Manning School in North Kensington (51° 31' 16" North, 0° 12' 48" West), which is located adjacent to a long-term air quality monitoring site in North Kensington (Bigi and Harrison, 2010). Further details on the campaign and location may be found in Bohnenstengel et al. (2015).

2.2 FAGE instrument description

The University of Leeds ground-based FAGE instrument was deployed to the North Kensington site and made measurements of OH, HO₂ and RO₂ radicals. Further details on the instrument for OH and HO₂ detection can be found in Whalley et al. (2010) with only an outline of the specific set-up and running conditions during ClearfLo described here. The radical measurements were made from a 20 ft air-conditioned shipping container which had been converted into a mobile laboratory. The instrument consists of two FAGE detection cells which were located on the roof of the shipping container, in a weather-proof housing, at a height of 3.5 m. A Nd:YAG pumped Ti:Sapphire laser (Photonics Industries) generated pulsed (repetition rate of 5 kHz), tunable near IR radiation, which was frequency doubled and tripled to provide UV light at 308 nm and was used to excite OH via the Q₁(1) transition of the A²Σ⁺, v'=0 ← X²Π_i, v''=0 band. On-resonance fluorescence was detected using a gated channel photo-multiplier and photon counting for a period of 300 s. The laser was then scanned beyond the OH transition (by 0.004 nm) and a background signal collected for a further 75 s to determine the contribution of laser, solar scatter and detector noise to the total signal for subtraction (OH_{WAVE}).

In previous configurations, the two detection cells were used to simultaneously detect OH by laser-induced fluorescence (LIF) (cell 1) and HO₂ by NO titration to OH followed by LIF (cell 2). The UV laser light was split upon exiting the laser and focussed into fibre optics (5 m length) for delivery to each cell individually. During ClearfLo, the two cells were coupled



together via a connecting side arm, which enabled light exiting cell 1 to pass into cell 2, and meant that light previously needed for the detection of HO₂ in cell 2 could be used for other applications (for example OH reactivity measurements, as was the case during this deployment (Stone et al., 2016)). As in previous configurations, the light exiting the fibre optic passed through a collimator coupled to a baffled entrance arm, this arrangement produced a beam profile of ~ 1 cm diameter which remained well collimated as it passed through both cells. A UV anti-reflective coated window was placed in the centre of the connecting arm to effectively seal the cells from each other. A further modification to the previously deployed configuration involved the coupling of a flow reactor to detection cell 2 to enable an RO₂ radical measurement. Further details on this approach are outlined below. Consequently, cell 1 was used for sequential measurements of OH and HO₂, with NO (BOC, 99.5%) injected into this cell during the second half of the online detection period.

2.3 RO_x-LIF description

An 83 cm long, 6.4 cm internal diameter flow reactor was coupled vertically to the second FAGE detection cell to facilitate detection of RO₂ radicals by LIF using the approach described by Fuchs et al. (2008). This flow reactor was held at approximately 30 Torr, with ~7.5 SLM ambient air drawn into the reactor via a 1 mm diameter pinhole. The flow reactor was operated in two modes. In the first, referred to as the HO_x mode, 250 sccm CO (BOC, 5% in N₂) was mixed with the ambient air close to the inlet to promote conversion of ambient OH to HO₂. In the second, referred to as the RO_x mode, 25 sccm of NO in N₂ (BOC, 500 ppmv) was also added to the CO flow which led to conversion of RO₂ to OH. The CO present rapidly reconverted any OH formed (or any OH sampled) to HO₂. Air (5 SLM) sampled by the flow reactor was transferred into the FAGE fluorescence detection cell (which was held at ~1.5 Torr) via a 4 mm diameter pinhole. 100 sccm NO (BOC, 99.5%) was injected into the fluorescence cell, converting HO₂ to OH for subsequent detection by LIF. In RO_x mode a measure of OH+HO₂+sum of RO₂ was obtained.

In laboratory tests, the relative sensitivity of the instrument to a range of different RO₂ species was investigated (see Table 1). Similar sensitivities were determined for the RO₂ species tested, therefore, we use the assumption that under ambient conditions individual RO₂ species are converted and, hence, detected with the same efficiency as methane-derived RO₂ radicals. The same assumption was drawn in the recent RO_x study in Wangdu, China (Tan et al., 2017). This assumption means that the concentration of RO₂ observed may be a lower estimate.

2.4 Calibration

The instrument was calibrated twice weekly on average using photolysis of a known concentration of water vapour at 185 nm within a turbulent flow tube to generate OH and HO₂, with the product of the photon flux at 185 nm and the water vapour photolysis time measured using a chemical actinometer (Commane et al., 2010). For RO₂, methane (BOC, CP grade, 99.5%) was added to the humidified air flow in sufficient quantity to rapidly convert OH to CH₃O₂. The limit of detection (LOD) at a signal-to-noise ratio of one for one data acquisition cycle lasting 7 minutes was ~4.5×10⁵ molecule cm⁻³ for OH, ~2.1×10⁶



molecule cm^{-3} for HO_2 and $\sim 6.9 \times 10^6$ molecule cm^{-3} for CH_3O_2 at a typical laser power of 13 mW in each cell. The measurements were recorded with 1 s time-resolution, and the accuracy of the measurements was $\sim 26\%$ (2σ).

2.5 Potential radical artefacts and corrections

2.5.1 OH

5 A small OH artefact signal (OH_{INT}) which derives from photolysis of O_3 by the 308 nm laser light, followed by the abstraction of an H atom from H_2O vapour within the FAGE cell, has been observed in laboratory tests. Such an artefact has been observed in other FAGE systems (Griffith et al., 2016; Fuchs et al., 2016; Tan et al., 2017), and although the reported magnitude of the interference is variable for different systems, the signal scales linearly with both O_3 and H_2O and displays a quadratic dependence with laser power. The following correction has been applied to the OH data presented here which corresponds to
10 5.2×10^5 molecule cm^{-3} of OH at 50 ppbv O_3 , 2% H_2O and 10 mW laser power (determined after the campaign but under the same experimental conditions):

$$\text{OH}_{\text{RAW CORR}} = \text{OH}_{\text{RAW OBS}} - \text{OH}_{\text{INT}} \quad (2)$$

where:

$$\text{OH}_{\text{INT}} (\text{molecule cm}^{-3}) = 520 (\pm 200) \times [\text{O}_3] (\text{ppbv}) \times [\text{H}_2\text{O}] (\%) \times \text{Laser power (mW)} \quad (3)$$

15 It should be noted that in later laboratory tests on the Leeds FAGE system with a modified nozzle design, the determined OH_{INT} was slightly lower than reported here. Fuchs et al. (2016) also report a variable artefact signal for the Jülich FAGE system. This variability introduces a high level of uncertainty into this correction. The OH_{INT} presented here should likely be considered an upper limit as any increase in the magnitude of this correction would lead to negative OH concentrations calculated during night-time periods.

20 Along with full characterisation of the O_3 - H_2O OH artefact signal, the Leeds FAGE system has subsequently been characterised with respect to other potential artefact signals, for example, an artefact deriving from reaction products of ozone and alkenes. Furthermore, in the most recent field campaigns, an inlet pre injector (IPI) has been used to chemically scavenge ambient OH, and provides an alternative method to determine background signals (to generate OH_{CHEM}) alongside the wavelength tuning approach discussed above (OH_{WAVE}). The laboratory interference tests and field comparison of OH_{CHEM}
25 and OH_{WAVE} in different environments will be subject of a future publication (Woodward-Massey et al., in preparation). In general, however, good agreement between OH_{CHEM} and OH_{WAVE} has been observed for the Leeds FAGE instrument (including during ambient measurements conducted in another urban environment in central Beijing with an OH_{WAVE} to OH_{CHEM} ratio of 1.04 and 1.07 in winter and summer respectively (Woodward-Massey et al., 2017)) and no significant artefact signal was observed in the interference tests conducted to date providing confidence in the OH measurements presented here.



2.5.2 HO₂ and RO_{2i}

Fuchs et al., (2011) and later Whalley et al., (2013) identified that specific RO₂ radical classes (primarily those derived from alkene and aromatic hydrocarbons, defined here as RO_{2i}) have the potential to decompose to OH in the presence of NO under typical FAGE cell conditions and, as a result, may be classed as an HO₂ interference. Depending on the type of FAGE cell and pressures employed, and NO concentration used, the level of interference can be deliberately varied (Whalley et al., 2013). During ClearfLo, two different NO concentrations (1.0 and 9.0×10¹³ molecule cm⁻³) were introduced into cell 1 to promote detection of (a) mainly HO₂ under low concentrations of added NO, and (b) HO₂ + RO_{2i} under high concentrations of added NO. With knowledge of the sensitivity to HO₂ and RO_{2i} at the two added NO concentrations, determined by adding a known concentration of HO₂ and ethene-derived RO₂ during calibration, and using the methodology outlined in Whalley et al., (2013), the concentration of RO_{2i} and interference-free HO₂ can be determined. In the following Results and Discussions we compare RO_{2i} derived from measurements using $\alpha_i = 0.72 \pm 0.09$ and $\alpha_i = 0.19 \pm 0.09$ at the high and low NO flow respectively to modelled RO_{2i}.

2.5.3 RO₂

Fuchs et al. (2008) described the potential of peroxy nitric acid and methyl peroxy nitric acid, HO₂NO₂ and CH₃O₂NO₂, (the concentration of which will be most elevated at high NO_x) to thermally decompose in the RO_x-LIF flow reactor. In this urban setting, the RO₂ signal that we attribute solely to non-interfering RO₂ species (RO_{2ni}) (determined by subtracting HO₂ + RO_{2i} measured in cell 1 from the total RO₂ signal measured by RO_xLIF in RO_x mode) may also include a contribution from CH₃O₂NO₂. Here, we refer to the measurement of non-interfering RO₂ species (RO_{2ni}) which includes a contribution from the thermal decomposition of CH₃O₂NO₂ as RO_{2ni}*. If the concentration of RO_{2ni} is dominated by CH₃O₂, it is possible to estimate the ambient concentration of CH₃O₂NO₂ from the radical measurements themselves and, thus, make a correction for this artefact without relying on model predictions of CH₃O₂NO₂. The methodology for this correction is outlined in the SI along with the RO_{2ni} data corrected for this potential artefact. Owing to the unknown fraction of the total RO_{2ni} that is CH₃O₂, we have left the data uncorrected in sections 3 and 4 below. It is worth noting, however, that this correction is most significant when NO concentrations peak during the morning; a time (as discussed in section 3.3.2) when the RO_{2ni} observations are under-estimated by model predictions.

