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Ozone impacts on vegetation in a nitrogen enriched and changing climate

Gina Mills [a, *], Harry Harmens [a], Serena Wagg [b], Katrina Sharps [a], Felicity Hayes [a], David Fowler [c], Mark Sutton [c], Bill Davies [d]

[a] Centre for Ecology and Hydrology, Environment Centre Wales, Deiniol Road, Bangor LL57 2UW, UK
[b] School of Biological Sciences, Bangor University, Deiniol Road, Bangor, Gwynedd LL57 2UW, UK
[c] Centre for Ecology and Hydrology, Bush Estate, Penicuik, Midlothian EH26 0QB, UK
[d] Lancaster Environment Centre, Lancaster University, Lancaster LA1 4YQ, UK

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ABSTRACT

This paper provides a process-oriented perspective on the combined effects of ozone (O_3), climate change and/or nitrogen (N) on vegetation. Whereas increasing CO_2 in controlled environments or open-top chambers often ameliorates effects of O_3 on leaf physiology, growth and C allocation, this is less likely in the field. Combined responses to elevated temperature and O_3 have rarely been studied even though some critical growth stages such as seed initiation are sensitive to both. Under O_3 exposure, many species have smaller roots, thereby enhancing drought sensitivity. Of the 68 species assessed for stomatal responses to ozone, 22.5% were unaffected, 33.5% had sluggish or increased opening and 44% stomatal closure. The beneficial effect of N on root development was lost at higher O_3 treatments whilst the effects of increasing O_3 on root biomass became more pronounced as N increased. Both responses to gradual changes in pollutants and climate and those under extreme weather events require further study.

1. Introduction

Air quality impacts on vegetation are frequently considered as effects of single stressors working in isolation from other air pollutants, climate and soil modifying factors. In this review, we bring together current knowledge on how these other factors impact on the responses of vegetation to tropospheric ozone (O_3), classed by many as the most damaging air pollutant to vegetation (Ashmore, 2005; Karnosky et al., 2007), and predict from a process-perspective what the impacts might be for vegetation growing in a future nitrogen (N) polluted and changing climate. Tropospheric or "ground-level" O_3 is a secondary air pollutant formed and destroyed by a series of complex photochemical reactions involving nitrogen oxides (NO_x), methane (CH_4), carbon monoxide (CO) and non-methane volatile organic carbons (NMVOC) (Avnery et al., 2011; Royal Society, 2008). Ozone concentrations are usually highest in rural and upland areas downwind of major conurbations, where many vulnerable ecosystems provide essential services to humankind. Ozone and its precursors are also transported around the world in air masses, adding to background concentrations across the continents (Royal Society, 2008). Uptake of O_3 by sensitive plant species alters leaf physiology (increased respiration, reduced photosynthesis, stomatal functioning, enhanced senescence), reduces growth (both above– and below-ground and altering C allocation), and alters phenology (e.g. timing and number of flowers) (for further details see review by Ainsworth et al., 2012). In isolation, these process-effects may be sufficient to impact on key ecosystem services including crop yield, C sequestration by trees and grasslands, water provision and biodiversity (e.g. Harmens and Mills, 2012; McLaughlin et al., 2007a; Mills et al., 2013; Sun et al., 2012). As well as responding to O_3, under open field conditions, vegetation is frequently also exposed to atmospheric N inputs in oxidised (e.g. NO_2) or reduced (e.g. NH_3) form (see van der Berg et al., this issue), to climate extremes such as drought, heat stress and/or flooding, and to nutrient stress. We consider here whether we can reach a consensus on the direction of change of such impacts and what the implications might be for vegetation.

Ozone uptake is via the stomatal pores in the leaf surface and thus any plant, pollutant, climatic or soil factor that influences

* Corresponding author.
E-mail address: gmi@ceh.ac.uk (G. Mills).

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stomatal functioning will modify the amount of O_3 taken up by the plant and subsequent effects. Models of stomatal uptake or "flux" have been instrumental in improving our understanding of the areas and vegetation types at greatest risk of damage from the pollutant. For example, in the last 15 years, European scientists have developed models of O_3 flux – the so-called DO_3SE model (Emberson et al., 2000; LRTAP, 2014), applied the model to show areas of greatest risk (e.g. Simpson et al., 2007) and shown that field evidence provided greater support for the application of this biologically more meaningful risk assessment method than one based on the atmospheric concentration of O_3 above the canopy (Mills et al., 2011). Effects of climate change on stomatal O_3 flux can be either direct – e.g. temperature, CO_2 and humidity effects on stomatal conductance or indirect via an influence on soil water potential (SWP) and plant development (Harmens et al., 2007; Vandermeiren et al., 2009). In addition, O_3 itself can, for example, modify the responses of plants to naturally occurring environmental stresses such as drought (e.g. Mills et al., 2009; Wilkinson and Davies, 2009, 2010) via effects on stomatal control and plant development (canopy and roots), and can feedback to global warming by reducing the C sink strength (e.g. Sitch et al., 2007). We first review current predictions of future O_3, N deposition and physical climates and then consider how these combined effects might occur in the two-factor combinations in which they are usually studied (e.g. CO_2 and O_3, drought and O_3, N and O_3). We then speculate on how all might interact together in a future O_3, N and CO_2-enriched climate, focussing on responses to subtle long-term changes as well as to the extreme climatic and pollutant events predicted to occur with greater frequency in the coming decades (IPCC, 2013).

