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# Deciphering the ozone-induced changes in cellular processes: a prerequisite for ozone risk assessment at the tree and forest levels

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## Abstract

• **Key message** Ozone, one of the major atmospheric pollutants, alters tree growth, mainly by decreasing carbon assimilation and allocation to stems and roots. To date, the mechanisms of O<sub>3</sub> impact at the cellular level have been investigated mainly on young trees grown in controlled or semi-controlled conditions. In the context of climate change, it is necessary to introduce a valuable defence parameter in the models that currently predict O<sub>3</sub> impact on mature trees and the carbon sequestration capacity of forest ecosystems.

• **Context** Air pollution is an important factor that affects negatively forest ecosystems. Among oxidative air pollutants, ozone is considered as the most toxic in terms of impact on vegetation.

• **Aims** This paper focuses on the negative impacts of ozone on trees in controlled conditions or in their natural environment. The current knowledge of the responses at cell level is presented and ways to improve their use for ozone risk assessment of forest stands are discussed.

• **Methods** Information was collected from original papers or reviews, providing an overview of the research conducted over the last 60 years.

• **Results** The negative effects of ozone on carbon assimilation and tree biomass production were reviewed and discussed, with a focus on effects on cell processes implied in cell defence, including stomatal regulation, detoxification, signalling, and biosynthesis of wood compound.

• **Conclusion** In the context of increasing significance of O<sub>3</sub> flux approach, this review intends to shed light into the black box of defence processes, which are playing a crucial part within the effective O<sub>3</sub> dose modelling. Today, it is recognized that tropospheric ozone inhibits tree growth and its role on the future carbon sink of the forest ecosystem is discussed along with the combination of other environmental factors like elevated temperature,

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water, and nitrogen supply, likely to be modified in the context of climate change.

**Keywords** Ozone impact · Defence · Carbon assimilation · Carbon allocation · Stomata · Detoxification · Signalling · Cell wall

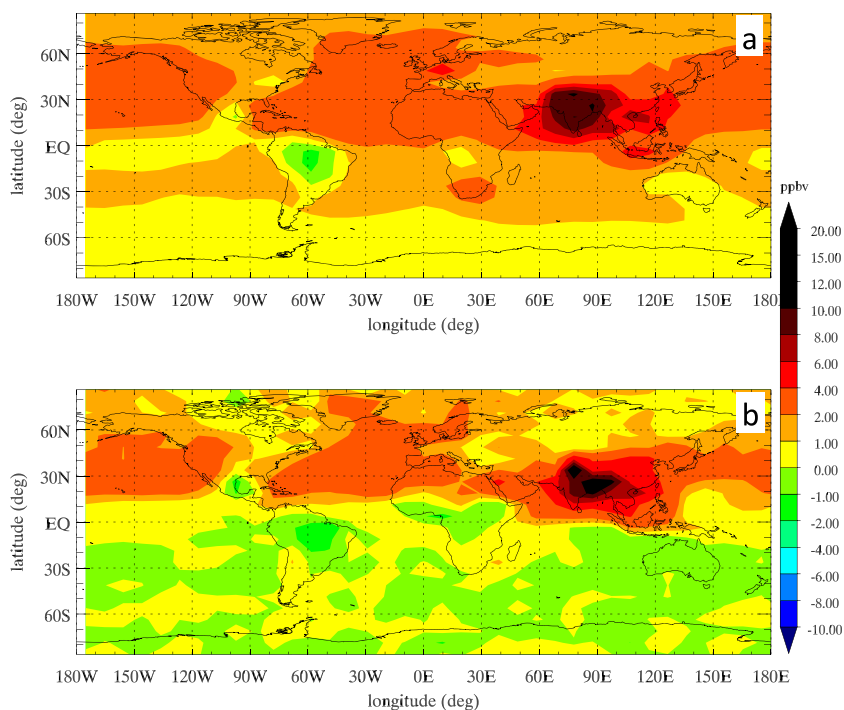
## 1 Introduction

As a result of human activities, Earth climate is assumed to change (IPCC 2013) as we enter a new geologic era, the Anthropocene (Barnosky et al. 2012; Steffen et al. 2011). It is now well established that the release of greenhouse gases (GHGs) affects climate on a global scale, since these gases modify radiative transfer and thus change the Earth's energy balance (IPCC 2013). Tropospheric ozone ( $O_3$ ), one of these GHGs and an important component of air pollution, is predicted to spread over large parts of the globe in the coming decades (Dentener et al. 2005; Fig. 1). In addition, this pollutant is thought to impact negatively forest productivity (Ainsworth et al. 2012), although species composition can modulate this effect (Wang et al. 2016). Since a large part of global forest areas is predicted to be exposed to  $O_3$  in the future (Fowler et al. 1999), carbon sequestration by forests may be reduced (Sitoh et al. 2007; Subramanian et al. 2015). Historically, the phytotoxic effect of photooxidants, including  $O_3$ , was first discovered in the 1950s in mixed conifer forests from the Los Angeles basin (Haagen-Smit et al. 1952). High concentrations of tropospheric  $O_3$  are an urban problem linked to car traffic and  $NO_x$  formation, but  $O_3$  or its precursors are easily airborne

and the pollutant can damage forest trees far from the source of emission. In the 1980s, several research groups showed that visible symptoms of injury on tree leaves in different regions of the USA were clearly related to the effect of photooxidants (Miller et al. 1997; Skelly et al. 1997). In Europe, in the middle of the 1980s, the German foresters were the first to draw attention to visible damages observed on coniferous trees, incriminating air pollution (Krause et al. 1986). The same observation was also made in Eastern France, leading to the development of a bilateral cooperation for exploring the causes of this problem between the French DEFORPA programme (1984–1991) and the German partners, followed with the common EUREKA programme EUROSILVA (1992–1994). In the 1990s, extended European cooperation (eight countries) started on the effects of ground-level  $O_3$  on trees and on the reduction of air pollutants, linked to the European Framework Programme for Research (e.g. STEP) and within the United Nations Economic Commission for Europe (UNECE).

The first experiments operated by these research programmes were conducted on trees in the field, e.g. in Germany (Weidmann et al. 1990) and in Austria (Wieser and Havranek 1993). Subsequently, experiments in controlled conditions (open-top chambers and phytotron chambers) were set up to decipher the mechanisms of  $O_3$  impact on leaves of young trees (Gerosa et al. 2009; Sandermann et al. 1997). The trees display a series of defence responses to  $O_3$  which, when overwhelmed, leads to different types of damages including leaf necrosis and growth reduction. The obtained results proved that, before visible symptoms appear,  $O_3$  affects the leaf metabolism by damaging the photosynthetic machinery and by increasing carbon use, further

**Fig. 1** Predicted differences in decadal averaged surface  $O_3$  concentrations (ppbv) comparing the 2020s and the 1990s for two global chemistry-transport models (from Dentener et al. 2005). **a** TM3 CLE Eulerian global chemistry-transport model using the current legislation (CLE) scenario; **b** STOCHEM CLE Lagrangian tropospheric chemistry-transport model using the current legislation (CLE) scenario



leading to reduced growth and productivity (Dizengremel 2001; Heath and Taylor 1997). The development of free-air CO<sub>2</sub> enrichment (FACE) projects provided a nice opportunity to study forest ecosystem responses to the increase of tropospheric O<sub>3</sub> combined or not with high CO<sub>2</sub> exposure (King et al. 2005). The gap between the two approaches (field versus controlled conditions) has recently been partly reduced through the German CASIROZ programme by developing a large set of ecophysiological and biochemical analyses on mature beech and spruce trees submitted to free-air O<sub>3</sub> fumigation (Matyssek et al. 2010b), on birches in Finland (Oksanen et al. 2007) and on conifers in Austria (Wieser et al. 2013). However, the degree of sensitivity to ozone is highly variable between tree species (Reich 1987; Wittig et al. 2009), raising the question of the underlying physiological processes (Matyssek et al. 2012).