2.6 Model description

A zero dimensional box model based on the Master Chemical Mechanism (MCMv3.2) (Jenkin et al., 2012) was used to predict radical concentrations for comparison with those observed. Complete details of the kinetic and photochemical data used in the mechanism are available at the MCM website (<http://mcm.leeds.ac.uk/MCM/home>). The model was run with a sub-set of the



MCM and treated the degradation of simultaneously measured trace VOCs, CH₄ and CO following oxidation by OH, O₃ and NO₃, and included ~15000 reactions and ~3800 species. The model was constrained by measurements of NO, NO₂, O₃, CO, CH₄, 62 individual VOC species measured by GC-FID and also 2D-GC PAN, HCHO, HNO₃, HONO, water vapour, temperature and pressure. The model was constrained with measured $j(\text{O}^1\text{D})$, $j(\text{NO}_2)$, $j(\text{HONO})$, $j(\text{HCHO})$, $j(\text{CH}_3\text{COCH}_3)$ and $j(\text{CH}_3\text{CHO})$ made using a spectral radiometer. For further instrumental details relating to all the model constraints please refer to Table 2 (and the references therein). For all other photo-labile species in the model, photolysis rates were scaled to the ratio of clear-sky $j(\text{O}^1\text{D})$, calculated using a two-stream isotropic scattering model (Hayman, 1997), to observed $j(\text{O}^1\text{D})$ to account for clouds. A constant H₂ concentration of 500 ppbv was assumed (Forster et al., 2012). The model inputs were updated every 15 minutes. For species measured more frequently, data were averaged to 15 minute intervals, whilst those measured at a lower time resolution were interpolated. The loss of all non-constrained, model generated, species by deposition or mixing was represented as a first order loss rate equivalent to 1 cm s⁻¹ in a boundary layer depth which varied from ~300 m during the night to 1800 m in the afternoon (estimated from vertical velocity variance (Barlow et al., 2015)) leading to lifetimes of ~8 hrs during the night and ~ 50 hrs during the afternoons.

The model was run for the entirety of the campaign in overlapping 7 day segments. To allow all the unmeasured, model-generated intermediate species time to reach steady state concentrations, the model was initialised with inputs from the first measurement day (23rd July) and spun-up for 5 days before comparison to measurements were made. Comparison of these 5 spin-up days demonstrated that the concentration of model generated species rapidly converged and there was less than a 1% difference in (for example) modelled OH concentration by the second spin up day. As a result of this, the model segments were run so as to overlap for 2 days only to reduce the computing time.

In all model scenarios a first of loss (k'_{loss}) for HO₂ was included to represent heterogeneous removal (Ravishankara, 1997):

$$k'_{loss} = \frac{\omega A \gamma}{4} \quad (4)$$

where ω is the mean molecular speed of HO₂ (equal to 43725 cm s⁻¹ at 298 K), γ is the aerosol uptake coefficient and A is the aerosol surface area density in cm² cm⁻³. A is calculated using data from an aerodynamic particle sizer instrument (TSI Inc, model 3321) which counts particles in 53 size bins ranging from 0.53 to 21.29 μm . For most of the scenarios considered γ was held constant at 0.1.

A series of distinct model scenarios were simulated to assess the sensitivity of the modelled radical concentrations to a number of model parameters. In the following results and discussions the radical measurements are compared (for the most part) to the base model scenario (MCM-BASE) which was run with the constraints outlined above.



3 Results

3.1 Radical observations and model predictions during the summer ClearfLo IOP

Near continuous radical measurements were made in London, from 23rd July to 17th August 2012. Typically, winds from the south west, ranging from less than 1 ms⁻¹ during the night to between 4 – 6 ms⁻¹ in the afternoon, were encountered. Close to the start of the campaign (24th – 27th July) and also later in the campaign (9th – 12th, 14th Aug), however, the wind direction switched to an easterly flow, bringing air that had passed over central London to the site, and wind speeds dropped. Fine weather prevailed during these easterly flows, with enhancements in air temperature and solar radiation (Fig. 1, top panel) observed. During these periods, radical concentrations (particularly the peroxy radicals) were elevated; the time-series of OH, HO₂, RO₂*i*, and the sum of RO₂ species (not including a HO₂ contribution) is presented in Figure 2. The concentration of a number of other species such as NO_x, CO and O₃ were also elevated (Fig. 1) during the easterly flows. Indeed, the concentration of the O₃ was observed to increase rapidly on the warmer days from sunrise, peaking during the afternoon at concentrations between 60 - 100 ppbv, and was found, on the 25th July, to exceed EU air quality recommendations of 60 ppbv for greater than a 6 hr period. The average diurnal profiles of the different radicals during south westerly and easterly flows are presented in Figure 3.

The short lifetime of OH (10 – 100 ms) measured directly in London from OH reactivity measurements (Whalley et al., 2016) dictates that OH exists in a photostationary steady state (PSS), where the rate of OH production is balanced by the rate of OH destruction (*f* is the fraction of O(¹D) that reacts with H₂O to form OH):

$$[\text{OH}]_{\text{PSS}} = \frac{\sum k \times [\text{OH}_{\text{source}}]}{\sum k \times [\text{OH}_{\text{sink}}]} = \frac{P_{\text{OH}}}{k_{\text{OH}}} \quad (5)$$

$$[\text{OH}]_{\text{PSS}} = \frac{\sum 2j(\text{O}^1\text{D})[\text{H}_2\text{O}][\text{O}_3]f + k_{\text{HO}_2+\text{NO}}[\text{HO}_2][\text{NO}] + j(\text{HONO})[\text{HONO}] + \text{ozonolysis}}{k_{\text{OH}}} \quad (6)$$

[OH]_{PSS} may be estimated from the rate of production from the sum of co-measured OH sources; here the rate of OH production from rate of reaction of HO₂ with NO, HONO photolysis, O₃ photolysis and the subsequent reaction of O¹D with H₂O vapour yielding two OH radicals, and ozonolysis is considered. The measured total OH reactivity (Whalley et al., 2016) which is representative of the sum of the concentration of all the individual OH sinks present multiplied by their bimolecular rate coefficients for reaction with OH is used as the denominator (E6).

The [OH]_{PSS} time-series is overlaid with the OH observations in Figures 2 and 3 and, on the whole is able to predict the observed [OH] reasonably well, particularly during the south-westerly flows. The campaign median ratio of the rate of OH production to the turnover rate of OH (*D*_{OH}), equal to the product of the total OH reactivity and the observed [OH] concentration, is close to 1 throughout the day (Fig. 4) highlighting consistency between the OH, HO₂ and OH reactivity observations. During the easterly conditions experienced at the beginning of the IOP, [OH]_{PSS} does under-predict the observed [OH] between 10 am – 6 pm (Fig. 2), however, suggesting that, if all the observed OH sources used in the PSS calculation are



correct, there may be a missing OH source under these conditions (discussed further below). The small OH interference deriving from the photolysis of O₃ within the FAGE cell (determined through laboratory tests) is corrected for in all OH data presented. Interestingly, when the wind direction switched for a second time to an easterly flow, [OH]_{PSS} reproduces the observed [OH] well (Fig. 2, lower panel).

5 A zero dimensional box model (MCM-BASE), which is run unconstrained to the radicals, but constrained to all other measured OH sources and constrained to the very detailed VOC observations, performs much better than [OH]_{PSS} during the first period of easterlies from late morning until late evening. However, under south-westerly conditions and also under the easterly conditions encountered during the second half of the campaign MCM-BASE over-predicts [OH] during the daytime by ~25% during the south-westerlies and by over a factor of 2 during the easterlies encountered at the end of the campaign. The box
10 model also has a tendency to under-predict the observed [OH] during the morning rush-hour (from dawn – 10 am) throughout the IOP. Since the model is able to reproduce the observed OH reactivity well during the easterly flows (Whalley et al., 2016), an under-estimation of the total sink term for OH is not the cause of the daytime (10am to 6pm) discrepancy. Rather, as can be seen in Fig. 2 and Fig. 3, this model significantly over-predicts the observed [HO₂] by close to a factor of 10 during the day under easterly conditions. The model over-estimates the total RO₂ concentration observed by close to a factor of two during
15 the easterly flows but predicts RO_{2*i*} well in this airmass. The modelled RO₂ over-estimate may either be due to the model over-estimating the sources of RO₂ or under estimating RO₂ sinks. Previous work (Whalley et al., 2016) highlighted that in general the model was able to capture the observed OH reactivity (*k*_{OH}) well during the easterly conditions encountered once the contribution to reactivity from mono-terpenes and the heavier-weight alkanes which derive from diesel emissions (Dunmore et al., 2015) was considered. This agreement between modelled and observed *k*_{OH} suggests that the production of RO₂ from
20 the oxidation of VOCs by OH should be reasonably well captured by the model and suggests that the model-measurement disagreement during the easterly flow may derive from an under-estimation of the RO₂ sinks. During the south-westerly flows, the model is able to capture the observed [RO₂] and [RO_{2*i*}] well during the afternoon on most days. However, the model under-predicts the observed [RO₂] throughout the morning hours and into the early afternoon (Fig. 3a). Our previous work highlighted that this model slightly under-predicted the observed OH reactivity (by ~25%) during south-westerly flows (Whalley et al.,
25 2016) and indicates, therefore, that a RO₂ source (from the oxidation of VOCs by OH) may be missing from the model under these conditions.

Despite the factor of 5 increase in modelled [HO₂] as the air-mass arriving at the site switched from south-westerly to easterly (the [HO₂] increase is driven to a large extent by the increase in [HCHO] a major source of HO₂ under easterly flows), [OH] observed (and modelled) increased by only ~35% on average. This demonstrates that the increase in OH sources was almost
30 entirely compensated for by an equivalent increase in OH sinks during these different flow regimes. The differences between [OH]_{PSS} and [OH]_{MCM-BASE} observed throughout the IOP reflects the impact of changing HO₂, the dominant OH source (see section 3.2) by an order of magnitude without changing the total OH sink term.



During 4 nights of the IOP, [RO₂] is predicted to be elevated, reaching concentrations of >1×10¹⁰ molecule cm⁻³ at 8 pm on 24th July. These high nighttime [RO₂] were not observed, to the magnitude predicted by the model, and other radical types (OH and HO₂) were not observed nor predicted to increase at the same time. These high modelled RO₂ excursions follow closely the modelled total alkyl nitrate profile and corresponds to evenings when NO and VOC concentrations were elevated.

5 The daytime observed and modelled radical profiles and the influence on in-situ ozone production is the primary focus of this paper and so nighttime modelled and observed radicals will not be considered further here other than to note that this model-measurement discrepancy may point towards a problem in the representation of the oxidation chemistry of the complex VOCs which were present at these times.

3.2 Model radical budget analysis

10 Figure 5 shows the main initiation, propagation and termination pathways in the model, and the hourly mean diel profiles of the modelled rates of primary radical initiation and termination are shown in Figure 6. Similar budget analyses have been conducted at other urban locations and may be compared and contrasted with the radical cycling here. During ClearfLo, the chain termination reaction of OH with NO₂, which leads to a net loss of radicals accounts for 24% of the modelled loss of OH between 6am – 9pm (21% for 11am-3pm if morning and evening rush-hours are excluded) (Whalley et al., 2016). For

15 comparison, this reaction contributed 20% to the total modelled OH loss in Los-Angeles during CALNEX (Griffith et al., 2016), 19% in Mexico City during MCMA-2006 (Dusanter et al., 2009), and just 11% in Birmingham during PUMA during the summer (11am-3pm) (Emmerson et al., 2005b).

