Effect of land use changes on air quality: Impacts of urbanization, urban vegetation, and agriculture

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ABSTRACT

Urbanization transforms natural landscapes into impervious surfaces, altering local climate and air quality. Greening strategies are increasingly adopted to mitigate these effects, yet their efficacy depends on complex interactions among land use, urban form, geography, and climate. Using an air quality model with an urban canopy scheme, we assess how changes in land use—urban expansion, agriculture, and urban parks—affect urban climate (temperature, wind) and chemical processes (dry deposition, biogenic emissions), ultimately influencing air pollutants (NO_2, O_3, VOC, and PMs). Applied to future land-use scenarios from the Urban Master Plan of the Metropolitan Area of Barcelona, our results show that increasing urbanization elevates surface temperatures and enhances evening O_3 levels by up to 8%. Replacing urban areas with agriculture or parks reduces O_3 (up to 10%) via increased evapotranspiration and enhanced dry deposition, yet raises NH_3 (up to 90%) and aerosols (up to 12%) from fertilizers. Urban greening increases O_3 (up to 5%) and SOA (up to 14%) from biogenic VOCs. Irrigation further boosts plant deposition, raising NO_2 and VOCs (up to 10%), and lowering O_3 by 6%. Our findings highlight the trade-offs of urban greening and underscore the need for integrated planning to improve air quality.

INTRODUCTION

Presently, more than half of the world's population resides in urban areas, and this number is expected to increase to 68% by 2050(1). Consequently, cities have become hubs of air pollution due to transport, heating, and cooling, causing 467,000 premature deaths per year(2) in Europe. As urban areas expand, particularly with the growth of residential, commercial, and recreational zones, traffic density tends to rise. Concurrently, industrial facilities, which are major sources of air pollution, often increase with urbanization as well. Besides, the increase in temperatures and heatwave events in cities(3) further intensifies the generation of pollutants(4), which not only has local consequences but also affects regions globally(5; 6; 7).

In most cities, accelerated urbanization continues to replace natural surfaces with impervious surfaces, leading to modifications in surface physical properties. Concrete buildings and asphalt-covered roads change land surface radiative fluxes, such as surface albedo, emissivity and thermal properties, including heat capacity and thermal conductivity. This effect is especially intensified in cities, where a high fraction of urban building materials have a far greater capacity to absorb heat during the day and radiate it back at night than vegetated areas do, giving rise to the urban heat island (UHI) effect (8). These changes in surface physical properties affect local temperature and convection and, in turn, influence air quality, which has been widely acknowledged in previous studies(9; 10; 11; 12; 13). For example, Li et al., (2019)(9) found that urbanization decreases ventilation and consequently increases ozone (O_3) and fine particulate matter ($PM_{2.5}$) concentrations. Civerolo et al. (2007)(11) found that the increase in temperature of 0.6 ºC due to urbanization in New York City contributed to an increase in the maximum 8h O_3 concentration of 6 ppb. Similarly, Chen et al. (2018)(13) found that modification of rural to urban land surfaces in Beijing led to increases in near-surface air temperature and a planetary boundary layer height (PBLH), which in turn led to increases (9.5 ppb) in surface O_3 and decreases (16.6 µg m^{-3}) in $PM_{2.5}$ concentrations. Furthermore, the study of Martori et al., 2022(14) found a positive relationship between higher building density and greater pollution in the city of Barcelona using spatial data analysis and econometric models. Consequently, cities are adopting strategies to significantly increase urban greenness to reduce the effect of the built environment and improve air quality. Cities such as Paris(15), Oslo(16), Copenhagen(17), and Barcelona(18) have committed to incrementing green strategies as a way to reduce air pollution and mitigate the effects of climate change.

Vegetation has been shown to directly improve air quality by diluting emissions(19), creating barriers to traffic emissions(20), and capturing particulate matter (PM) pollution and ground-level nitrogen dioxide (NO_2) and O_3 through deposition on leaves and transfer through plant stomata(19; 21; 22; 23). Indirectly, urban vegetation reduces surface temperatures and consequently impacts photochemistry, reducing chemical reaction rates, especially for secondary air pollutants, such as O_3(24; 25). Furthermore, temperature regulation by vegetation minimizes energy use, thereby reducing the emissions generated by heaters and coolers(26). However, there are cases in which greening can deteriorate air quality.