In 2000, to meet the request from policy makers to the scientific community for quantitative information about O<sub>3</sub> effects, a specific European programme for the assessment, validation and mapping of visible O<sub>3</sub> injury on the vegetation was set up, based on the ICP Forests monitoring network. The concept of a critical level for O<sub>3</sub>, introduced by UNECE (LRTAP convention 2004), was originally based on the accumulated exposure over a threshold concentration of 40 ppb (AOT40). For forest trees, exceeding an AOT40 value of 5 ppm.h accumulated over one growing season would cause growth reduction (LRTAP convention 2004). More recently, a flux-based concept was developed in order to take into account the actual O<sub>3</sub> flux in the leaf through the stomata (Emberson et al. 2000; Grünhage et al. 2004; Karlsson et al. 2004). Indicators such as AFstY (accumulated stomatal flux above a threshold of  $Y \text{ nmol m}^{-2} \text{ s}^{-1}$ ) or POD<sub>Y</sub> (phytotoxic ozone dose above a threshold flux of  $Y \text{ nmol m}^{-2} \text{ s}^{-1}$ ) appear more representative of the impact of O<sub>3</sub> on vegetation as, being based on the Jarvis multiplicative model of stomatal conductance (Jarvis 1976), they take into account temperature, water vapor pressure deficit (VPD), light, soil water potential, concentration of O<sub>3</sub> and the plant phenology. These indices simulate the uptake of O<sub>3</sub> in leaves and represent the hourly average flux accumulated above a threshold  $Y$  by a leaf during the plant growth. Hoshika et al. (2014) also proposed to take into account a stomatal O<sub>3</sub> flux per net photosynthesis rate rather than stomatal O<sub>3</sub> flux only. All these flux-based methodologies rely on empirically derived relationships, linking stomatal O<sub>3</sub> flux to tree biomass loss through a series of models at leaf, tree and forest levels. These relationships thus provide estimates of the effective O<sub>3</sub> dose, i.e. the fraction of O<sub>3</sub> flux exceeding the plant detoxification capacity, under consideration of the environmental conditions (Matyssek et al. 2004; Musselmann et al. 2006; Dizengremel et al. 2008; Heath et al. 2009; Buker et al. 2015). The importance of the detoxification capacity of the tree was recently emphasized (de Temmerman et al. 2002; Dizengremel et al. 2013; Dizengremel et al. 2008) and was previously raised to explain

the differences in O<sub>3</sub> sensitivity of tree species (Pell et al. 1999). Detoxification capacity depends on a network of molecular and physiological processes which needs to be deciphered in order to identify a reliable parameter, integrated in models and allowing a more accurate risk assessment. In brief, the integration in field models of a pertinent ozone-damaging factor, identified at the cellular level, could improve risk assessment and help policy makers with related socio-economic decisions.

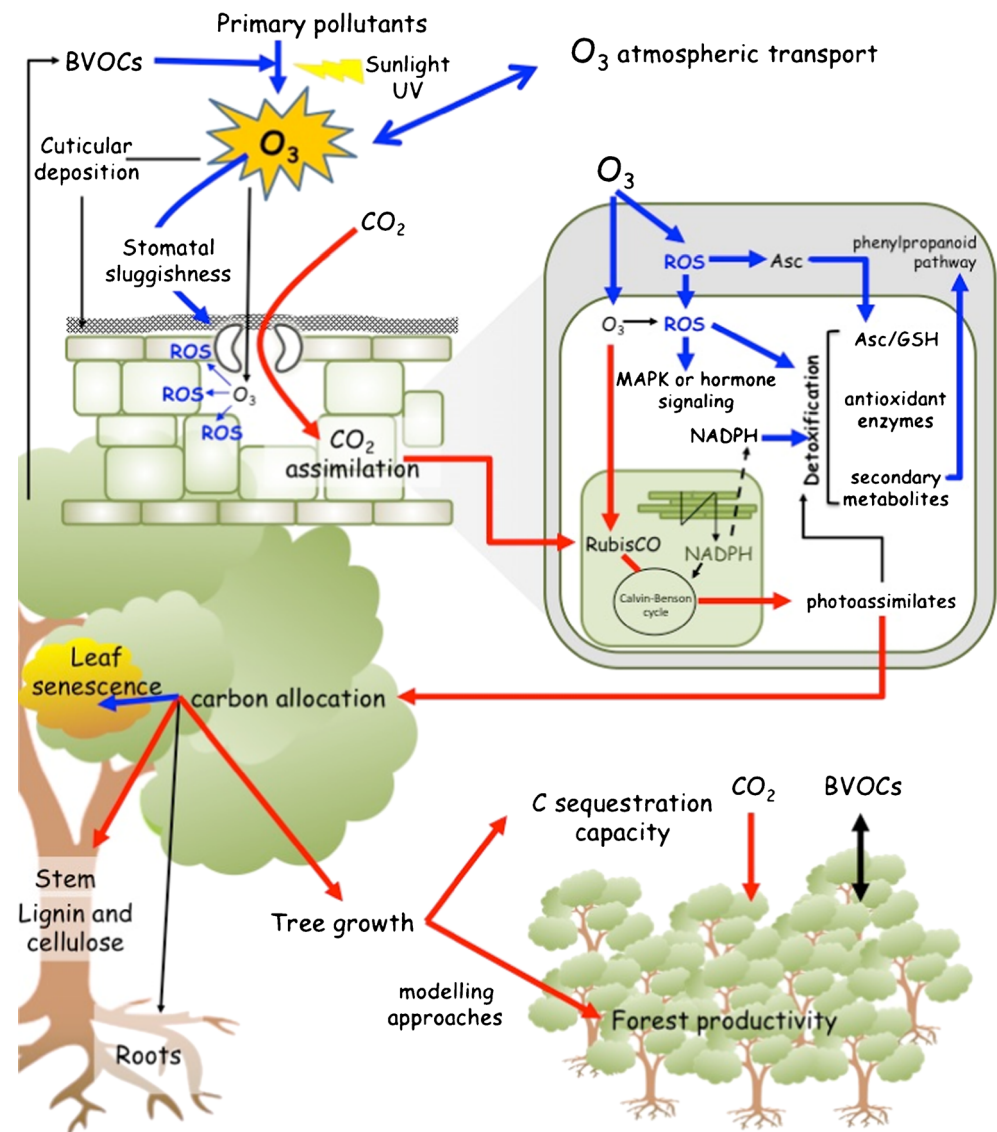
Since 60 years, O<sub>3</sub> research has covered a large spectrum of interest, including detailed studies of the cellular events induced by acute doses. In this review, acute doses correspond to observed peak concentrations with values above 120 ppb for several days, while chronic exposure refers to long-term exposures (weeks, months, years) to ozone concentrations below 120 ppb. The increasing interest to study the effects of this gas is partly due to the fact that the exposure to high O<sub>3</sub> concentrations elicits a strong oxidative stress at the tissue level. Moreover, with hourly peak O<sub>3</sub> concentrations at peri-urban regions reaching 200 ppb (Feng et al. 2014), acute conditions are de facto validated in natural conditions inducing serious damage to plants. O<sub>3</sub> research has also known a progressive extent in the study of chronic exposure with values not higher than 100 ppb over the growth period. However, experiments in controlled conditions with slightly higher O<sub>3</sub> exposure (up to 120 ppb) applied every day during a shorter time (no more longer than 3 weeks/1 month) were also frequently used to mimic the effect of these ambient long-lasting O<sub>3</sub> exposures. Even though the increase in tropospheric O<sub>3</sub> concentrations has recently flattened in mid-latitudes of the Northern Hemisphere (Oltmans et al. 2013), assessing the effects of O<sub>3</sub> on forest trees remains a timely question considering that the threshold for a negative impact on growth has been already reached. In this context, we present a survey of works on the effects of this oxidative pollutant in a relative short term on tree physiology (Fig. 2). These information are necessary to extrapolate on a longer term the O<sub>3</sub> effect at the forest ecosystem scale, which have also to integrate with the complex inter-relationships among the environmental factors occurring over the life span of a tree.

## 2 Formation, transport, deposition of O<sub>3</sub>, and leaf damage

### 2.1 Formation

In the troposphere, under the action of sunlight, primary pollutants like nitrogen oxides and hydrocarbons are able to form photochemical air pollutants, referred to as secondary pollutants, namely peroxyacetyl nitrate (PAN), O<sub>3</sub> and to a lower extent aldehydes and ketones (Becker et al. 1985). The future of PAN is closely related to atmospheric

**Fig. 2** Overview of  $O_3$  effects on trees, from cell metabolism to forest ecosystem scale, highlighting (i) the perception of the pollutant at the leaf scale, (ii) the cellular responses implying detoxification and  $CO_2$  assimilation and (iii) the carbon allocation to the various plant organs and the consequences on tree growth and on carbon sequestration at the forest level. Where arrows are present, red and blue indicate an  $O_3$ -driven inhibition and stimulation, respectively. BVOCs biogenic VOCs



temperatures and could contribute to  $O_3$  formation in the warmer lower atmosphere (Singh 1987). The presence of  $O_3$  in the troposphere was initially considered as the result of a stratospheric  $O_3$  transfer, which only accounts for up to 25 % of tropospheric  $O_3$  (LRTAP convention 2010). In fact,  $O_3$  formation mainly results from complex processes already detailed in several reviews (Becker et al. 1985; Royal Society 2008; Stockwell et al. 1997). Briefly, once nitrogen dioxide ( $NO_2$ ) is formed, it endures a photodissociation caused by short radiations (between 280 and 430 nm) producing nitrogen monoxide ( $NO$ ) and free oxygen atoms. The free oxygen atom presents a high excitation level leading to a reaction with  $O_2$  to form  $O_3$ . The subsequent reaction of  $O_3$  with  $NO$  can lead to the destruction of  $O_3$  in a non-polluted area, where the  $NO_2/NO$  ratio is low. However, one parameter susceptible to unbalance these reactions is the presence of volatile organic compounds (VOCs) including  $CH_4$  (Royal Society 2008). VOCs are able to oxidize

$NO$ , increasing the  $NO_2/NO$  ratio and shifting the reactions towards  $O_3$  accumulation. Finally, the production of  $O_3$  in the troposphere is linked to changing precursor concentrations, a relationship that highlights the non-linearity of the  $O_3$ -VOC- $NO_x$  system (Monks et al. 2015). Vegetation and particularly forests are natural VOC producers (Sharkey et al. 2008). In a context of warming climate, VOC emissions are projected to increase and to contribute to the occurrence of  $O_3$  peaks or the increase of the tropospheric  $O_3$  background level. Finally, the photolysis of  $O_3$  leads to additional radicals that can react with carbon monoxide and organic species, leading to additional  $O_3$  production (Royal Society 2008).