In terms of total radical destruction reactions, OH+NO₂ accounts for 32% (red area, Fig. 6, lower panel), with net (forward – backward) RO_x + NO₂ to PAN species accounting for 35% (green area, Fig. 6, lower panel). As shown in Figure 6, the

20 termination of RO_x is dominated by loss of the OH radical (by reaction with NO₂) during the morning, whilst during the afternoon, radical termination is dominated by the loss of RO₂ species via PAN formation. In Birmingham, during the PUMA campaign (Emmerson et al., 2005b) net PAN formation reactions contributed close to 50% of the total radical destruction pathways reflecting the high OVOC fraction of total VOCs present, particularly aldehydes, in Birmingham. The photolysis of HONO is the dominant primary radical source in London, accounting for 40% of the total radical initiation steps between

25 11am -3pm (Lee et al., 2016); the photolysis of formaldehyde to yield two HO₂ radicals contributes 20%, whilst O(¹D)+H₂O contributes 12% and ozonolysis reactions, only 9%. HONO photolysis contributed a significant fraction to radical initiation in the MCMA-2006 and CALNEX studies also. In Birmingham the contribution of HONO photolysis as a primary radical source was likely under-estimated as HONO was not measured directly. In London, the model significantly under-estimated [HONO] if only gas-phase reactions were considered (Lee et al., 2016). Ozonolysis reactions were identified as the most important

30 primary source of radicals in Birmingham, with these reactions accounting for 25% of the radical initiation, which is much more significant than for London and highlights the very different VOC profile that exists in these two major UK cities. The PUMA campaign took place in Birmingham in the 2000 and so the difference in the VOC speciation may reflect, in part, the



change in VOC emissions in the UK over the past decade. As shown by Figure 6, blue area, ozonolysis reactions form an increasingly significant fraction of the radical initiation reactions during the afternoon hours and, along with VOC+NO₃ reactions, accounts for all the nighttime radical initiation reactions. Formaldehyde acts as a significant source of HO₂ radicals in London via photolysis and its reaction with OH. The latter, (OH to HO₂) propagation step (including OH+HCHO, but also OH+CO, OH+aromatics and OH+O₃) accounts for 27% of all the OH reactions in London. This OH to HO₂ propagation step, which is lower at other urban sites (19% and 20% during CALNEX and MCMA-2006 and 11% in Birmingham) contributes, in part, to the high modelled HO₂ concentration predicted for ClearLo.

3.3 Observed and modelled HO_x radical behaviour as a function of NO

As highlighted in Fig. 2 and 3, the degree of model-to-measured agreement varies depending on the chemical conditions encountered, which changed as a function of the wind direction and time of day. To gain further insight into chemical regimes under which model performance becomes compromised, the observed and modelled radical trends as a function of NO is considered and are shown in Fig. 7 and 8.

3.3.1 Low NO

This analysis highlights that under low [NO] conditions (< 1 ppbv) the median [OH]_{MCM-BASE} and [OH]_{obs} agree reasonably well (black and blue squares, Fig. 7, upper panel), whilst the [OH]_{PSS} (orange squares) under-estimate the observations by ~35% (and up to a factor of 3 during the first easterlies). Beyond 1 ppbv [NO], [OH]_{obs} and [OH]_{PSS} are in good agreement.

In several other urban studies, during which a range of NO_x conditions were encountered, a tendency to under-predict the observed [OH] at [NO] below 1 ppbv has been reported (Kanaya et al., 2007; Lu et al., 2012; Lu et al., 2013). As noted above in section 3.1, the differences in the MCM-BASE and PSS model predictions observed for ClearLo at low [NO] derives from the large over-estimation of HO₂ by the box model. If the box model is missing a large peroxy radical sink (discussed further in section 4 below) and, as such, a model constrained to the observed HO₂ provides a better representation of the OH sources, then, in agreement with the findings from Tokyo and China, an OH source important under low NO_x conditions must be missing from the model mechanism for London. In both central Tokyo (Kanaya et al., 2007) and Beijing (Lu et al., 2013) an observed-to-modelled OH ratio of ~ 2 – 3 was found at 200 pptv NO. At [NO] 0.2 – 1 ppbv, the median [OH]_{obs} to [OH]_{PSS} ratio was ~1.5 for ClearLo; the mean ratio was ~3 reflecting the larger difference between the observed OH and [OH]_{PSS} during the first easterlies. Lu et al., (2013) considered an additional recycling mechanism of HO₂ to OH by an unknown species and found that the rate of recycling required to reconcile the modelled and measured OH in Beijing was roughly half that required in during an earlier study conducted in the Pearl River Delta (Lu et al., 2012).

In contrast to the findings from these field observations made in China where [HO₂] modelled and observed were generally in reasonable agreement, the peroxy radical concentrations measured in London, particularly HO₂ concentrations, were greatly



over-estimated by MCM-BASE under lower NO_x conditions. It is interesting to note that HO_2^* which comprises HO_2 and a fraction of RO_2 radicals that rapidly decompose to HO_2 within low pressure FAGE cells (RO_2i) (Fuchs et al., 2011; Whalley et al., 2013) more closely follows the decrease in HO_2 predicted by the box model as a function of NO , than the interference-free HO_2 concentration that was observed (Fig. 7b). It is possible that previous HO_2 measurements made at urban sites which did not correct for the RO_2 artefact (Fuchs et al., 2011; Whalley et al., 2013) may have masked a problem with model predictions of HO_2 under lower NO conditions. Recent radical observations made in Wangdu, a rural site in the North China Plain, however, which corrected HO_2 for possible RO_2 interferences, did not highlight any model deviation from measured HO_2 under low NO conditions (Tan et al., 2017). The model over-estimation of HO_2 (and higher peroxy radicals) during ClearfLo, therefore, may be a reflection of the model's skill to predict radical propagation in the presence of the complex VOC mix which was observed in central London. The VOCs observed in London included a range of long-chain hydrocarbons deriving from diesel emissions as well as a range of monoterpene emissions from biogenic sources. This breakdown in model performance will be discussed further in section 4. As shown in Fig. 7b, the HO_2 and HO_2^* observations display the greatest deviation from each other at the lowest NO concentrations encountered and as NO concentrations increased the two measurements merged (i.e. HO_2 represented an increasing fraction of the HO_2^* signal). The lowest $[\text{NO}]$ tended to occur during the daytime after the morning rush-hour and so considering the diurnal profile of RO_2 radicals (i.e. peak concentrations during the day, low concentrations during the night and early morning) this trend is perhaps expected. There was variability in the $\text{HO}_2:\text{HO}_2^*$ ratio day-to-day with a smaller ratio observed under easterly conditions compared to south-westerly conditions. The similarity in the HO_2 and HO_2^* concentrations at high NO (Fig. 7b) demonstrates that the HO_2 artefact signal from RO_2 radicals is unlikely to contribute to any model measured discrepancies under high NO_x conditions as discussed below.

3.3.2 High NO

During ClearfLo, at $[\text{NO}] > 15$ ppbv, encountered primarily during the mornings, the modelled $[\text{OH}]_{\text{MCM-BASE}}$ underpredicted the observed $[\text{OH}]$. However, $[\text{OH}]_{\text{PSS}}$ was able to reproduce the OH measurements well (out to $[\text{NO}] = 25$ ppbv), as seen in Fig. 7. The under-prediction in OH by the MCM-BASE corresponds to an under-prediction in HO_2 between 15 – 30 ppbv NO. An under-prediction in HO_2 at elevated $[\text{NO}]$ has been highlighted during a number of earlier urban studies (Martinez et al., 2003; Ren et al., 2013; Brune et al., 2016). Brune et al. (2016) measured HO_2 concentrations that were a factor of ten greater than those predicted when NO concentrations reached 10 ppbv during the CalNex study which took place in Bakerfield, US. During ClearfLo, the modelled and observed HO_2 were in good agreement under NO concentrations ranging from 7 – 15 ppbv, but beyond 15 ppbv the model did begin to under-estimate the observations by approximately a factor of 3. It should be noted that the number of radical observations made under these elevated $[\text{NO}_x]$ were relatively few and, in fact at $[\text{NO}]$ concentrations greater than 30 ppbv, the model and observed HO_2 converge once more (Fig. 7). At high NO_x , Brune and co-workers report a measured OH production rate (P_{OH}) (determined by summing the rates of production from all measured OH sources) which was about twice the measured OH turnover rate (determined from the product of the total OH reactivity and observed $[\text{OH}]$) highlighting an inconsistency between the OH, HO_2 and OH reactivity observations. In contrast to this, as demonstrated by the



good agreement between the median observed OH production and OH loss rates during ClearfLo (Fig. 4), the $[\text{HO}_2]$ observed at the highest NO is supported by the observed $[\text{OH}]$ and OH reactivity.

The median modelled and measured total RO_2 and RO_{2i} trend as a function of NO are shown in Fig. 8. The model predicts RO_2 well at $[\text{NO}] < 1$ ppbv; the over-prediction of total RO_2 during the first easterlies does not bias the overall median model trend which, instead largely reflects the good agreement between modelled and measured RO_2 under the dominating south-westerly conditions at low $[\text{NO}]$. In contrast to the reasonable agreement between modelled and observed HO_2 at high $[\text{NO}]$, the model increasingly under-predicts the total RO_2 concentration (particularly RO_{2ni}) at $[\text{NO}]$ beyond ~ 3 ppbv. As highlighted in Fig S3 in the SI, applying a correction to the RO_2 data to account for the possible decomposition of $\text{CH}_3\text{O}_2\text{NO}_2$ with the RO_xLIF flow reactor, leads to an improved agreement between the modelled and observations for RO_2 under high NO conditions although the extent to which $\text{CH}_3\text{O}_2\text{NO}_2$ decomposes within the flow reactor is highly uncertain.

4 Discussion

4.1 Possible explanations for the differences between observed and modelled peroxy radical concentrations at low NO

A number of possible explanations for the differences between the observed and modelled peroxy radical concentrations under the low NO conditions have been explored through a series of model scenarios (detailed below and also in the SI). The impact of $[\text{NO}_x]$ deviations from a photo-stationary steady state in the real atmosphere as well as under-estimating the heterogeneous loss of HO_2 to aerosol surfaces are discussed in the SI and model runs highlighting the sensitivity of the modelled radical concentrations to these parameters are presented in Fig. S4. Enhancing the rate of HO_2 termination in the model, e.g. by enhancing the uptake probability of HO_2 to aerosols only improves the HO_2 modelled to measured agreement by a modest amount, and so, given the dominant reactions involving HO_2 are radical propagating, with the reaction of $\text{RO}_2 + \text{NO}$ acting as the largest source of HO_2 and the reaction of HO_2 with NO (recycling OH) acting as the dominant HO_2 sink (Fig. 5), this raises the question whether the model discrepancy relates to uncertainties in the RO_2 oxidation chemistry and the cycling of RO_2 to HO_2 . Of particular relevance are the reactions involving the complex RO_2 species deriving from VOCs emitted from diesel and biogenic sources.