For example, biogenic emissions of volatile organic compounds (BVOCs) can cause an increase in O_3 concentration and secondary organic aerosols (SOAs)(27; 28). Vegetation has also been shown to change convection behaviour in cities, resulting in the concentration of pollutants in certain neighbourhoods(29). Simulations of streets with trees show that while trees reduce their temperature, they can also increase pollutants by reducing the wind velocity in the street and limiting its dispersion(30). The same study showed that on streets close to green areas, where gases emitted by road traffic accumulate, trees can increase NO_2 concentrations by up to 20% and reduce O_3 concentrations by 30%, and in green areas, with no traffic emissions, trees can reduce NO_2 concentrations by 5%, with modest increases in O_3 of up to 2%(30). Indeed, where and how to implement vegetation in urban areas is complex, and trade-offs need to be carefully examined. Another study revealed that urban greening and white roofs decrease O_3 concentrations due to a reduction in temperature but also increase primary pollutants due to reduced vertical mixing(25).

Urban and peri-urban agriculture is a type of urban greening that is promoted in cities to increase vegetation and self-sufficient food supplies(31) ; however, it has not been studied as extensively as other types of green infrastructure in terms of temperature and air quality variability. Extensive green areas such as crop fields could reduce surface temperature and affect air quality indirectly and directly by removing pollutants through deposition. Irrigation can modify meteorological conditions such as soil moisture, surface air temperature, humidity and PBLH(32; 33), affecting photochemical reaction rates and increasing the concentration of hydroxyl radicals (HO)(32), the dominant tropospheric daytime oxidant, responsible for the removal of many pollutants, thereby maintaining air quality. Furthermore, strategies of nutrient circularity in cities can reduce the consumption of mineral fertilizers and their related impacts compared with those of conventional crop production systems (34; 35; 31). However, producing crops locally also has an associated cost in terms of air quality. Ammonia (NH_3) emissions are generated through nitrification and denitrification reactions when manure and fertilizers are applied to fields(36), causing lung damage and death and are known to contribute significantly to the formation of ammonium aerosols (NH_4+), which also negatively impact air quality and health(37). In fact, ammonia accounts for the formation of 50% of all $PM_{2.5}$ air pollution in Europe(38). Another relevant source of agricultural air pollution is methane (CH_4), which results from fermentation, manure management, rice cultivation and residue burning(39). In addition to being a greenhouse gas, CH_4 oxidizes to form ground-level ozone(40).

This study aims to provide new knowledge to the study of urban air quality by analysing the impacts of land-use change on urban air quality, with a special focus on urban greening and agriculture. More specifically, we hope to elucidate the role of biogenic sources and dry deposition sinks, the increase in NH_3 concentrations and aerosol formation from agriculture, and the indirect effects of irrigation on urban photochemistry and O_3-NO_X-VOC chemistry. To do so, we adapt the air quality model WRF-Chem coupled with an urban canopy scheme BEP-BEM to study changes in meteorology (temperature, PBLH, and wind speed), chemical processes (dry deposition and biogenic emissions) and air quality under different land-use scenarios. The Metropolitan Area of Barcelona (AMB) serves as a relevant case study given its recent proposal to expand peri-urban agriculture and illustrates how other cities can evaluate urbanization, peri-urban agriculture and urban parks.

RESULTS

Case study

The Metropolitan Area of Barcelona (AMB), located in Catalonia (Spain) in the northeastern part of the Iberian Peninsula, serves as a case study to investigate the impacts of land-use change on urban air quality (see Fig. 1a). The AMB has a complex orographic territory with different altitudes and several peripheral mountain ranges and depressions(41). The AMB is characterized by a Mediterranean climate, with dry and hot summers and clear skies. With more than 3 million people, the AMB is the most populated urban area on the Mediterranean coast and comprises 36 municipalities including the city of Barcelona. The city of Barcelona annually reports some of the highest air pollution levels in Europe; in particular, the average exposure of the population to the city during 2022 was 15 µg m^{-3} for $PM_{2.5}$ particles and 31 µg m^{-3} for NO_2, values that triple the recommended guideline value established by the WHO(42). Furthermore, O_3 precursors (NO_x- nitrogen oxides and VOCs) generated in the AMB are transported inland, generating O_3 and affecting the rural area of the Vic Plain, which results in frequent exceedances of hourly O_3 levels (180 µg m^{-3})(43; 5).