## 2.2 Transport of the pollutants in the troposphere

Most of air pollutant emissions, including  $O_3$  precursors, originate from regions within the mid-latitudes, where long-range

transport of air masses is dominated by westerly winds. These winds convey emissions from source regions to downwind regions at interregional and even intercontinental scales (Stohl and Eckhardt 2004). The lifetime of O<sub>3</sub> in the free troposphere varies from weeks to months, which is compatible with long-range transport that occurs on timescales of days to weeks (LRTAP convention 2010). Due to the stronger winds at high altitudes, O<sub>3</sub> formed or transported into the mid- and upper troposphere travels further and faster than O<sub>3</sub> remaining in the lower troposphere, below 3 km in altitude (LRTAP convention 2010). O<sub>3</sub> formation can also occur at distance from precursor source regions, when polluted air masses arrive at a downwind region and meet conditions that promote O<sub>3</sub> formation (Lin et al. 2012).

When confined within the atmospheric boundary layer, O<sub>3</sub> has a relatively short lifetime (hours to days) due to dry deposition at terrestrial surfaces (Wesely and Hicks 2000). Dry deposition occurs when O<sub>3</sub> is taken up or absorbed onto surfaces (vegetation, soil, materials) that provide a chemical sink for O<sub>3</sub> decomposition (Cape et al. 2009). Surface removal represents an important control on the near-surface O<sub>3</sub> concentrations and constitutes a major term in the global mass balance of tropospheric O<sub>3</sub> (Fowler et al. 2009). While molecular processes become important very close to surfaces (less than 1 mm), turbulent transfer represents the main driver of gas exchange between vegetation and the atmosphere. Forests being aerodynamically rough surfaces, the rates of turbulent exchange between the atmosphere and forests exceed by an order of magnitude or more those over crops or grasslands (Fowler et al. 1999). As a consequence, forests represent a major sink for O<sub>3</sub> dry deposition.

### 2.3 O<sub>3</sub> deposition to soil and vegetation

The foliage of forest trees acts as the dominant O<sub>3</sub> sink in the atmosphere-forest interaction, and the canopy structure has a noticeable effect on its uptake (Zhang et al. 2006). Forest ecosystems have the capacity to remove O<sub>3</sub>, through both stomatal and non-stomatal mechanisms (Fig. 2) (Dizengremel et al. 2013; Fares et al. 2013b). The stomata are the main entry point of ozone into the leaves, and the non-stomatal mechanisms of ozone deposition include cuticular deposition, deposition at the soil surface and destruction by chemical reactions (NO<sub>x</sub>, biogenic VOC). Most studies of vertical O<sub>3</sub> concentration gradients show that only minor variations occur throughout forest canopies during daytime, mainly due to convective mixing caused by solar radiation (Jaggi et al. 2006). Overall, during daytime, the O<sub>3</sub> concentrations below the canopy of various forest types are 0 to 15 % lower than those measured at the top of the canopy (Andreae et al. 2002; Fontan et al. 1992; Joss and Graber 1996). However, stronger gradients appear at night because of a greater air stability, which limits the exchange between the

canopy and the atmosphere above, and due to radiative cooling inducing strong temperature inversions near the ground (Skelly et al. 1996). As a consequence, the O<sub>3</sub> concentrations near the forest floor are lower than those measured in the canopy or in the atmosphere above, especially at night (Fontan et al. 1992). By combining O<sub>3</sub> concentration measurements and stomatal conductance estimations, it is possible to get a rather good knowledge of the dose absorbed by the plant. However, the determination of the O<sub>3</sub> stomatal conductance is far from trivial. Whatever the method used, the values are subject to uncertainties. Many measurements and modelling studies of O<sub>3</sub> flux for various canopies and different seasons exist (Massman 2004; Padro 1996; Wesely and Hicks 2000). However, there are still large doubts concerning the processes controlling O<sub>3</sub> deposition to plant surfaces (Ashmore et al. 2007), and therefore in the partitioning of the O<sub>3</sub> flux between stomatal and non-stomatal uptakes, whose relative contributions vary with canopy type and with the season of the year (Tuovinen et al. 2009; Zhang et al. 2006). Many studies have been dedicated to this question, and most of them show an enhancement of O<sub>3</sub> deposition with increased surface wetness.

Significant (20–80 % of total) non-stomatal O<sub>3</sub> fluxes have been observed in different forests in southern European conditions (Cieslik 2009), which often limit the gas flux through stomata. A 10-year-long measurement in a boreal Scots pine forest in Finland showed that the non-stomatal O<sub>3</sub> deposition in the daytime during the growing season varied within 26–44 % of total deposition (Rannik et al. 2012). Another decade-long dataset collected in a mixed temperate forest in Belgium showed larger non-stomatal fractions, exceeding 60 % even during the daytime in summer (Neiryneck et al. 2012).

The correct quantification of the different components of the deposition, including the stomatal fraction, is also required when assessing the possible feedbacks between O<sub>3</sub> uptake rates and plant injury or damage, photosynthesis and plant defences. Scaling functional processes of forest trees from leaves and compartments (soil, canopy) to stands, ecosystems and, finally, the landscape level (Wieser et al. 2008) is fundamental for understanding the capacity of forest ecosystems to mitigate air pollution effects and to adapt to changing environmental conditions (Matyssek et al. 2012).

### 2.4 Leaf damage

As a strong oxidant, O<sub>3</sub> causes several types of visible injury, including chlorosis and necrosis (<http://hermes.wsl.ch/didado/ozoniwww.page0?sprache=E>). These symptoms, well characterized in controlled conditions, could be also observed on leaf trees in rural areas and mountains, downwind from cities (Dalstein et al. 2002; Feng et al. 2014; Miller et al. 1994) or in forested areas exposed to ambient O<sub>3</sub> concentrations high enough to produce phytotoxic effects (de Vries et al. 2014). In this latter case, it is obvious that the

interpretation is more doubtful, conditioned by the complex interactions between  $O_3$  and environmental factors inside the canopy of adult trees or linked to water and mineral availability of the soil (Bussotti and Ferretti 2009; Manning 2005).  $O_3$  can also induce an accelerated senescence and leaf abscission (Gielen et al. 2007; Karnosky et al. 2005; Ribas et al. 2005). Finally, although visible symptoms can be useful for detecting an  $O_3$  impact, their relationship with growth reduction is not always found (de Vries et al. 2014).

### 3 Cellular and molecular mechanisms impacted by $O_3$

#### 3.1 Impact on guard cells

Ainsworth et al. (2012) reported that a reduction in stomatal conductance in plants exposed to chronic elevated  $O_3$  (range to 80–120 ppb) could be attributed to a direct effect of  $O_3$  on photosynthesis and to a resultant increase in internal  $CO_2$  concentration. However, alternative reactions might explain this response (Pell et al. 1992). In fact, studies reported that stomata are impaired by chronic  $O_3$  exposure in their ability to close rapidly in response to environmental stimuli (McAinsh et al. 2002; Reich et al. 1984). More recently, it was shown that stomata open and close more slowly in response to changing light conditions, VPD or  $CO_2$  concentrations as a result of 120 ppb  $O_3$  exposure (Dumont et al. 2013) or with 1.5- to 2-fold  $O_3$  ambient level (Paoletti 2005; Paoletti and Grulke 2010). Some molecular aspects were studied in order to explain how  $O_3$  modifies the signals involved in opening or closure processes (Vahisalu et al. 2010). Dumont et al. (2014) showed on poplar genotypes that modification of stomatal responses by an exposure to 120 ppb of  $O_3$ , such as stomatal sluggishness (Fig. 2), does not result from ultrastructural changes but from a disturbance of ion fluxes and a regulation of the gene expression involved in signal transduction. The expression of a majority of the studied genes coding for plasma membrane and vacuolar channels was inhibited by  $O_3$ , especially the expression of genes coding for the plasma membrane proton ATPase (AHA11) and the vacuolar calcium channels (CAX1 and CAX3) (Dumont et al. 2014).