4.1.1 Uncertainties in the model chemistry under low NO conditions

Hydrocarbon autoxidation processes which are known to readily occur in the liquid phase (Bolland, 1949) were, until recently, thought to be unimportant in the gas-phase owing to the low probability of intermolecular H-atom abstraction. The low probability is due to the low concentration of hydrocarbons in the atmosphere and the competition between intra-molecular H-shift reactions (from a C-H to an R-O-O bond forming a peroxide) and bimolecular reactions of the RO_2 radical (e.g. with NO, HO_2 or RO_2). There is increasing evidence, however, that autoxidation processes are occurring in the atmosphere, which can quickly lower the volatility of VOCs and promote SOA formation. Laboratory studies have shown that monoterpenes, α -pinene and limonene, following initial attack by ozone or OH form highly oxidised RO_2 radicals within a few seconds via repeated



H-shift from C-H to an R-O-O bond and subsequent O₂ additions (Crouse et al., 2011; Jokinen et al., 2014; Ehn et al., 2014; Berndt et al., 2016). Mass spectrometric signals relating to these highly oxidised RO₂ species have also been observed during field measurements (Jokinen et al., 2014). Autoxidation processes could be relevant during ClearLo and omission of these processes in the model mechanism could account for some of the radical over-prediction observed under the lower NO conditions, particularly under easterly flow conditions when elevated concentrations of monoterpenes were observed. At high NO, bimolecular RO₂+NO reactions likely out-compete intramolecular processes. Importantly here, autoxidation steps which involve intramolecular H-atom abstraction from a C-H to an O-O bond and subsequent addition of O₂ to reform a more oxidised RO₂ radical do not generate HO₂. Jokinen et al., (2014) observed a high formation rate of organic nitrates (of the order of 30%) when NO was added to experiments which would serve to further decrease RO₂ to HO₂ propagation. For ClearLo conditions, a model run unconstrained to the mono-terpenes and the heavier-weight alkanes (MCM-VOC-STANDARD) under-estimated OH reactivity (Whalley et al., 2016) with the missing OH reactivity fraction largely reconciled by the model-generated intermediates which derive from alpha-pinene and limonene. If the current oxidation mechanism for these species is inaccurate, the reactivity attributed to these oxidation products could be wrong and instead may derive from other oxidised species. The missing reactivity in the MCM-VOC-STANDARD run can be included by adding a single OH to RO₂ conversion to the model equivalent to the missing reactivity (in s⁻¹) at each time stamp. To represent an autoxidation pathway, we convert OH to MCM species C6H5O2. This RO₂ species is formed via a minor phenol + OH channel. C6H5O2 does not readily convert to HO₂ by reaction with NO (due to the lack of available H on the alpha C) and, instead, following oxygen atom abstraction, C6H5O reacts with NO₂ to form a nitro-phenol, or reacts with ozone to reform C6H5O2; which reacts with further NO and so on. We do not consider the reactions of C6H5O2 to be representative of what is actually occurring, but rather choose this species to represent a mechanism by which the propagation of RO₂ to HO₂ is inhibited. Inclusion of OH→C6H5O2 leads to a ~30% decrease in modelled HO₂ (see pale blue vs black diel profiles in Fig. 9) and close to a 50% decrease in modelled HO₂ if the heterogeneous loss to aerosol is enhanced also by increasing the HO₂ uptake probability to aerosols from 0.1 to 1 (purple vs black diel profiles in Fig. 9). Including autoxidation in the model improves the agreement with the observations during the daytime for all radical species apart from the total RO₂ species observed during south westerly flows. As discussed in section 3.1, however, the model is, likely, missing VOCs under this air-mass regime (implied from the under-prediction of the observed *k*_{OH}) and this may contribute to the model under-prediction of RO₂. The under-prediction of RO_{2*j*} when an autoxidation step is included is due to the choice of OH→RO₂ conversion species (i.e. C6H5O2 is an RO₂ species that does not decompose to HO₂ in the presence of NO within a FAGE cell). The model under-prediction of RO_{2*j*} suggests that at least some of the RO₂ species that undergo autoxidation are species that would decompose to HO₂ within a FAGE cell.

The partitioning of larger, lower volatility OVOCs to the aerosol-phase may be another important step which is not included in the model. The partitioning of gases to the aerosol-phase will reduce RO₂ to HO₂ propagation and act as a net radical sink and omitting this process may further contribute to some of the discrepancy between observed radical concentrations and MCM-BASE predictions.



As depicted by the model radical flux (Fig. 5) roughly half (162×10^5 molecule $\text{cm}^{-3} \text{s}^{-1}$ of 314×10^5 molecule $\text{cm}^{-3} \text{s}^{-1}$ between 11 am – 3 pm) of the modelled RO_2 radicals that react with NO eventually form HO_2 (via an alkoxy radical, RO). If, however, RO_2 to HO_2 propagation is over-estimated by the model due to the uncertainties outline above, modelled HO_2 may become artificially high. A steady state $[\text{HO}_2]$ can be estimated (without the use of a model) by balancing the dominant HO_2 production and destruction reactions (first and second order loss processes) that occur:

$$\begin{aligned}
 & k_{\text{CO}+\text{OH}}[\text{CO}][\text{OH}] + 2 \times j(\text{HCHO}_{\text{radical channel}})[\text{HCHO}] + (\alpha \times k_{\text{RO}_2+\text{NO}}[\text{RO}_2][\text{NO}]) \\
 & = k_{\text{HO}_2+\text{HO}_2}[\text{HO}_2]^2 + k_{\text{HO}_2+\text{NO}}[\text{NO}][\text{HO}_2] + k_{\text{HO}_2+\text{RO}_2}[\text{RO}_2][\text{HO}_2] + k_{\text{HO}_2+\text{O}_3}[\text{O}_3][\text{HO}_2] + k_{\text{Loss to Aerosols}}[\text{HO}_2] \quad (7)
 \end{aligned}$$

α is equal to the fraction of RO_2 radicals which propagate to HO_2 (which is roughly half in MCM-BASE).

Eqn. 16 can be rewritten as a quadratic equation for HO_2 and then solved for HO_2 to yield the following solution:

$$[\text{HO}_2] = \frac{-b + \sqrt{(b^2 - 4ac)}}{2a} \quad (8)$$

where

$$a = 2 \times k_{\text{HO}_2+\text{HO}_2} \quad (9)$$

$$b = k_{\text{HO}_2+\text{NO}}[\text{NO}] + k_{\text{HO}_2+\text{RO}_2}[\text{RO}_2] + k_{\text{HO}_2+\text{O}_3}[\text{O}_3] + k_{\text{Loss to Aerosols}} \quad (10)$$

$$c = k_{\text{CO}+\text{OH}}[\text{CO}][\text{OH}] + 2 \times j(\text{HCHO}_{\text{radical channel}})[\text{HCHO}] + (\alpha \times k_{\text{RO}_2+\text{NO}}[\text{RO}_2][\text{NO}]) \quad (11)$$

Using the observed RO_2 and OH concentrations in equations E8 – E11 above to calculate $[\text{HO}_2]$, good agreement between HO_2 observed and HO_2 calculated can be achieved if α equal to 0.15 is assumed as shown in Figure 10. The value for α , however, is sensitive to the rate coefficient, $k_{\text{RO}_2+\text{NO}}$ used with the $[\text{HO}_2]$ in Fig. 10 calculated using $k_{\text{RO}_2+\text{NO}} = k_{\text{CH}_3\text{O}_2+\text{NO}}$ ($= 7.7 \times 10^{-12} \text{ cm}^3 \text{ molecule}^{-1} \text{ s}^{-1}$ at 298K). If some of the RO_2 species contributing to the total RO_2 measured react faster with NO (as is the case for $\text{CH}_3\text{CO}\cdot\text{O}_2$ radicals, $k_{\text{RO}_2+\text{NO}} = 2 \times 10^{-11} \text{ cm}^3 \text{ molecule}^{-1} \text{ s}^{-1}$ at 298K), α would become < 0.15 . This low fraction of RO_2 to HO_2 conversion (if this is the cause for the observed and modelled discrepancy) compared to $\alpha \sim 0.5$ in MCM-BASE highlights a significant misunderstanding in the oxidation chemistry mechanism of the larger more complex VOCs. This misunderstanding likely becomes increasingly important in low NO, high VOC environments such as forests.

4.2 Impact of the model uncertainties on predictions of *in situ* ozone production

Poor representation of the observed peroxy radical concentration leads to significantly more ozone production predicted by the model than is calculated from the observed concentrations under low NO conditions (Fig. 11) using E1 (which is repeated



below for clarity). Conversely, significantly less ozone production is predicted by the modelled peroxy radicals than by the observed peroxy radicals as [NO] increases.

$$P(O_3) = (k_{HO_2+NO}[HO_2][NO] + k_{RO_2+NO}[RO_2][NO]) - (k_{OH+NO_2+M}[OH][NO_2][M] + k_{RO_2+NO_2+M}[RO_2][NO_2][M] + \text{deposition}) \quad (1)$$

- 5 As highlighted in section 3.3, the model's failure to predict the observed HO₂ radical concentrations is most evident under low NO_x conditions, typically experienced during the afternoon hours and particularly during easterly flows. At this time the observed ozone concentrations peaked (Fig. 11) due to reduced destruction by titration with NO. At NO concentrations <3 ppbv, the ozone production rate determined from the modelled peroxy radical concentrations remains relatively constant at ~3 ppbv hr⁻¹ until very low levels of NO. P(O₃) calculated with the observed peroxy radical, however, decreases to ~1 ppbv hr⁻¹.
- 10 Under higher NO_x conditions, [NO]>3 ppbv, the ozone production rate determined from the modelled peroxy radical concentrations is, up to, an order of magnitude lower than the ozone production rate calculated from the observations (which at the highest [NO] reaches ~30 ppbv hr⁻¹). The calculation of ozone production from many earlier urban studies often relied on an inferred RO₂ concentration, estimated from the measured [HO₂] and assumed value of RO₂:HO₂, as measurements of total RO₂ were not available (e.g. (Ren et al., 2013; Brune et al., 2016)). In these studies, under high NO conditions, the P(O₃) calculated from the observed HO₂ (and inferred RO₂) was significantly greater than P(O₃) calculated using the modelled HO₂ and RO₂, reflecting the model under-estimation of HO₂ at high NO reported from these studies. In the recent Wangdu study conducted in China, Tan et al., (2017), using observed RO₂, demonstrated that models may under-predict ozone production at high NO due to an underestimation of the RO₂ radical concentration rather than under-estimation of HO₂. In the Wangdu study modelled and measured HO₂ were in good agreement at high NO. From the rate of ozone production calculated from the modelled and measured peroxy radicals for ClearfLo, we would draw similar conclusions as drawn by Tan and co-workers, i.e. that there is missing RO₂ at high NO, if the correction for decomposition of CH₃O₂NO₂ was not applicable. Although there are some uncertainties surrounding the magnitude of CH₃O₂NO₂ decomposition in the RO_xLIF cell (which is experimentally difficult to determine), in agreement with Tan et al. we find no evidence from the ClearfLo HO₂ dataset that there is a significant model bias for HO₂ which influences the model predicted P(O₃) under the elevated NO conditions encountered.
- 25 The discrepancy between model and observations at low NO may arise from the model uncertainties in the treatment of the oxidation and removal of the complex VOC species observed as discussed above. The oxidation of these complex species tend not to be included in air quality models used to predict ozone and other secondary pollutants. The MCM, however, is used as the benchmark mechanism against which simpler mechanisms used within air quality models are tested (Malkin et al., 2016) and so the chemistry of these complex VOCs present in the urban atmosphere and the impact they have on peroxy radical concentrations needs to be adequately resolved.
- 30



5 Conclusions

Measurement and model comparisons of OH, HO₂, RO₂i and total RO₂, have displayed varying levels of agreement as a function of NO_x. Under higher NO_x conditions the box model increasingly under-predicted total [RO₂] and, as a consequence, ozone production derived from the predicted peroxy radicals is up to an order of magnitude lower than from the observed peroxy radicals.