Heatwave (HW) events, with prolonged high-temperature and stagnant conditions that worsen air quality(4), are becoming more frequent and severe in AMB(3). Because of the strong link between

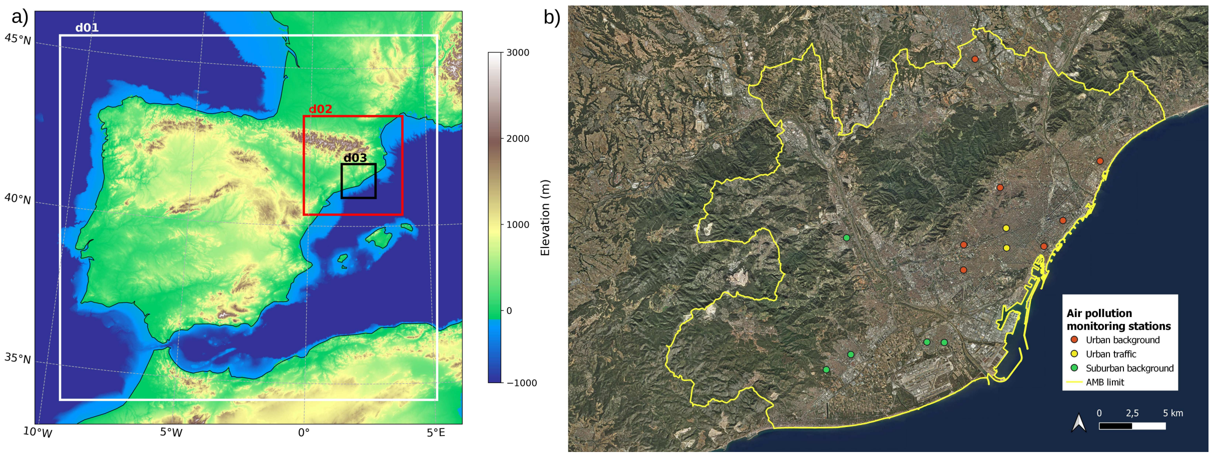

Figure 1. a) Domains of the WRF-Chem model for this study with 9, 3 and 1 km grid size for d01 (Iberian Peninsula), d02 (Catalonia) and d03 (AMB), respectively. b) Detailed map of the AMB indicating the urban and suburban air pollution monitoring stations (Xarxa de Vigilància i Previsió de la Contaminació Atmosfèrica, XVPCA).

pollution formation and high temperatures, we selected a heatwave period, 4th-7th July 2015, with maximum temperatures reaching 40 °C in the interior of the AMB and 35 °C on the coast. The minimum temperatures were also considerably high during the heatwave, reaching above 25 °C in some urban neighbourhoods. The HW was provoked by a trough situated northwest of the Iberian Peninsula advected warm air from the south/southwest, which, in addition to the strong irradiation received due to the stability at the surface level, resulted in increased temperatures(3).

According to the URBAG map provided by Mendoza Beltran et al. (2022, 2023)(44; 45), the AMB land cover distribution is as follows: 42% of the area is made up of forests and shrublands, whereas 45% is made up of built-up areas such as roads, urban parks, and urban areas. Currently, only 8% of the territory is covered by agricultural land, following a significant reduction over the previous 70 years (46). The local food supply can be provided by peri-urban agriculture, which may be more sustainable than typical agricultural systems that are located far away. Therefore, peri-urban agriculture is becoming increasingly important for urban areas such as the AMB.

Impacts of urbanization, agriculture and urban parks on air quality

We study the impacts of urbanization, agriculture and urban parks on chemical processes (biogenic sources and dry deposition sinks) and air quality by comparing the three scenarios (URB, AGR and PARK) against the reference scenario (REF), as shown in Figures 2-6. We first show the average changes in dry deposition velocities of NO_2 and O_3 (cm/s) between 6 and 21 UTC when the deposition fluxes are the highest because the atmosphere is usually less stable during the daytime (Figure 2). Next, Figure 3 shows how the variation in biogenic emissions (mol/km^2/h) varies depending on landuse during the day (6-21 UTC) which is when emissions are most significantly influenced by temperature and light. Finally, we show the air quality results as the difference in concentrations for the morning (6-8 UTC) and evening (19-21 UTC) hours in Figures 4 and 5, respectively, when the most significant changes in pollution concentrations are found. The results for a period before the heatwave (1-3 July 2015), with temperatures 3-4 degrees lower than during the heatwave, are shown in the Supplementary (Figures S1 and S2). Finally, Figure 6 summarizes changes in these chemical processes and in air quality for each scenario. In addition, Figure S3 in the Supplementary shows the air quality for the REF scenario.