There is also more recent evidence that stomatal conductance is not universally reduced by elevated  $O_3$  concentration, but that leaf age and tree developmental stage can alter the degree to which  $O_3$  affects stomatal conductance (Uddling et al. 2009). Danielsson et al. (2003) and Pleijel et al. (2002) added the effect of  $O_3$  to the phenology function (related to the reduction of the stomatal conductance in senescing leaves) of the stomatal conductance model in potato and wheat. That new function is being increasingly used, but more knowledge is needed to determine if the effects of  $O_3$  on trees should really be integrated in the same way as to link the  $O_3$  function

to another function than phenology function so far. Further research is needed (i) to characterize  $O_3$  impacts on stomatal function as well as the interaction with other abiotic stresses like drought and (ii) to improve stomatal conductance models like the DO3SE model (Büker et al. 2012). In these models, we need to improve the estimation of the start and end of the growing season today and in the future.

#### 3.2 Detoxification

Once  $O_3$  enters the sub-stomatal chamber, it rapidly induces the formation of reactive oxygen species (ROS) like hydrogen peroxide, singlet oxygen and hydroxyl radicals, increasing the oxidative load to the apoplastic fluid (Fig. 2). The primary effect of ROS is the alteration of membrane and enzyme proteins, e.g. Rubisco (Dizengremel 2001; Heath 2008; Pell et al. 1992). The ability to limit the occurrence of ROS in the apoplast could confer the  $O_3$  tolerance of aspen clones (Oksanen et al. 2004). Apoplastic ascorbate (Asc) is considered as the first line of defence against  $O_3$ , as observed in herbaceous plants (Conklin and Barth 2004) but also in beech (Haberer et al. 2007; Luwe and Heber 1995), silver birch (Padu et al. 2005) or spruce and pine needles (Polle et al. 1995). However, the Asc levels in the apoplast are not sufficient to explain the different degrees of  $O_3$  sensitivity in poplar clones (Ranieri et al. 1999). The lack of any direct relation between species sensitivity and apoplastic Asc levels was attributed (i) to the high level of Asc oxidation in this compartment, (ii) to its low concentration relative to the total cell content and (iii) to the occurrence of other antioxidants (Castagna and Ranieri 2009; Dizengremel et al. 2013).

At the whole-leaf level, changes in Asc content increased or decreased according to the exposure level and the duration of  $O_3$  treatment, the leaf age, the growth stage or the position of the leaf in the canopy (Haberer et al. 2007; Strohm et al. 2002; Tausz et al. 2004; Wellburn et al. 1996). Thus, the higher  $O_3$  sensitivity of young beech in phytotrons compared to adult forest trees in the field has been partly attributed to differences in detoxification capacity and notably total ascorbate concentration (Nunn et al. 2005). From a series of studies conducted on poplar exposed to  $O_3$  in controlled conditions or in the field, it is difficult to establish a clear relation between the constitutive level of Asc and the genotype sensitivity (Di Baccio et al. 2008; Dumont et al. 2014; Yun and Laurence 1999). However, a higher level of dehydroascorbate (DHA) in a sensitive poplar genotype (Dumont et al. 2014) exposed to 120 ppb  $O_3$  may express a lower capacity of the genotype to regenerate the reduced form (AsA), a hypothesis supported by a lower NADPH content in the leaves of this genotype (Dghim et al. 2013a).

In the absence of glutathione in the apoplast, the Asc regeneration process implies the transport of DHA in the

cytosol followed by the functioning of the intra-cellular ascorbate-glutathione cycle (Noctor 2006). This cycle sustains AsA regeneration in the cytosol. Glutathione is known as an antioxidant, able to directly react with ROS, sometimes functioning in a compensatory manner to Asc but also with specific functions (Noctor et al. 2012). Total glutathione and/or reduced glutathione content generally increased in leaves of tree species under O<sub>3</sub> fumigation (100 to 120 ppb) (Dumont et al. 2014; Wellburn et al. 1996). In adult beech trees, the glutathione level is affected by canopy position, but O<sub>3</sub> exposure (twofold ambient concentration) involved higher content in both shade and sun leaves (Herbinger et al. 2005). In some works, the differences in the constitutive levels of glutathione between poplar genotypes appeared to contribute to the higher tolerance to chronic O<sub>3</sub> exposure (Di Baccio et al. 2008; Dumont et al. 2014). However, attempts to increase glutathione content and glutathione reductase activity in transgenic poplar were unsuccessful in increasing tolerance, at least to acute (300 ppb) O<sub>3</sub> exposure (Strohlm et al. 2002).

Phenolic compounds are also recognized in leaf like important metabolites to cope with elevated or chronic O<sub>3</sub> exposure of trees (Fares et al. 2010; Kontunen-Soppela et al. 2007; Peltonen et al. 2005; Yamaji et al. 2003). For some phenolics and condensed tannins, a potential role in the O<sub>3</sub> tolerance has been claimed (Haikio et al. 2009; Kontunen-Soppela et al. 2007). Furthermore, in accordance with an increased level in phenolics, the induction of the shikimate and phenylpropanoid pathways shared by the flavonoid, anthocyanin, tannin, stilbene and lignin biosynthesis has been well documented under O<sub>3</sub> exposure (see Cabané et al. 2012 for review). For some tree species that emit large amounts of VOCs, it is also interesting to consider the potential role of these compounds like antioxidants, as found for isoprene in poplar (Loreto et al. 2001) and for monoterpenes in oak (Loreto et al. 2004). However, recent works on poplar showed that the relationship between O<sub>3</sub> tolerance and the ability to emit isoprene is not so clear (Behnke et al. 2009; Calfapietra et al. 2008). The involvement of polyamines like radical scavengers and protectant against O<sub>3</sub> has been also considered although these studies are limited (Ludwikow and Sadowski 2008).

In addition to metabolites, a large panel of antioxidant enzymes are involved in the defence mechanisms to decrease the ROS level or to regenerate the reduced form of some antioxidants (Fig. 2). Firstly, extracellular enzymes like superoxide dismutases (SOD) and peroxidases (with ascorbate or phenolics as preferential electron donors) were identified as significant contributors to the mitigation of ROS generation in the apoplast of birch (Padu et al. 2005) and poplar leaves (Castagna and Ranieri 2009). When the apoplastic antioxidant capacity is overwhelmed, other isoforms of these enzymes were implied to limit the generation of cytoplasmic ROS. Thus, total SOD and/or peroxidase activities generally increased in O<sub>3</sub>-treated leaves of trees (Bernardi et al. 2004;

Diara et al. 2005; Sehmer et al. 1998; Tuomainen et al. 1996) even though conflicting results have been mentioned (Heath and Taylor 1997). It has been claimed that the different affinities for ROS of the antioxidant enzymes either may be linked to the regulation of ROS as signalling actors or may be responsible for the removal of excess ROS (Mittler 2002). More generally, maintaining a cellular steady state of the ROS in responses to stresses is assigned to a complex enzyme network with actors like thioredoxin-dependent peroxidases (including peroxiredoxins and glutathione peroxidase), glutaredoxins and glutathione-S transferases (Foyer and Noctor 2011; Mittler et al. 2004; Rouhier and Jacquot 2005). Hence, the regulation of the enzymatic antioxidant system involves a fine redox regulation (Jacquot et al. 2013). For some of these enzymes, their characterization in trees (mainly poplar) is recent (Navrot et al. 2006; Rouhier 2010) but their regulation under O<sub>3</sub> or other oxidative pollutants is still undefined. Moreover, in these stress conditions, the source of reducing power to supply some of the ROS-scavenging systems remains to be elucidated (Dghim et al. 2013b; Dizengremel et al. 2008). Finally, to propose a new index for O<sub>3</sub> risk assessment integrating the plant detoxification capacity, a complex network of metabolites and enzymes has to be taken into account. But for trees, the variations of these parameters along the successive growing seasons as well as the canopy position must also be considered.