1.1. Trends and future projections

Over the past 150 years, increasing anthropogenic emissions of O_3 precursors and increased stratospheric mixing of O_3 into the lower layers of the atmosphere have led to an increase in the global average O_3 concentration from 20–30 ppb to 30–50 ppb, with significant spatial and temporal variability (IPCC, 2013). With the global population estimated to exceed 9 billion by 2050, associated increased demand for resources such as fossil fuels, energy production, transport and agriculture is likely to further increase O_3 precursor emissions and reactive nitrogen (Nr, all N compounds excluding N_2, and including NH_3, NO_3, NO_x and N_2O) in the environment (Oltmans et al., 2006; Winiwarter et al., 2011). Future O_3 and Nr trends will not only depend on anthropogenic emission levels of precursors, but also on trends in temperature, humidity and solar radiation. For example, a multi-model study of impacts of climate change alone on O_3 concentrations in Europe predicts increases in the mean O_3 concentration in the range 0.9–3.6 ppb for 2040-49 climates compared to 2000-09 climates, if precursor emissions remain constant (Langner et al., 2012). Despite overall rises in pollutant levels globally, there have been some (partial) success stories in reducing emissions in some regions of the world leading to large spatial heterogeneity in predictions of future pollutant concentrations. Here, we consider some of the changes that have occurred in recent decades, and using the latest modelling, speculate on future O_3 and Nr concentrations and the feedbacks to climate of these short-lived climate forcers (SLCFs). Further details on ozone and reactive nitrogen chemistry and trends can be found in recent reviews by Monks et al. (2015) and Fowler et al. (2015) respectively.

1.2. Ozone

In assessing all available data from ground-level monitoring stations in the Northern Hemisphere, Cooper et al. (2014) showed that the ozone concentrations have been rising by 1–5 ppbv per decade since the start of comparable records in the 1950s–1970s. Set against these rising background O_3 concentrations, regional controls on precursor emissions have led to reductions in peak O_3 concentrations in North America and Europe over recent decades. For example, in the once highly polluted California South Coast Air Basin (SoCAB), the annual 8 h mean O_3 concentration has declined between 1973 and 2010, from over 300 ppb to approx. 100 ppb (Pollack et al., 2013) and declines in peak O_3 concentration of ca. 30 ppb from 1990 to 2006 have been reported in many parts of Europe (Simpson et al., 2014; Tørseth et al., 2012). Even so, potentially plant-damaging O_3 episodes are still occurring in the USA and Europe when climatic conditions conducive to O_3 formation (hot, dry and sunny days) coincide with precursor abundance. For example, in 2006, two significant O_3 episodes occurred in Europe between 17–22 July and 25–28 July with O_3 concentrations in excess of 90 ppb experienced in many countries (EEA, 2007). Often used as an example of a future climate year for Europe, these high episodes in 2006 suggest that the immediate threat caused by O_3 episodes will remain under climate change.

In estimating future regionally averaged O_3 concentration from 14 global transport models for a range of Representative Transport Pathways (RCP) emission scenarios without changes in climate, Wild et al. (2012) predicted very small decreases in global surface O_3 in 2050 relative to 2000 of 2 ± 0.5 ppb for RCP2.6, 0.8 ± 0.54 ppb for RCP4.5 and 0.4 ± 0.2 ppb for RCP6.0 and an increase of 1.5 ± 0.5 ppb for the most pessimistic scenario, RCP8.5. Predictions to 2050 are similar for North America and Europe with a decrease of 2–4 ppb for RCPs 4.5 and 6.0, whilst in Asia, O_3 concentrations for these RCPs range from little net change (RCP6.0) to a 3 ppb increase and 2.5 ppb decrease for RCP4.5 in south and east Asia, respectively (Wild et al., 2012; reproduced in Fig. 1). Over 75% of these predicted changes in O_3 can be attributed to changes in methane abundance – a clear target for international pollutant emission control negotiations (HTAP, 2010).

Having a dual role as both a pollutant and an SLCF, increasing O_3 concentrations have been, and will in the future continue to be, influential in global warming. As well as directly influencing radiative forcing as a greenhouse gas (Shindell et al., 2013), O_3 also suppresses the global land carbon sink by reducing photosynthesis and carbon sequestration thereby increasing the CO_2 concentration in the atmosphere and indirectly increasing radiative forcing (Collins et al., 2010; Sitch et al., 2007). Global warming alone would lead to increases in O_3 production (Langner et al., 2012; Rasmussen et al., 2013), potentially negating beneficial effects of controls of precursor emissions. This so-called "climate penalty" arises from increased BVOC emissions (contributing, for example, to 8% of predicted increases in O_3 in China by 2050 (Wang et al., 2013)), accelerated photochemical reactions leading to more O_3 formation and increased stagnation of air masses (for further information, see Rasmussen et al., 2013). The climate effect of ozone is higher in NO_x-saturated areas such as the SoCAB than in NO_x-limited regions of California such as the San Joaquim valley (Rasmussen et al., 2013). Similar differences in magnitude and direction of the O_3 -climate penalty are predicted at the regional scale, for example, modelling by Wang et al. (2013), suggested a climate-change benefit under present day emissions for 2050 of a 5 ppb reduction in O_3 in the less populated west and a climate penalty of a 3 ppb increase in the more densely populated eastern provinces of China for the IPCC A1B scenario. Adding domestic and hemispheric emission projections as well as climate change for 2050, Wang et al. (2013) predicted similar increases overall in O_3 concentration of 9 and 10.3 ppb for east and west China respectively. Their modelling suggests large differences in attribution, with domestic and global

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