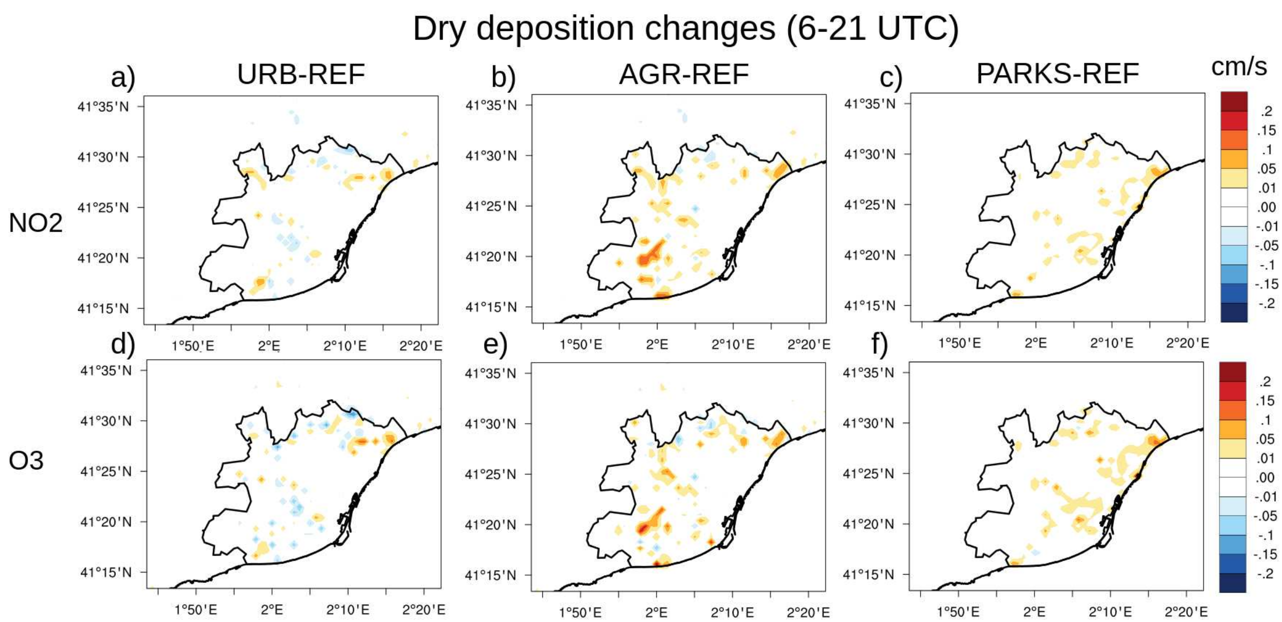

Figure 2. Dry deposition average changes due to urbanization (URB-REF), agriculture (AGR-REF) and urban parks (PARK-REF). For the dry deposition analysis the 6-21 UTC has been chosen because are the time of the day that these processes are more significant.

Impacts of urbanization

We found that dry deposition velocities decrease with urbanization (Fig. 2a,d). In grids with more than a 10% increase in the urban built environment, we found a decrease in the dry deposition velocities of up to 15% and 20% (third quartile) for NO_2 and O_3 (Fig. 6a), respectively. Biogenic emissions were also lower for the URB scenario, as expected (Fig. 3a,d). We found a reduction of up to 50% and 45% (third quartile)