### 3.3 Carbon assimilation and leaf senescence

One of the first O<sub>3</sub>-driven decreases in growth was characterized by Reich (1983) in greenhouse-grown trees and directly correlated to a reduction in net CO<sub>2</sub> assimilation rate (Fig. 2). This correlation was then confirmed in fumigation chamber, open-top chamber and field fumigation systems across several tree species (Reich 1987). In a meta-analytic review, Wittig et al. (2007) showed that the O<sub>3</sub>-driven decrease in carbon assimilation reached 14 % for angiosperm trees grown in ambient background O<sub>3</sub> relative to charcoal-filtered air. Differently, gymnosperms were not significantly affected. However when exposed to severe O<sub>3</sub> concentration (85 ppb), net CO<sub>2</sub> assimilation was similarly decreased (up to 19 %) in both angiosperms and gymnosperms. The average decrease in CO<sub>2</sub> assimilation was progressively greater as the O<sub>3</sub> treatment increased (Wittig et al. 2007). O<sub>3</sub> impact on photosynthesis could be threshold dependent, highlighting the importance to better define critical O<sub>3</sub> threshold across species and/or environmental conditions. Differences in photosynthesis response to O<sub>3</sub> may partly be explained by interspecific variability of stomatal response to O<sub>3</sub> and therefore gas flux entering the leaf. Over hundreds of individuals indicate an 11% and 13 % decrease in average in CO<sub>2</sub> uptake and stomatal conductance, respectively (Wittig et al. 2007). Stomatal limitation under O<sub>3</sub> may in part explain



the reduced photosynthetic CO<sub>2</sub> uptake in restricting CO<sub>2</sub> diffusion from the atmosphere to the intercellular space and therefore CO<sub>2</sub> availability at the site of carbon fixation (see section on stomatal regulation above).

In addition, O<sub>3</sub> has been widely shown to affect both the Calvin-Benson cycle and photochemistry activity (Fig. 2) (Saxe 2002). A decrease in Rubisco activity and content was strongly correlated with cumulative O<sub>3</sub> exposure in loblolly pine (Dizengremel et al. 1994). Similarly, in Aleppo pine, both Rubisco and Rubisco activase levels were reduced under O<sub>3</sub> (Pelloux et al. 2001). In angiosperms also, multiple works supported the idea of O<sub>3</sub>-driven alteration of Rubisco activity (Gaucher et al. 2003; Lutz et al. 2000; Matyssek et al. 1991; Pell et al. 1992). However, it remains unclear whether the decrease in Rubisco activity is due to enhanced degradation, ROS-mediated protein oxidation, decreased Rubisco activase activity or altered gene expression (Brendley and Pell 1998; Dizengremel 2001; Heath 2008; Pell et al. 1994). It is well documented that chronic as well as acute O<sub>3</sub> exposures impact gene regulation (Ernst 2013; Renaut et al. 2009) even though acute episodes could provoke more important changes than chronic ones (Ainsworth et al. 2012; Ernst 2013). Furthermore, proteomic analysis confirms the downregulation of a large number of proteins involved in the Calvin-Benson cycle in poplar leaves exposed to chronic O<sub>3</sub>, as well as proteins involved in chloroplastic electron transport (Bohler et al. 2007). O<sub>3</sub> also decreases chloroplast size and cell starch content (Oksanen et al. 2004). Chlorophyll and carotenoid levels have been shown to decrease under O<sub>3</sub>, resulting likely in a less active photochemistry and a slower electron transport rate (Bagard et al. 2008). It is, however, unclear whether the slower electron transport rate under O<sub>3</sub> is due to a lower leaf pigment level or a downregulation of PSII activity in order to avoid photooxidative damage. In addition, photochemistry can also be diminished under a long-lasting reduction of stomatal conductance. Such electron transport slowdown would likely limit NAD(P)H production and availability for anabolic process, detoxification and photosynthetate synthesis. The drastic decrease in chlorophyll and Rubisco contents is correlated with accelerated leaf senescence (Miller et al. 1999). Genes involved in senescence and protein turnover are upregulated in *Populus tremuloides* leaves exposed to O<sub>3</sub> (Gupta et al. 2005). Early senescence would restrict the “return on investment” in leaf buildup and therefore drastically affect plant carbon budget.

The reduced net CO<sub>2</sub> assimilation in O<sub>3</sub>-exposed leaves is largely driven by a decrease in gross CO<sub>2</sub> assimilation rate by the Rubisco, but it also results from increased CO<sub>2</sub> losses through an enhanced respiration, as observed in poplar (Bagard et al. 2008; Noormets et al. 2001; Reich 1983), Norway spruce (Küppers and Klumpp 1988), Scots pine (Kellomaki and Wang 1998; Skärby et al. 1987), birch (Matyssek et al. 1997) and beech (Kitao et al. 2009). The

increase in CO<sub>2</sub> efflux from respiration is supported by an enhanced glycolysis, pentose-phosphate pathway and TCA cycle activity (Dizengremel 2001). A central enzyme linking these metabolic pathways, the phosphoenolpyruvate carboxylase, is strongly upregulated in O<sub>3</sub>-exposed leaves of a wide range of species (Dizengremel 2001; Dizengremel et al. 2012). PEPc activity produces oxaloacetic acid, a precursor for malate and pyruvate synthesis, and can therefore support the rising demand in organic acids for TCA cycle decarboxylation or anaplerotic pathways. PEPc-induced pathways could play a central role in O<sub>3</sub> tolerance in providing additional carbon skeletons and NAD(P)H to detoxification processes (Dizengremel et al. 2009). Additionally, the increase in dark respiration under O<sub>3</sub> is likely supported by a higher contribution of alternative pathways of the mitochondrial electron transport chain (Dizengremel 2001) that may be driven by a higher PEP content and subsequent higher pyruvate level (activator of alternative oxidase). The enhancement of the mitochondrial alternative electron transport would help to maintain a high respiratory rate in avoiding any respiratory control through oxidative phosphorylation and therefore contribute to reducing power availability for detoxification and repair of cellular damage (Dizengremel et al. 2009).

### 3.4 Cell wall component biosynthesis in leaves and stems

In addition to inducing a diverse range of defence responses, O<sub>3</sub> has been shown to modify the cell wall. Anatomy analysis revealed thickened cell walls with pectinaceous projections in leaves of deciduous trees showing visible leaf symptoms (Gunthardt-Goerg et al. 1997; Gunthardt-Goerg et al. 2000). These observations suggested strong rearrangements in the cell wall organization and in its component biosynthesis in leaves of trees subjected to O<sub>3</sub>. However, most studies focused on one component, lignin.

O<sub>3</sub> has been shown to stimulate phenylpropanoid metabolism in leaves of many tree species and under different fumigation protocols involving both acute and chronic exposure (Fig. 2). O<sub>3</sub> increased both enzyme activities and related gene transcript levels involved in lignin biosynthesis in *Pinus sylvestris* (Rosemann and Heller 1991; Zinser et al. 1998), *Picea abies* (Galliano et al. 1993a; Galliano et al. 1993b; Heller et al. 1990), *Populus* spp. (Cabané et al. 2004; Di Baccio et al. 2008; Koch et al. 1998; Wustman et al. 2001), *Betula pendula* (Pääkkönen et al. 1998a; Tuomainen et al. 1996) and *Fagus sylvatica* (Jehnes et al. 2007; Olbrich et al. 2005). The responses of the phenylpropanoid metabolism can be both fast (Koch et al. 1998) and substantial (Cabané et al. 2004), and the induction levels were generally correlated with O<sub>3</sub> concentrations (Cabané et al. 2004; Galliano et al. 1993b; Rosemann et al. 1991). Stimulation of the phenylpropanoid pathway was often maintained during the whole period of O<sub>3</sub> exposure (Cabané et al. 2004; Galliano et al. 1993a) and could

even continue after the end of the treatment (Tuomainen et al. 1996). Most of the above results were obtained from trees growing in controlled conditions or open-top chambers, and the same trend was also observed in more natural conditions such as free-air O<sub>3</sub> fumigation facilities (Betz et al. 2009; Wustman et al. 2001). All these results unambiguously demonstrate that the phenylpropanoid pathway is upregulated in leaves under O<sub>3</sub> exposure and is therefore probably involved in defence and acclimation mechanisms. As a consequence of the phenylpropanoid pathway stimulation, the lignin content must increase as a result of O<sub>3</sub> exposure. Indeed, such results were found in leaves of sugar maple (Boerner and Rebbeck 1995), poplar (Cabané et al. 2004) and beech (Betz et al. 2009; Jehnes et al. 2007; Olbrich et al. 2010b). However, the effects of O<sub>3</sub> fumigation on lignin content in leaves were not so clear. Thus, no modifications of lignin content were recorded in western yellow pine (Tingey et al. 1976), black cherry and yellow poplar (Boerner and Rebbeck 1995), loblolly pine (Booker et al. 1996), birch (Oksanen et al. 2005) and holm oak (Baldantoni et al. 2011). These varying results may be explained by potential error due to different techniques (Klason, LTGA, etc.) used to determine lignin content as well as their relative (in)sensitivity (Dence 1992), especially in the case of weak variations between control and treated samples. Another explanation could be species-specific differences in response to O<sub>3</sub> treatment. For example, conifers never showed increased lignin content. Stimulation of the phenylpropanoid pathway could in such cases be associated with a modification of the pool of soluble phenolic and not necessarily lead to increased lignification (Tingey et al. 1976). Nevertheless, the newly synthesized lignin in leaves displayed changes in its structure (Betz et al. 2009; Cabané et al. 2004). Lignin was enriched in carbon-carbon interunit bonds and in H-units indicating the production of a more condensed lignin than usual. Moreover, lignified cells were observed in the mesophyll or epidermis near the necrotic lesions (Cabané et al. 2004) where ROS (H<sub>2</sub>O<sub>2</sub>) were also shown to be accumulated (Pellinen et al. 2002). These results support the idea that stress lignins are synthesized in response to and in defence against O<sub>3</sub> or ROS excess. Due to its scavenging properties, lignin may act as an antioxidant (Blokchina et al. 2003; Dizhbite et al. 2004).