A large uncertainty in peroxy radical cycling, RO₂→HO₂, has been identified under lower NO conditions experienced during the daytime. We hypothesise that uncertainties in the degradation mechanism of RO₂ deriving from complex biogenic and diesel related VOC, species which were particularly elevated and dominated the OH reactivity under easterly flows when the model measurement discrepancy was largest, may account for the model measurement disagreement. Autoxidation processes now known to play a role in the chemical oxidation of mono-terpenes in the gas-phase, and which can enhance SOA formation, may serve to reduce the rate of RO₂ to HO₂ propagation under lower NO conditions. Omission of this oxidation process from the model mechanism leads to more ozone production predicted using modelled peroxy radical concentrations versus those measured, at a time when ozone destruction (by NO titration) is slow. Although air quality models do not typically consider these VOC types and tend to run with simplified chemistry schemes, the MCM is viewed as a benchmark mechanism against which these simpler chemistry schemes may be tested. Hence, these uncertainties in the mechanism identified here need to be critically assessed through further laboratory and field measurements.

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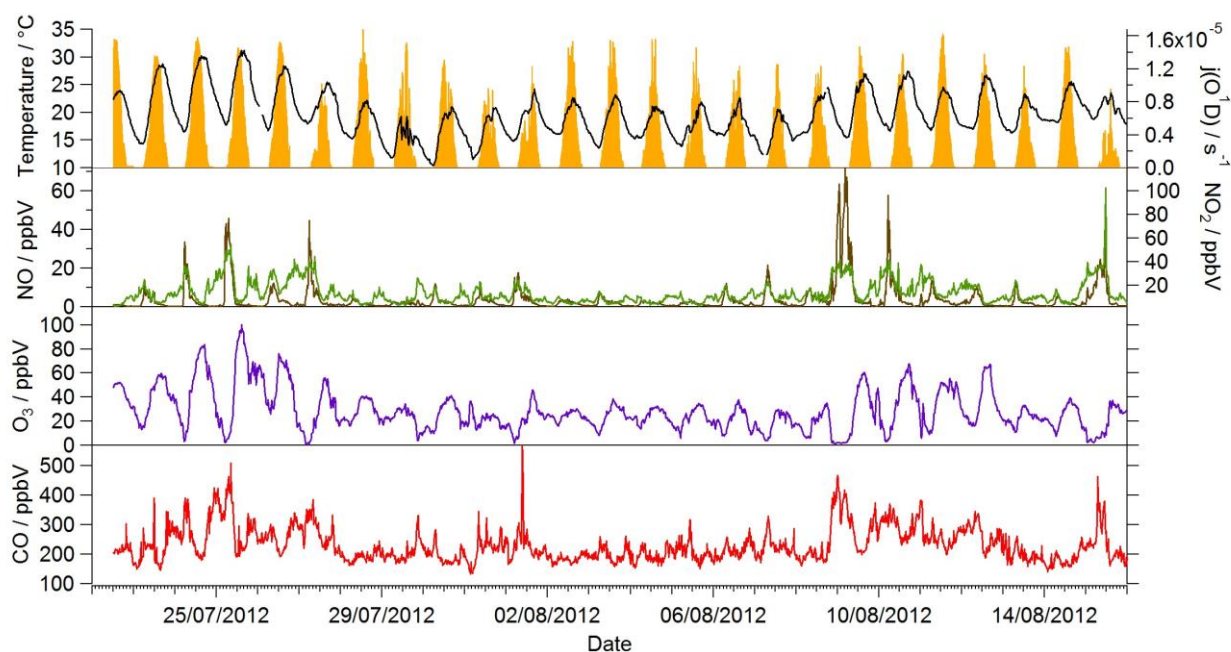


Figure 1: Observed temperature (black line), $j(\text{O}^1\text{D})$ (orange area), NO (brown line), NO₂ (green line), O₃ (purple line) and CO (red line) mixing ratios during the summer ClearLo IOP. Data time resolution is 15 minutes.

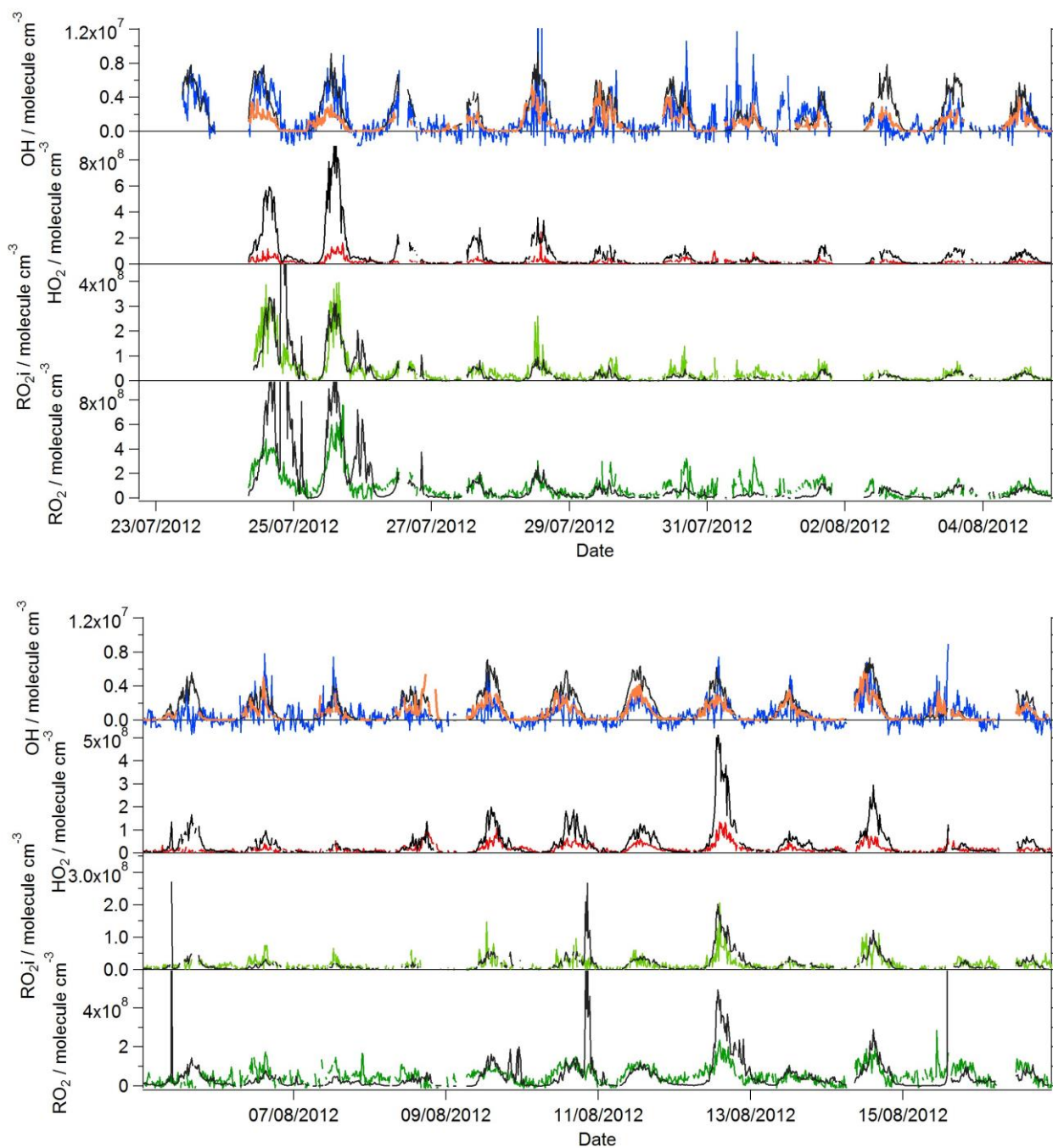


Figure 2: Observed (coloured lines) and MCM-BASE modelled (black lines) OH, HO₂, RO_{2i} and RO₂ during the summer ClearfLo IOP; steady state [OH] ([OH]_{PSS}) is displayed by the orange line.