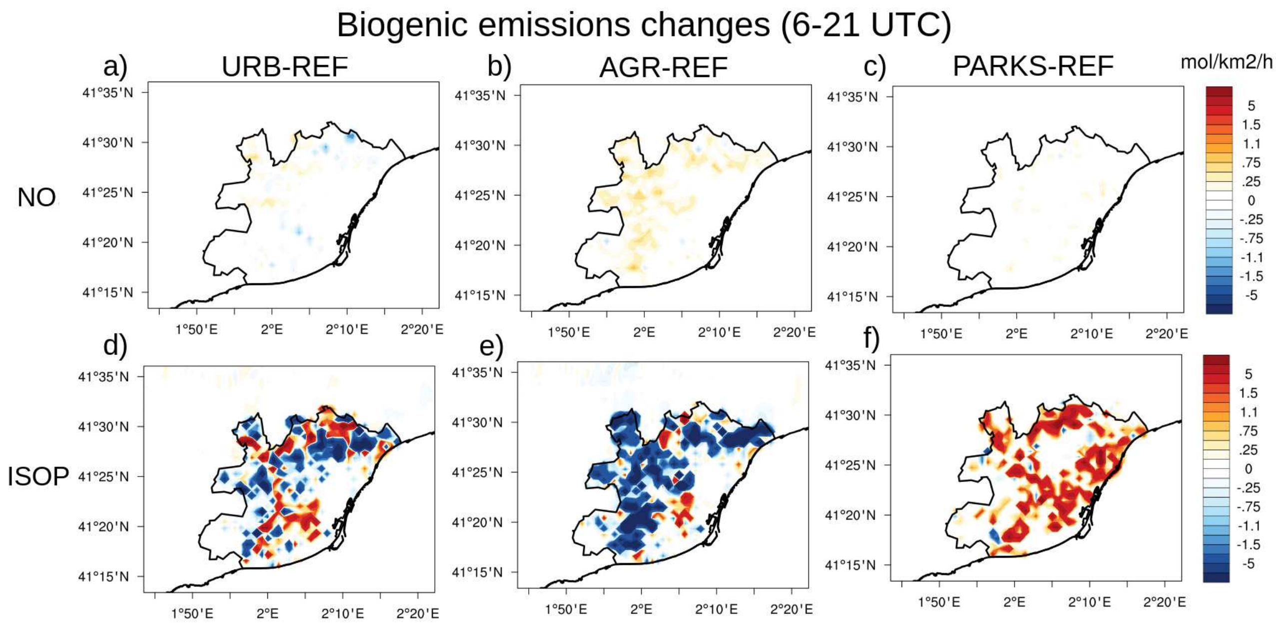

Figure 3. Biogenic emissions average changes due to urbanization (URB-REF), agriculture (AGR-REF) and urban parks (PARK-REF). For the biogenic emissions analysis the 6-21 UTC has been chosen because are the time of the day that these processes are more significant.

for NO and isoprene (Fig. 6d), respectively. Both dry deposition and biogenic emissions depend strongly on the surface type being low over urban built areas compared with areas with vegetation.

In terms of concentrations, we found increases in NO_2 (1.5 µg m^{-3}, 8%), VOC (5 µg m^{-3}, 10%), PM_{10} (up to 1 µg m^{-3}, 12%) and $PM_{2.5}$ (up to 1 µg m^{-3}, 10%) in the north- eastern part of the AMB over areas where the land-use evergreen needleleaf forest has been replaced by the large low-rise urban category during the morning hours (see Figs. 4 and S4 in the Supplementary). We attribute this to lower morning temperatures (0.5 °C) in the URB scenario than in the REF case because of the lower heating rate in the urban environment than in the to rural environment, which causes the urban temperature to become lower than the rural temperature in the morning(78; 79). However, in areas where croplands have been replaced by urban areas, we see an increase in O_3 concentrations of 3% (2 µg m^{-3}) following an increase in temperature (0.5 °C) and in the PBLH (50 m) in the morning. During the evening, when the UHI effect is the strongest, there is a clear decrease in the primary pollutants NO_2 (up to 7 µg m^{-3}, 25%) and VOCs (up to 25, 25%) over new urban areas (Figs. 5 and 6) following an increase in temperature (1 °C), and PBLH (120 m) and a reduction in the wind speed, which consequently increases the O_3 concentration (up to 8 µg m^{-3}, 15%). Fig. S5 in the Supplementary provides a detailed meteorological analysis.