Since the response to O<sub>3</sub> has been extensively studied in leaves, few studies have analysed the O<sub>3</sub> response of stems. This is understandable since leaves show clear O<sub>3</sub>-induced damage and a fast response while stems react much later (Richet et al. 2012). An increase in lignin content was observed in stems of poplar and birch fumigated for 3 years in a free-air fumigation experiment (Kaakinen et al. 2004), but this observation was not maintained after 5 years (Kostiainen et al. 2008). In a recent study, O<sub>3</sub> was observed to repress the phenylpropanoid pathway in poplar wood (Richet et al. 2011), probably as a result of reduced cambial growth. However, the relative cell wall lignin content increased due to an O<sub>3</sub>-

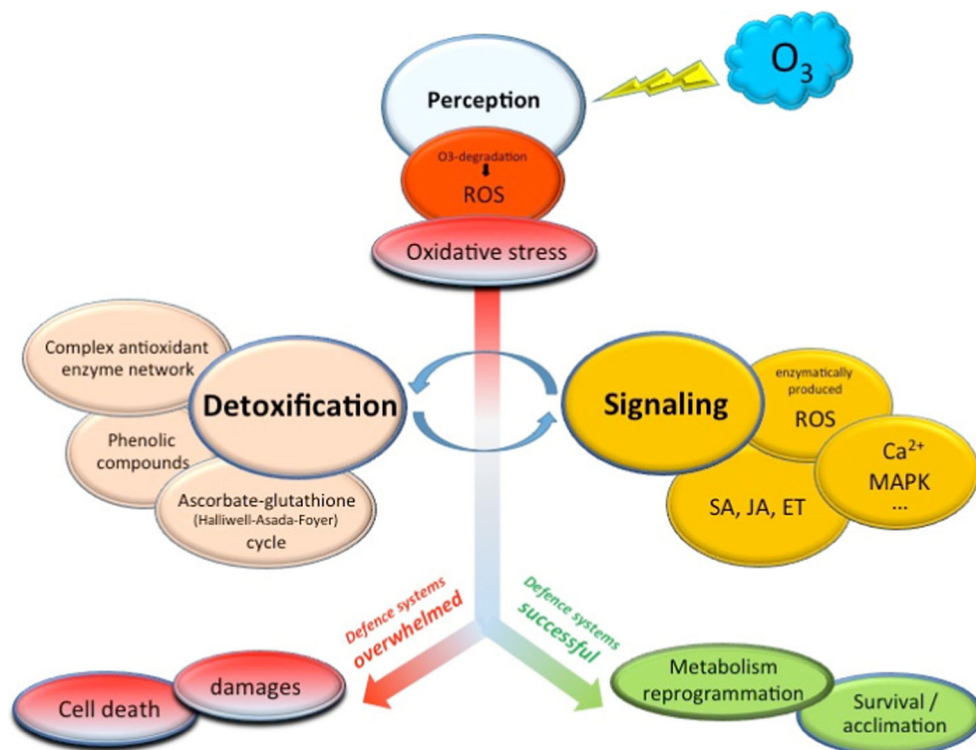
induced reduction in cellulose biosynthesis, thereby modifying the cellulose to lignin ratio (Fig. 2). The stem response seems to correspond to a metabolic adjustment due to the reorientation of the metabolism to stress acclimation in leaves, rather than to a specific defence mechanism. O<sub>3</sub> would not impact directly the stem organ (Richet et al. 2012). It was hypothesized that the modification of the cellulose to lignin ratio in the stem could allow the tree to maintain radial and height growth while minimizing carbon cost (Richet et al. 2011). More detailed analyses are needed to draw definite conclusions.

### 3.5 O<sub>3</sub>-induced signalling in trees

O<sub>3</sub>-signalling events are very quickly initiated during O<sub>3</sub> exposure, leading to plant survival or acclimation. Acclimation first implies O<sub>3</sub> perception and concomitant signalling cascades, ultimately succeeding in re-programming tree metabolism (Fig. 3). In order to elicit and decipher signalling processes, trees were exposed to acute O<sub>3</sub> dose (>150–200 ppb) for few hours. In these conditions, some signalling actors, as calcium or protein kinases, were well characterized in herbaceous plants (Baier et al. 2005; Vainonen and Kangasjarvi 2014; Vaultier and Jolivet 2015). However, O<sub>3</sub>-induced signalling is by far less documented in trees. Direct extrapolation from model herbaceous plants to trees is not always possible, as woody plants may possess their own defence signalling systems (Dizengremel 2001; Koch et al. 2000). In beech (55–60 years old) exposed to an experimental enhanced free-air O<sub>3</sub> setting (up to 150 ppb), the expression level of genes connected with signalling, i.e. ethylene (ET) biosynthesis-related genes *ACC* (1-aminocyclopropane-1-carboxylic acid) *synthase* or *oxidase*, was increased (Jehnes et al. 2007; Olbrich et al. 2010a). Besides, an environmental genomic study performed on 5-year-old trembling aspen exposed to 1.5 times ambient O<sub>3</sub> showed that many genes involved in signal transduction were upregulated, e.g. ET biosynthesis-related genes such as *ACC oxidase* or a gene coding for a mitogen-activated protein kinase (MAPK) (Gupta et al. 2005). MAPK cascades are well-known components of stress-induced signalling pathways, and several studies revealed their activation in response to O<sub>3</sub> exposure in herbaceous plants (Ahlfors et al. 2004; Samuel et al. 2000). In trees, Hamel et al. (2005) showed in hybrid poplar cell suspensions and leaf tissue that O<sub>3</sub> (500 ppb) induced a rapid and transient activation of at least two MAPKs, independently or upstream of both salicylic acid (SA) and jasmonic acid (JA) signalling. However, to date, involvement of the O<sub>3</sub>-activated MAPKs in regulating O<sub>3</sub> sensitivity in poplar is still to be investigated.

Trees respond to O<sub>3</sub> exposure by producing other signalling molecules as ROS (Diara et al. 2005; Moura et al. 2014; Pellinen et al. 1999). The O<sub>3</sub>-induced ROS production and subsequently the formation of necrosis are part of the

**Fig. 3** Schematic representation of interactions between signalling and detoxification processes in the cell of plants exposed to ozone. The acclimation of the plants to this pollutant is upon a tight control of these two processes. *ET* ethylene; *JA* jasmonic acid; *MAPK* mitogen-activated protein kinase; *SA* salicylic acid



similarities shared with early senescence and hypersensitive response elicited by the incompatible plant-pathogen interaction (Moura et al. 2014; Pellinen et al. 1999; Tuomainen et al. 1996). ROS generated either (i) directly from O<sub>3</sub> degradation or (ii) actively enzymatically produced could be crucial components of O<sub>3</sub>-induced signalling (Vainonen and Kangasjarvi 2014).