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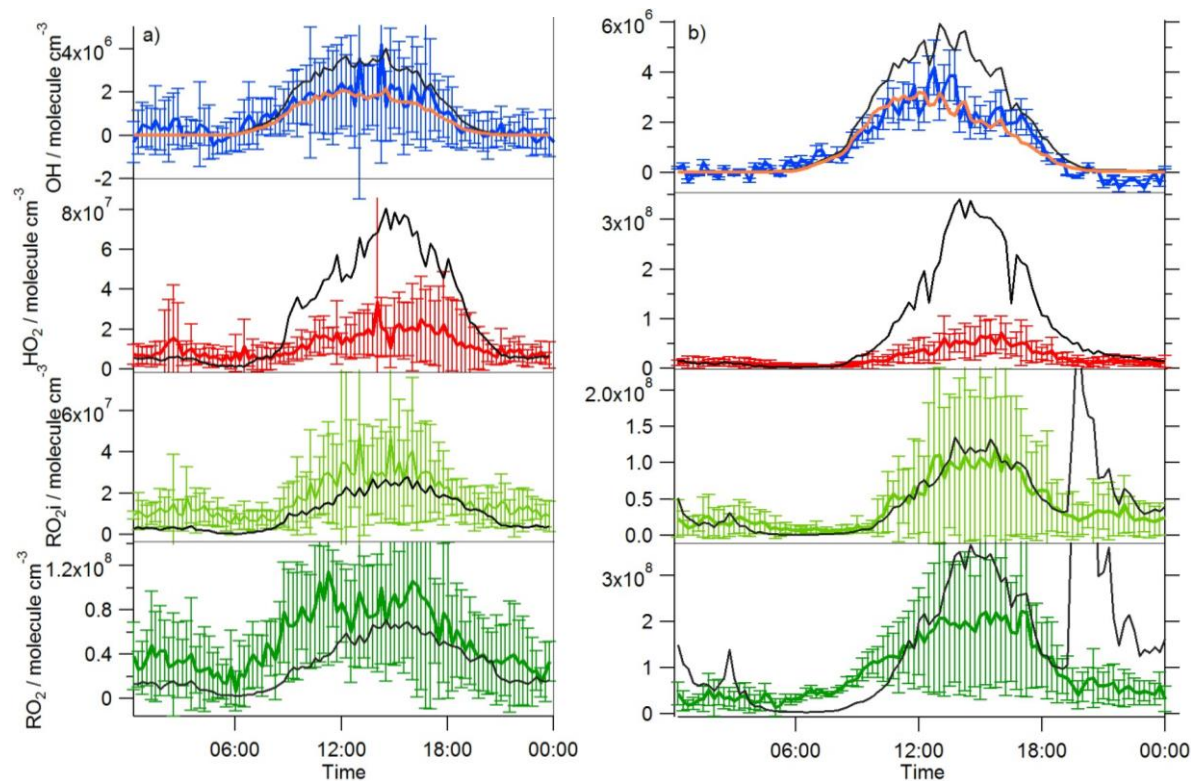
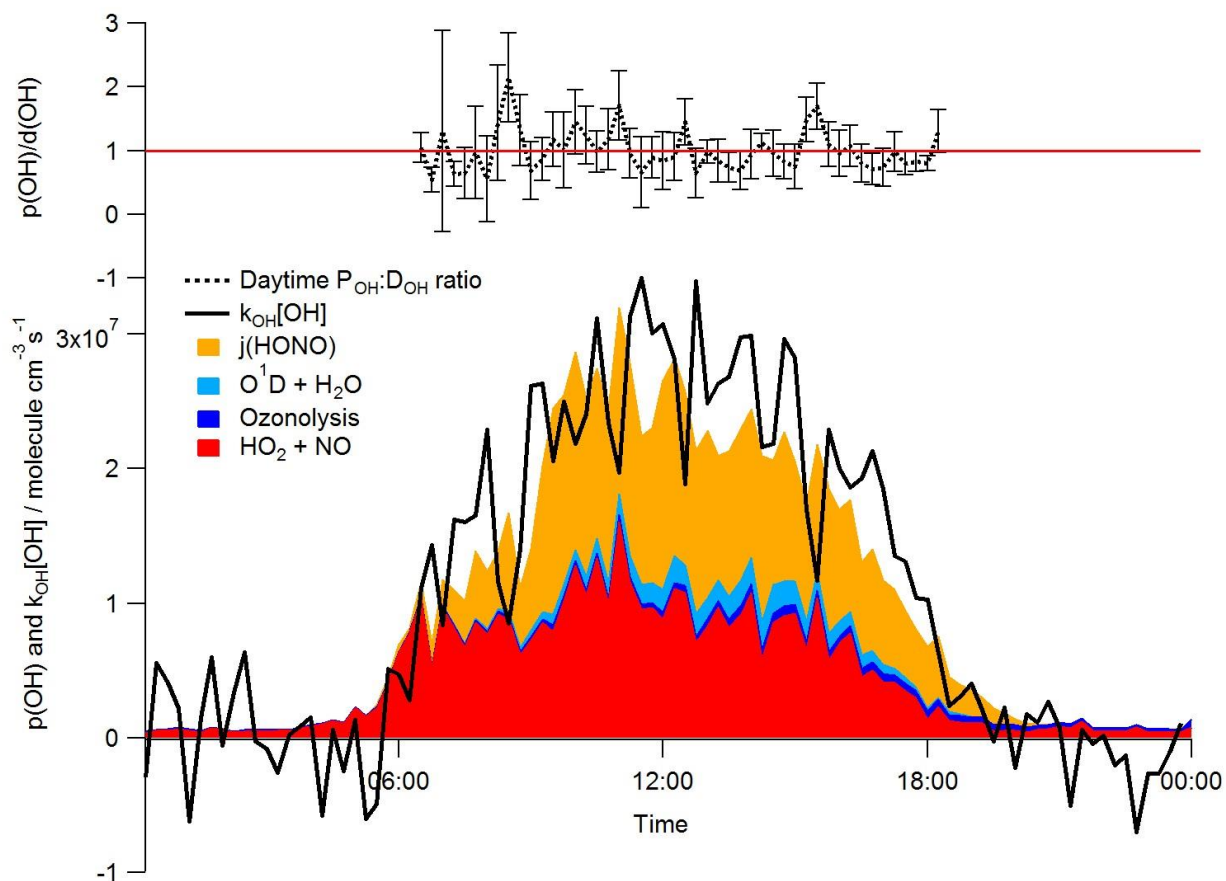


Figure 3: Average diel observed (coloured lines with error bars) and MCM-BASE (black line) OH, HO₂, RO_{2i} and RO₂ profiles during a) south-westerly and b) easterly flows; [OH]_{ps} is displayed by the orange line. The error bars represent the 1 σ variability in the observations.



5 **Figure 4: Median diurnal profiles for the whole campaign of the observed $D_{OH} = k_{OH} \times [OH]$. The summed rate of production of OH (P_{OH}) from the photolysis of HONO, the reaction of $O(^1D)$ with H_2O , ozonolysis reactions and the reaction of HO_2 with NO is overlaid. The dashed black line represents the median daytime (06:30 – 18:30) $P_{OH} : D_{OH}$ ratio; error bars highlight the 1σ standard deviation of this ratio. The red line represents a ratio of 1.**

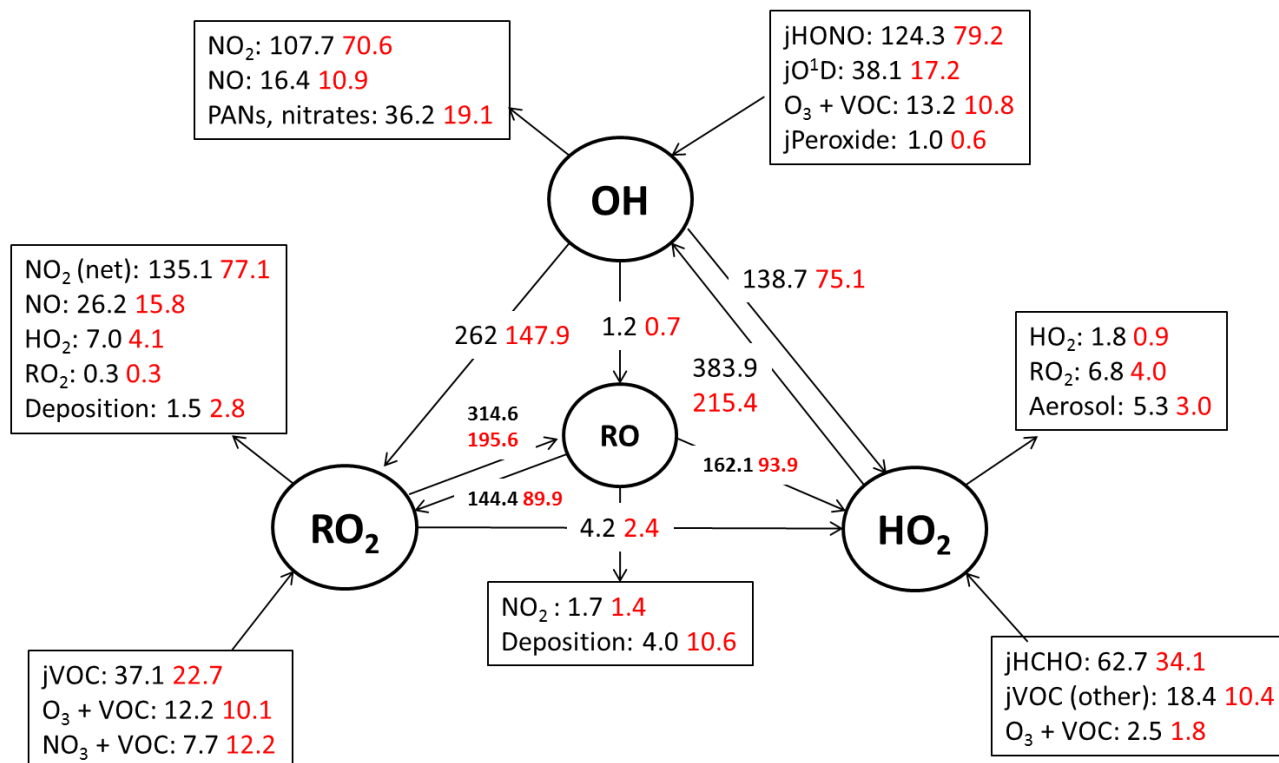


Figure 5: Mean daytime (11am – 3pm, black number and 6am – 9pm, red number) rates of reaction for formation, propagation and termination of radicals in units of $10^5 \text{ molecule cm}^{-3} \text{ s}^{-1}$ for the whole campaign period.

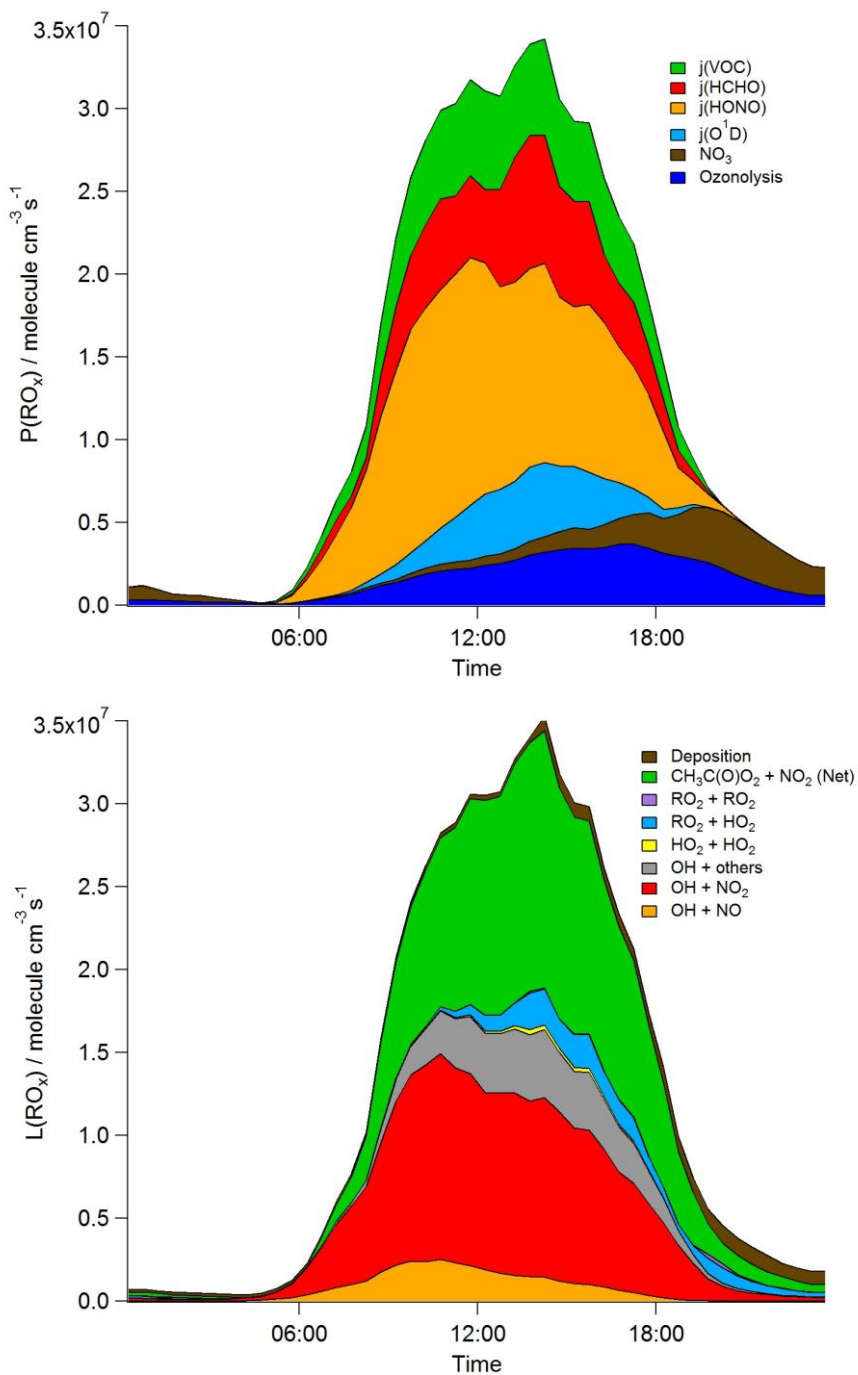
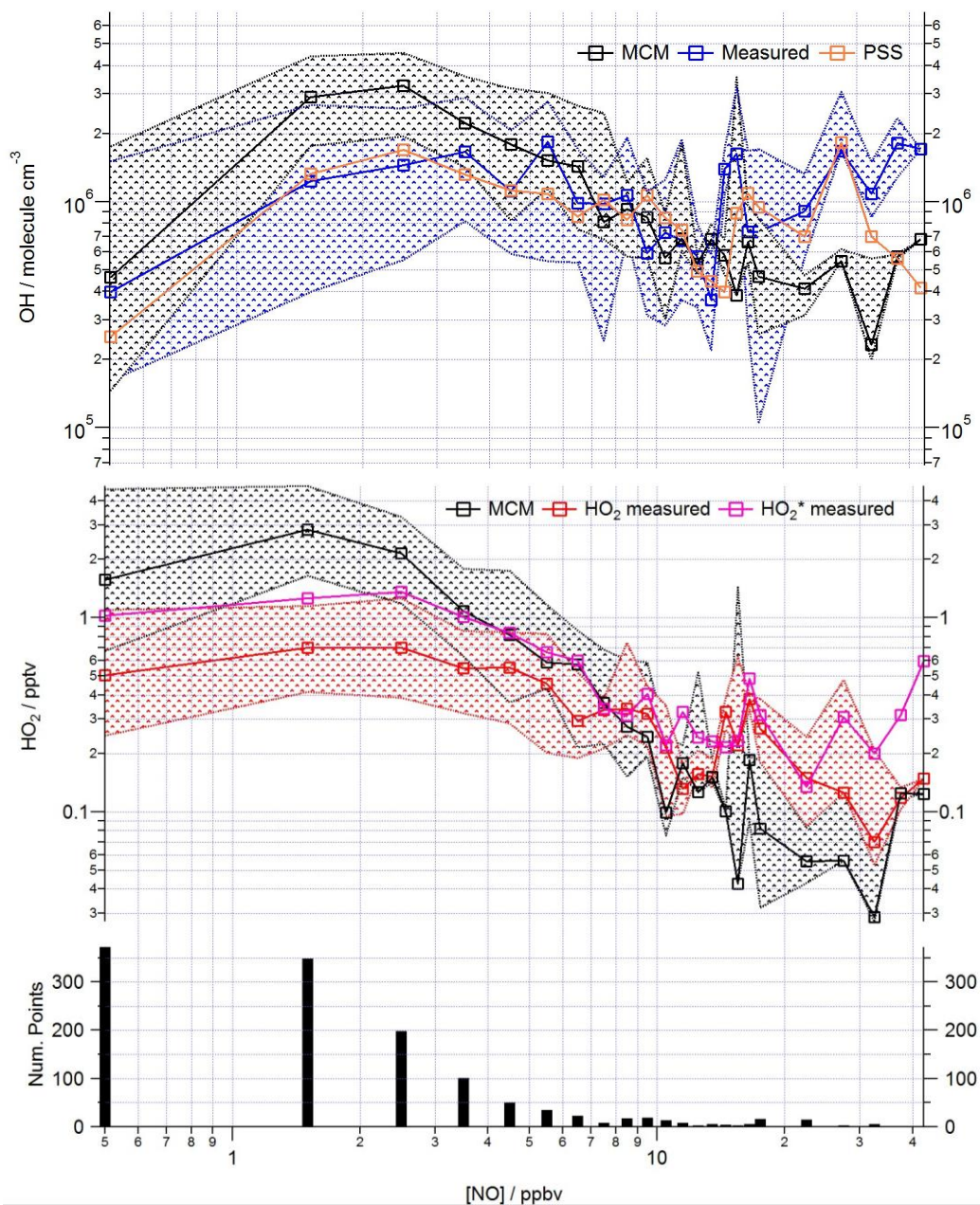


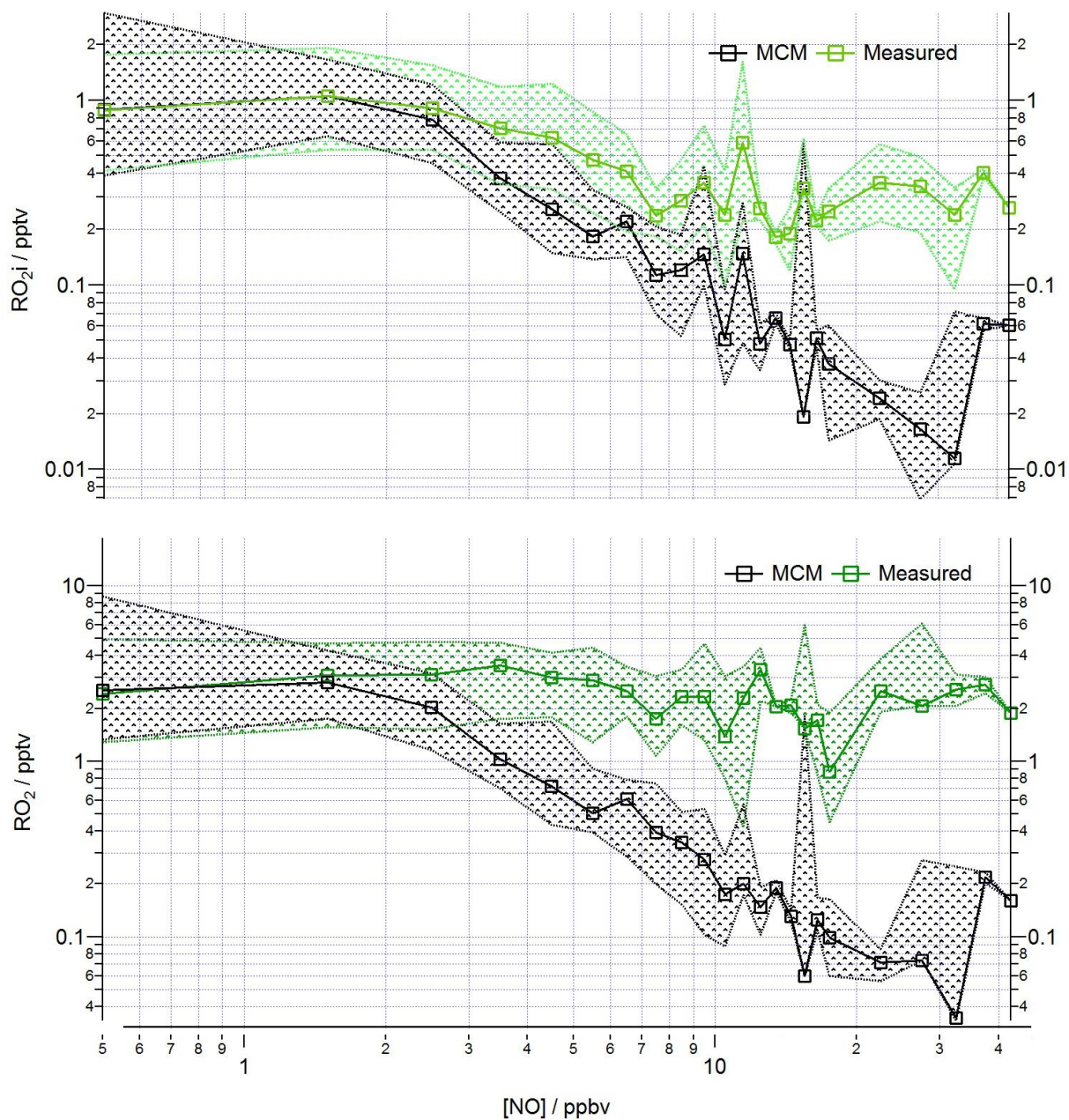
Figure 6: Mean diurnal profiles of MCM-BASE modelled rates of RO_x initiation (upper panel) and termination (lower panel) reactions for the whole campaign period. ‘ $\text{CH}_3\text{C}(\text{O})\text{O}_2 + \text{NO}_2$ (Net)’ represents the net (forward - backward) $\text{CH}_3\text{C}(\text{O})\text{O}_2 + \text{NO}_2 \leftrightarrow$ PAN species.

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Figure 7: Observed and modelled HO_x behaviour as a function of NO for the whole campaign period. (a) Median OH measured (blue squares), OH modelled (base MCM model = black squares, steady state calculation = orange squares). (b) Median HO₂ measured (red squares), HO₂* measured (pink squares), HO₂ modelled (black squares). Patterned areas represents the 25/75th percentiles. Data are filtered for daytime hours between 6 am and 7 pm and binned by [NO] with a bin width = 1 ppbv for [NO] between 0 – 20 ppbv and bin width = 5 ppbv for [NO] between 20 – 45 ppbv. The number of points in each bin is displayed in the lower panel.



5 **Figure 8:** Observed and modelled RO_x behaviour as a function of NO for the whole campaign period. (a) Median RO_{2i} measured (green squares), RO_{2i} modelled (black squares). (b) Median RO₂ measured (green squares), RO₂ modelled (black squares). Patterned areas represents the 25/75th percentiles. Data are filtered for daytime hours between 6 am and 7 pm and binned by [NO] with a bin width = 1 ppbv for [NO] between 0 – 20 ppbv and bin width = 5 ppbv for [NO] between 20 – 45 ppbv.