Concerning hormones, Koch et al. (1998) suggested the role of SA in mediating some O<sub>3</sub>-induced responses in trees. O<sub>3</sub> can induce lesion formation via the activation of programmed cell death, and SA perception is required for the activation of a hypersensitive cell death pathway (Koch et al. 2000). Poplar has higher constitutive levels of free SA compared with herbaceous plant species such as tobacco and *Arabidopsis* (Diara et al. 2005; Koch et al. 2000). In birch and in hybrid aspen, free SA accumulated in response to acute O<sub>3</sub> conditions (Vahala et al. 2003a; Vahala et al. 2003b). Similarly, a significant increase in the conjugated pool of SA was observed in poplar during acute O<sub>3</sub> fumigation (150 ppb for 5 h) (Diara et al. 2005). Optimal SA concentration is required to fine-tune the plant response in order to achieve the maximum stimulation of defence responses with minimal induction of cell death (Diara et al. 2005). JA was also evidenced as an important O<sub>3</sub>-induced signal molecule in trees as acute O<sub>3</sub> exposure increases endogenous JA levels in poplar or in birch (Koch et al. 2000; Koch et al. 1998; Vahala et al. 2003b). JA has at least two different roles in O<sub>3</sub> responses: one in lesion formation and the other in lesion containment (Kangasjarvi et al. 2005). Activation of SA- and JA-

mediated signalling pathways, which may be important in triggering defence responses against oxidative stress, leads to O<sub>3</sub> tolerance (Koch et al. 1998). Lesion propagation and containment in O<sub>3</sub> damage are under the control of ET (Kangasjarvi et al. 2005). O<sub>3</sub> exposure leads to ET release from leaves, in poplar clones (Diara et al. 2005; Kargiolaki et al. 1991), in birch (Vahala et al. 2003b) and in pine needles (Telewski 1992). In aspen, O<sub>3</sub> caused a clear concentration-dependent response in ET evolution (Vahala et al. 2003a). Marked increases in the pool of free ACC, precursor of ET, and in ACC synthase transcripts were also detected in poplar (Diara et al. 2005). The role of ET under chronic (75 ppb) and acute O<sub>3</sub> (up to 200 ppb) was investigated in aspen and silver birch by Vahala et al. (2003a; 2003b). Comparing results obtained on different species, herbaceous or not, Diara et al. (2005) hypothesized a “pro-survival” role for ET and the existence of a threshold below which ET would not trigger lesion development. ET can serve as a mediator of either survival or cell death, depending on the magnitude of synthesis and its temporal pattern (Vahala et al. 2003b). In hybrid aspen, ET accelerated leaf senescence under low O<sub>3</sub>, but under acute O<sub>3</sub> elevation, ET signalling seemed to be required for protection from necrotic cell death (Vahala et al. 2003a).

Of course, complex interactions between hormones are involved in tree response to O<sub>3</sub> exposure. All the three hormonal signalling pathways: SA, JA and ET, were involved in cell death induced by a short exposure to high O<sub>3</sub> concentration (200 ppb for 8 h) in birch (Vahala et al. 2003b). Early high ET production may antagonize the late SA accumulation, and

conversely, increased SA production may downregulate ET accumulation and thus prevent the ET-dependent cell death (Vahala et al. 2003b). In poplar, difference in O<sub>3</sub> sensitivity would depend on differences in the modulation of signal transduction pathways, i.e. the timing and magnitude of SA and ET production, as well as on cross-talking with other signalling molecules (Diara et al. 2005). Further investigations are really needed in trees to unravel this puzzling network triggered by O<sub>3</sub>, particularly under realistic chronic O<sub>3</sub> doses. In these conditions, not only the signalling pathways at the onset of an O<sub>3</sub> episode in natural conditions must be considered but also the impacts, some days or weeks later, when the cellular defence mechanisms are overwhelmed, followed with the beginning of cell death and the occurrence of necrosis (Fig. 3). Finally, it is still necessary to decipher the steps by which defence reactions like the ascorbate-glutathione cycle are under the control of signalling (Fig. 3).

## 4 Impact of O<sub>3</sub> on tree growth, forest productivity and carbon sequestration

### 4.1 Carbon allocation and tree growth

A meta-analysis by Wittig et al. (2007) concluded that significant decreases in both photosynthesis and stomatal conductance of trees under O<sub>3</sub> may negatively affect both carbon sequestration and transpiration. A limitation of extrapolating these data to mature forests is that the estimates are largely based on individual juvenile trees growing in a non-competitive environment, and extrapolation of results from seedlings may not be appropriate for predicting the response of mature trees and forests to O<sub>3</sub> (Chappelka and Samuelson 1998; Ollinger et al. 1997). But recently, Matyssek et al. (2010a) concluded that adult and juvenile trees of pioneer and climax tree species show similar growth sensitivity to chronic O<sub>3</sub> stress, although the underlying response mechanisms may differ. Tree growth and productivity are expected to decrease under O<sub>3</sub> considering aforementioned effects of this pollutant (Fig. 2). Indeed, lower growth and diameter have been observed in a wide range of tree species after long-term exposure to O<sub>3</sub> (Booker et al. 2009; Karnosky et al. 2005; King et al. 2005; McLaughlin et al. 2007; Pretzsch et al. 2010; Tjoelker et al. 1994). Summarizing hundreds of studies, Wittig et al. (2009) reported a decrease in total biomass, leaf area, root to shoot ratio, height and diameter in trees exposed to chronic O<sub>3</sub> concentration (in the range 40–100 ppb) relative to charcoal-filtered atmosphere. However, the intensity of O<sub>3</sub> effects on tree carbon uptake and growth differs depending on tree age, the loss in biomass production following O<sub>3</sub> exposure being greater in young compared to older trees (Herbinger et al. 2005). Since O<sub>3</sub> reduces carbon gain by limiting stomatal diffusion, lowering Rubisco activity,

inducing early senescence and increasing carbon cost for tissue repair and antioxidant synthesis, it drastically decreases source strength and carbon availability for export to sink tissues. Additionally, increased soluble sugar content and carbohydrate retention have been observed in source tissue exposed to chronic O<sub>3</sub> concentration (from 0 to 110 ppb) (Friend and Tomlinson 1992; Grantz and Farrar 1999; Grantz and Yang 2000), suggesting a decrease in carbon export under O<sub>3</sub>. A lower leaf sucrose export and higher carbohydrate level would lead to feedback regulation of photosynthesis and therefore partly explain the reduction in carbon assimilation. Several works have shown a decrease in allocation to roots and root to shoot biomass ratio in response to O<sub>3</sub> (Gorissen et al. 1994; Grantz and Farrar 2000; Grantz and Yang 2000; Rennenberg et al. 1996; Spence et al. 1990). Given that mature leaves preferentially allocate carbon resources to stems and roots (Gordon and Larson 1970; Matyssek et al. 2010b; Rangnekar and Forward 1969), it appears logical that the O<sub>3</sub>-induced early senescence would primarily affect root growth. Such modification may therefore have drastic impacts on tree surrounding rhizosphere and tree survival to environmental constraints such as drought (Agathokleous et al. 2016).

### 4.2 Ozone impact on forest productivity and carbon sequestration: results from free-air fumigation experiments and modelling studies

Fowler et al. (1999) used the 3-D chemistry-transport model STOCHEM (Collins et al. 1997) to simulate the global distribution of tropospheric O<sub>3</sub> from 1860 to 2100. The results indicate that the area covered by forests exposed to >60 ppb increased from 0 in 1860 to 8.3 million km<sup>2</sup> in 1990, i.e. 24 % of global forest area. According to this study, this area could reach 17 million km<sup>2</sup> in 2100, that is half of the projected global forest area, if precursor emission rates remain constant (Fowler et al. 1999). In order to advance from exposure assessment to impact prediction, subsequent modelling studies implemented the linear empirical model of O<sub>3</sub> impact on tree biomass production developed from experimental data in the pioneer work of Reich (1987). The results indicated that biomass production of forest ecosystems would be reduced by 3–22 % in the northeastern USA as an effect of tropospheric O<sub>3</sub> concentrations recorded during the period 1987–1992 (Ollinger et al. 1997). With a similar approach, Subramanian et al. (2015) found that biomass growth of forest trees in Sweden could be reduced annually by 4.3–15.5 % for conifers and 1.4–4.3 % for birch by current O<sub>3</sub> when compared to prehistoric O<sub>3</sub>. Proietti et al. (2016), by combining satellite productivity estimates, O<sub>3</sub> measurement data and impact functions, found that current O<sub>3</sub> concentrations could reduce gross primary productivity of European forests by 0.4–30 %

along a North-West-South-East transect. An alternative, flux-based methodology emerged from the International Cooperative Programme on the effects of air pollution on vegetation (Mills et al. 2013) under the UNECE Convention on Long-Range Transboundary Air Pollution (LRTAP). In this approach, the European Monitoring and Evaluation Program (EMEP) and Rossby Centre Regional Climate (RCA3) models provided O<sub>3</sub> and meteorological input data that fed the Deposition of Ozone for Stomatal Exchange (DO<sub>3</sub>SE) model, which simulates O<sub>3</sub> dry deposition (Büker et al. 2012) and subsequently O<sub>3</sub> stomatal fluxes and phytotoxic O<sub>3</sub> doses (POD<sub>y</sub>) as described in the LRTAP convention manual (2010). O<sub>3</sub> flux-response relationships were applied to calculate biomass and carbon losses for tree species groups and representative species. The data were combined to land cover data and overlain to EMEP and RCA3 resolved grids. Finally, European forest inventory and carbon sequestration datasets were used to calculate absolute carbon losses due to O<sub>3</sub> in Europe. Applying a generic parameterization for deciduous and conifer trees, the authors estimated a reduction of carbon sequestration in the living biomass of trees by 12 % (EMEP input data) to 16 % (RCA3 input data) for the year 2000 as compared to pre-industrial O<sub>3</sub> levels (Mills et al. 2013).