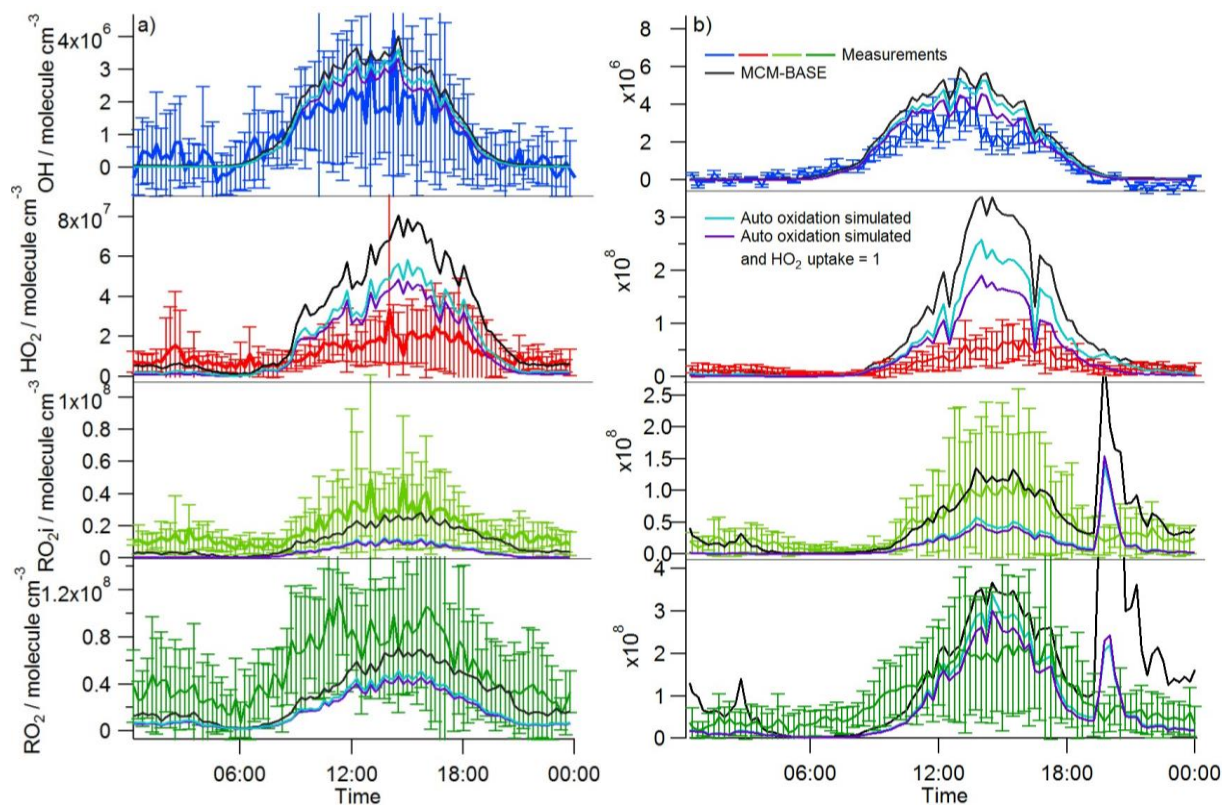


Figure 9: Average diel observed and modelled HO_x profiles during a) south-westerly and b) easterly flows. The base model predictions are represented by the black line. The model scenario run with standard VOC species only and missing reactivity represented by converting OH to $\text{C}_6\text{H}_5\text{O}_2$ is represented by the pale blue line; the purple line represents the model scenario run with standard VOC species only and missing reactivity represented by converting OH to $\text{C}_6\text{H}_5\text{O}_2$ and an enhanced heterogeneous loss of HO_2 , $\gamma_{\text{HO}_2} = 1$.

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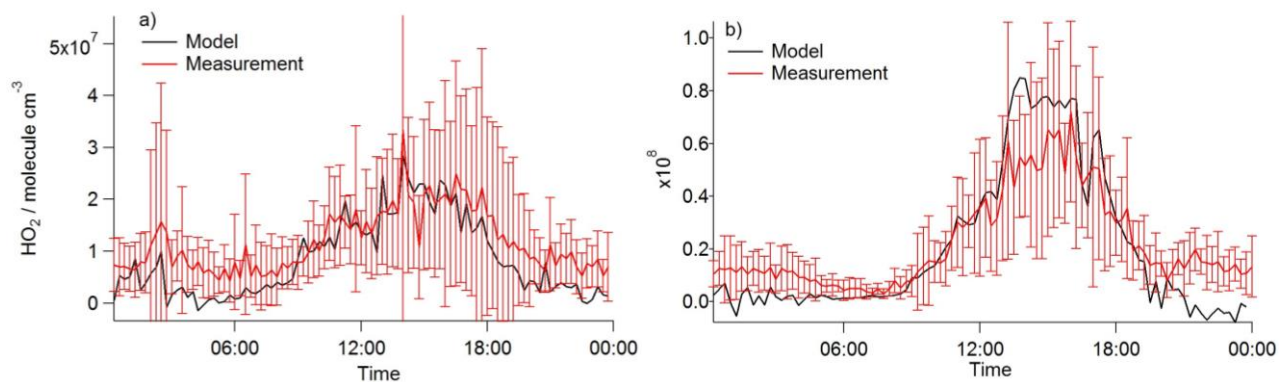


Figure 10: Average diel profiles of HO₂ concentration observed (red line with error bars) under a) south-westerly and b) easterly conditions. Overlaid is HO₂ calculated using the solution to the HO₂ quadratic expression (E7) (represented by the black line) with α equal to 0.15.

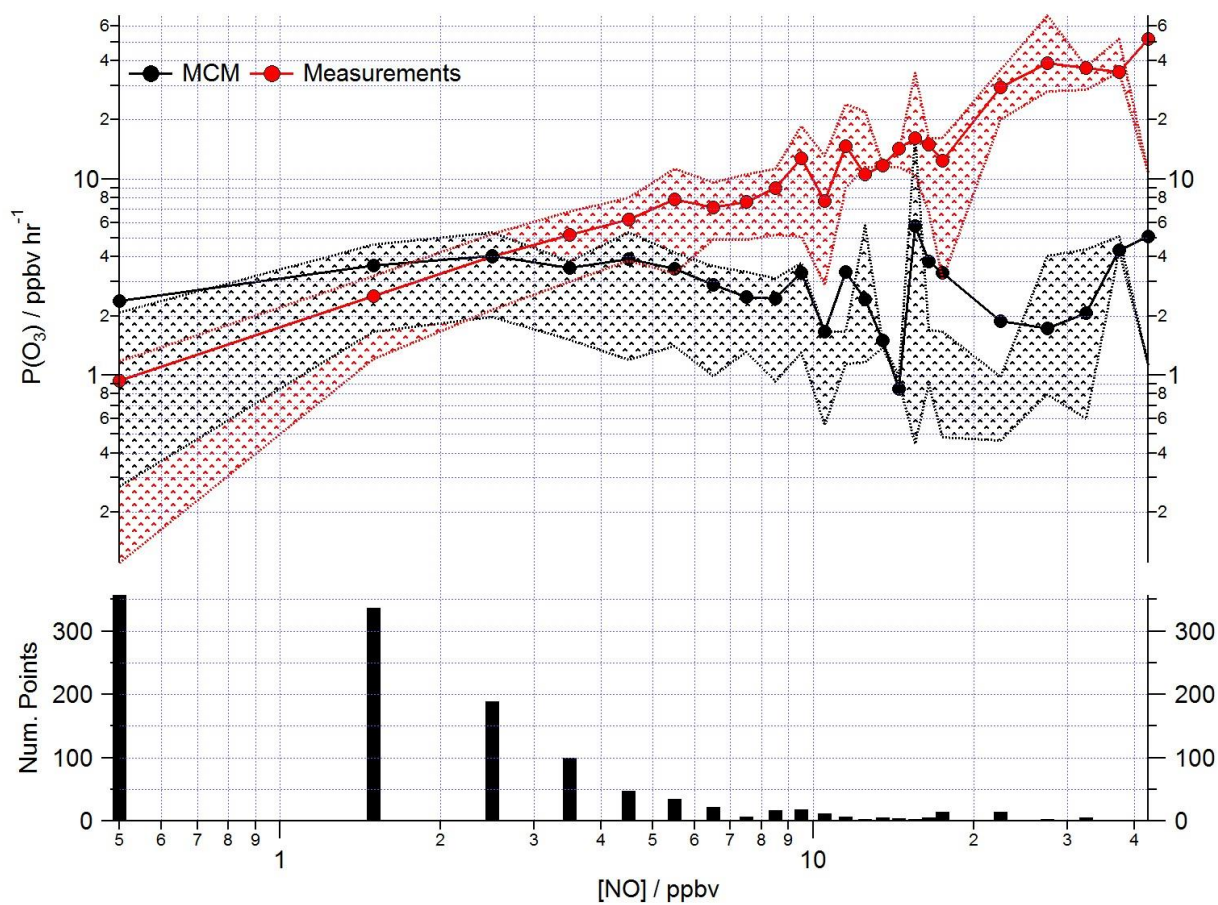


Figure 11: Mean ozone production ($ppbv\ hr^{-1}$) calculated from observed (red squares) and modelled (black squares) RO_x concentrations using E1 as a function of NO. Data are filtered for daytime hours between 6 am and 7 pm and binned by [NO] with a bin width = 1 ppbv for [NO] between 0 – 20 ppbv and bin width = 5 ppbv for [NO] between 20 – 45 ppbv.

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Table 1: Relative efficiency of RO₂ to HO_x conversion for different RO₂ species

Hydrocarbon	RO ₂ relative sensitivity
Methane	1.0±0.03
Isoprene	1.0±0.05
Ethene	0.94±0.04
Toluene	0.88±0.05
Butane	0.78±0.03
Cyclohexane	0.79±0.02

**Table 2: Listing of the concurrent measurements made during ClearfLo**

Measurement	Instrument	Technique	LOD	Reference
O ₃	Thermo 49i series	UV Absorption	0.05 ppbv	
CO	Aerolaser 5002	VUV-fluorimetry	1 ppbv	(Gerbig et al., 1999)
NO, NO ₂	Air Quality Design Inc.	Chemiluminescence with LED NO ₂ converter	1.8 pptv (NO), 5.5 pptv (NO ₂)	(Lee et al., 2009)
HONO	LOPAP	Long-path absorption photometry	3 pptv (4 min)	(Heland et al., 2001)
PAN	GC-ECD	Gas chromatography with electron capture detection	5 pptv (90 s)	(Whalley et al., 2004)
HCHO	Aerolaser 4021 analyser	Hantzsch reaction	<0.05 ppbv	(Salmon et al., 2008)
Actinic flux	Ocean optics QE65000	Spectrometer coupled to 2π quartz collection dome	-	
j(O ¹ D)	Meteorologie Consult	Filter radiometry	-	(Bohn et al., 2016)
C1-C8 hydrocarbons	(DC)-GC-FID	Dual-channel gas chromatography with flame ionisation detection	1 – 40 pptv	(Hopkins et al., 2003)
C6-C13 hydrocarbons	GCxGC-FID	2 dimensional gas chromatography with flame ionisation detection	0.01 – 0.2 pptv	(Lidster et al., 2014)
OH, HO ₂ , RO ₂	FAGE	Laser induced fluorescence	See text	(Whalley et al., 2013)
k _{OH}	LP-LIF	Laser flash photolysis, laser induced fluorescence	2.1 s ⁻¹	(Stone et al., 2016)
Meteorological parameters	Davis Vantage Vue	Met station	-	
Boundary layer depth	Halo-Photonics scanning Doppler lidar	Doppler lidar	30 m	(Barlow et al., 2015)
Aerosol surface area	TSI Inc, model 3321	Aerodynamic particle sizer spectrometer	0.001 particle/cm ³	(Peters and Leith, 2003)