Either exposure- or flux-based modelling studies provide congruent estimates of forest biomass production reduction as an effect of current O<sub>3</sub>. However, the impact functions used rely on O<sub>3</sub> exposure- or flux-response relationships that were derived for relatively young trees (<10 years of age) exposed to O<sub>3</sub> under semi-natural, non-competitive conditions (Karlsson et al. 2007; Reich 1987). Although epidemiological studies suggest that such functions are applicable to mature trees within forest stands (Braun et al. 2010), whether conclusions on the effects of O<sub>3</sub> on forests can be drawn from the extrapolation of results obtained on seedlings remains a matter of debate (Samuelson and Kelly 2001). In this respect, the free-air concentration enrichment experiment Aspen FACE led in Rhinelander (WI, USA) provided concordant results. In this study, young forest stands were subjected to an O<sub>3</sub>-enriched atmosphere (1.5 × ambient) during 11 years from seedling establishment to maturity (Karnosky et al. 2003). The results showed significant reductions in the total biomass of young stands of trembling aspen (−23 %), aspen-sugar maple (−14 %) and aspen-paper birch (−13 %) (King et al. 2005), but these results reflect in main part the impact of O<sub>3</sub> on young trees during their initial, rapid growth stage. Because it was conducted on a mature forest stand composed of 60-year-old trees in a 30-m closed canopy, the free-air O<sub>3</sub> fumigation experiment led in the Kranzberg forest in Germany represents a valuable alternative to FACE systems (Matyssek et al. 2010b). This study showed that stem biomass production of beech trees exposed to elevated O<sub>3</sub> (2× ambient, <150 ppb)

during 8 years was reduced by more than 40 %, but also highlighted the strong influence of environmental factors such as drought on tree responses to O<sub>3</sub>. The combination of stem growth, sap flow velocity and O<sub>3</sub> measurements has been used to investigate the effect of O<sub>3</sub> on mature trees in a mixed deciduous forest in eastern Tennessee (USA) (McLaughlin et al. 2007). This study revealed that daily events of high O<sub>3</sub> exposure (daily maximum hour ≥100 ppb for 1 day or ≥85 ppb for two consecutive days) could decrease stem growth by up to 30 to 50 % over a season, which suggests that episodes of acute exposure might have consequences on tree biomass production that modelling approaches cannot predict. Combined with climate-controlled branch cuvettes, sap flow measurements could represent a valuable alternative to heavy and expensive free-air fumigation experiments for studying the impact of O<sub>3</sub> on adult forest trees (Wieser et al. 2012).

Meta-analyses (Wittig et al. 2009) as well as modelling studies based on response functions estimate that O<sub>3</sub> has a significant impact (−5 to −30 %) on the net primary productivity of forest ecosystems, which might impair their capacity for carbon sequestration (Ainsworth et al. 2012). However, field experiments on mature forest stands have shown that many factors can modulate tree responses to the pollutant (e.g. environmental conditions, stand dynamics and competition) and remain to be considered in modelling approaches. The main challenge of stomatal deposition models is to accurately predict stomatal conductance in response to environmental drivers (Emberson et al. 2000). Recently, Fares et al. (2013a), Hoshika et al. (2011) and Nunn et al. (2010) applied Jarvis's model parameterized with environmental observations with field data, and they were able to predict well stomatal conductance. The recent study of Wang et al. (2016), which simulated the O<sub>3</sub> impact on forest composition and ecosystem dynamics over 500 years, indicated that elevated O<sub>3</sub> could even lead to an increase in forest productivity due to diversity change and compensatory processes at the community scale.

## 5 Combination of O<sub>3</sub> and other environmental factors

The interaction of O<sub>3</sub> with other abiotic or biotic factors can first be considered through its combination with other pollutants, which can occur simultaneously or sequentially. A review on tree exposure to pollutant mixtures showed that the observed responses were highly variable according to tree species, age, genotype, composition of rain solution and soil type (Chappelka and Chevone 1992). In a context of global change, a range of constraints, including high CO<sub>2</sub>, increased temperature, altered precipitation and drought episodes, will also affect trees exposed to O<sub>3</sub> pollution episodes. A first interaction has been investigated in a context of rising atmospheric CO<sub>2</sub> concentrations, which

generally results in a reduction of stomatal conductance (Ainsworth et al. 2012). Thus, based on simulated stomatal O<sub>3</sub> uptake, the flux of O<sub>3</sub> entering the leaves would be decreased (Klingberg et al. 2011; Sitch et al. 2007). However, the reduced stomatal conductance on a long-term exposure under high CO<sub>2</sub> seems uncertain considering contrasting results from FACE experiments (Uddling et al. 2010). The stage of plant and stand development, as well as the consideration of overstorey/understorey species, would influence the stomatal response. Drought also reduces stomatal conductance, with a potential subsequent restriction of O<sub>3</sub> effects. In fact, the protective effect of drought would only occur in severe drought conditions while under low water restriction the damage caused by O<sub>3</sub> appeared additive (Matyssek et al. 2006; Pääkkönen et al. 1998b). However, there are clear inter- and intra-specific differences in response to the combination of drought and O<sub>3</sub> (Dixon et al. 1998). The protective effect of drought may be the result of stomatal exclusion of O<sub>3</sub> but also the induction of defence reactions (Matyssek et al. 2006). Other works underlined that the combined effects of drought and O<sub>3</sub> could also decrease the antioxidant capacity of leaf cells in a higher extent than with the constraint alone, leading to a higher susceptibility to oxidative stress (Wellburn et al. 1996). Thus, in these conditions of combined stresses, the fine-tuning between O<sub>3</sub> uptake and defence capacity appears crucial (Matyssek et al. 2006). Because of changes in plant metabolism, carbon assimilation and allocation and chemical leaf defences, O<sub>3</sub> may also modify plant responses to biotic factors as insect or plant pathogens. Trees can be weakened by O<sub>3</sub>, promoting biotic attack (Chappelka and Chevone 1992; Dowding 1988). However, these studies also underlined that a great number of factors linked to the environment, the host plant or the pathogen may modulate the O<sub>3</sub>-host-pathogen interactions and their consequences. Some works attempted to better understand these interactions and identify underlying biochemical and physiological mechanisms. Using Aspen FACE, investigations were carried out on the impact of both increased CO<sub>2</sub> and O<sub>3</sub> concentrations on forest insects in a poplar canopy (Percy et al. 2002). In response to O<sub>3</sub>, the production and chemical composition of leaf cuticular waxes and the concentrations of protective compounds in the leaf were modified. These modifications may explain an increase in rust infection under O<sub>3</sub> while a higher abundance of leaf-chewing insects as well as aphids have been observed, an effect alleviated by the combination of CO<sub>2</sub> and O<sub>3</sub>. Changes in leaf morphology and composition, including phenolic content, induced by O<sub>3</sub> appeared to be determinant in explaining the larger deleterious effect of an herbivorous insect on O<sub>3</sub>-treated aspen trees (Freiwald et al. 2008). Finally, the O<sub>3</sub> impact on carbon allocation must be also considered as a factor that

modulates the beneficial interaction of fungi and plants via mycorrhization (Nikolova et al. 2010; Pritsch et al. 2009), an aspect that needs to be clarified.

## 6 Concluding remarks

It is now widely accepted that O<sub>3</sub> is able to reduce the growth of forest trees. Even though the results could differ in intensity between experiments conducted in phytotrons or in FACE systems, between young and mature trees, the major impact is a decreased carbon assimilation resulting in a reduced carbon sequestration. The precise mechanisms leading to this loss of available carbon are still to be totally deciphered, notably the respective roles of stomatal resistance and detoxification processes, which determine the tree sensitivity. In addition, the mechanisms underlying the O<sub>3</sub> transduction signal begin to be clarified in acute conditions while it remains fragmented in chronic O<sub>3</sub> exposure, particularly for trees. Another consequence of the O<sub>3</sub> effect is the modification of carbon allocation to the different organs of the tree, which can lead to changes in wood quality and quantity. In this context, alterations of the functioning of forest ecosystems need also to be better investigated by taking into account the interactions between O<sub>3</sub> and other abiotic (CO<sub>2</sub>, water, temperature, etc.) and biotic stresses. Emphasis should be put on water availability, a factor already mentioned as determinant to scale O<sub>3</sub> effects from seedlings to forest trees (Samuelson and Kelly 2001). Contradictory reports of antagonistic or synergetic effects of O<sub>3</sub> and CO<sub>2</sub> or drought clearly show that additional research effort is needed. The advancement of an integrative knowledge of O<sub>3</sub> impact from the leaf cell to the tree level will allow a significant improvement of the existing models of O<sub>3</sub> risk assessment on ecosystems.

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