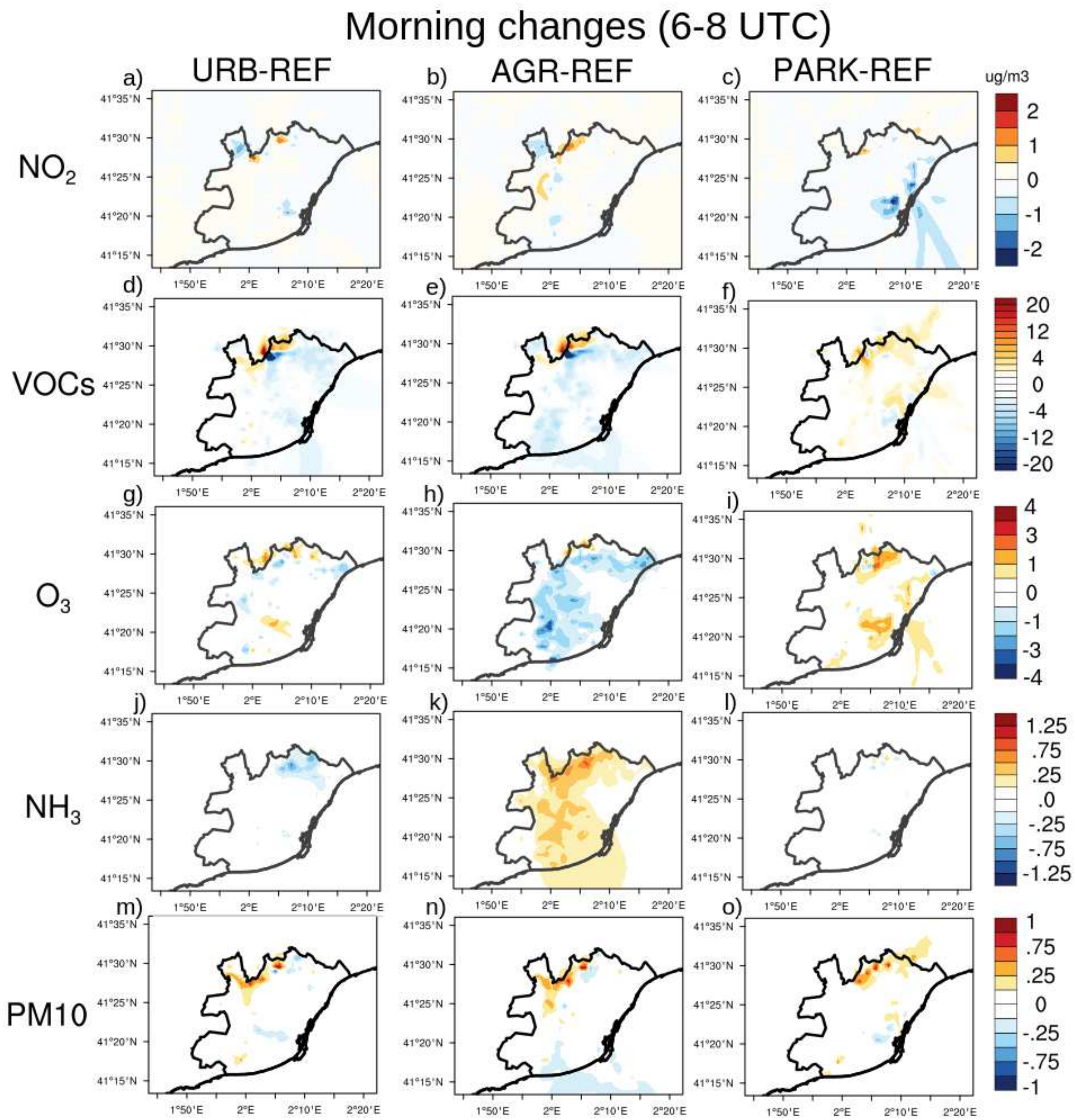

Figure 4. Morning (6-8 UTC) air quality average changes (in µg m^{-3}) due to the increase of urbanization (URB-REF), agriculture (AGR-REF) and urban parks (PARK-REF).

Impacts of agriculture

The expansion of agricultural fields in the AGR scenario results in an increase in the dry deposition velocity, as shown in Figure 2b,e. The grid cells of the inner-most domain (1 km × 1 km) with more than a 10% increase in agricultural area were found to have higher NO_2 and O_3 dry deposition velocities up to 48% (third quartile) and 8% (third quartile), respectively than those than in the REF scenario (Fig. 6b), where velocities ranged from 0.1-0.2 and 0.3-0.6 cm/s, respectively. This has caused an increase in pollution removal over new agricultural fields. Our simulations estimate an increase in biogenic NO (up

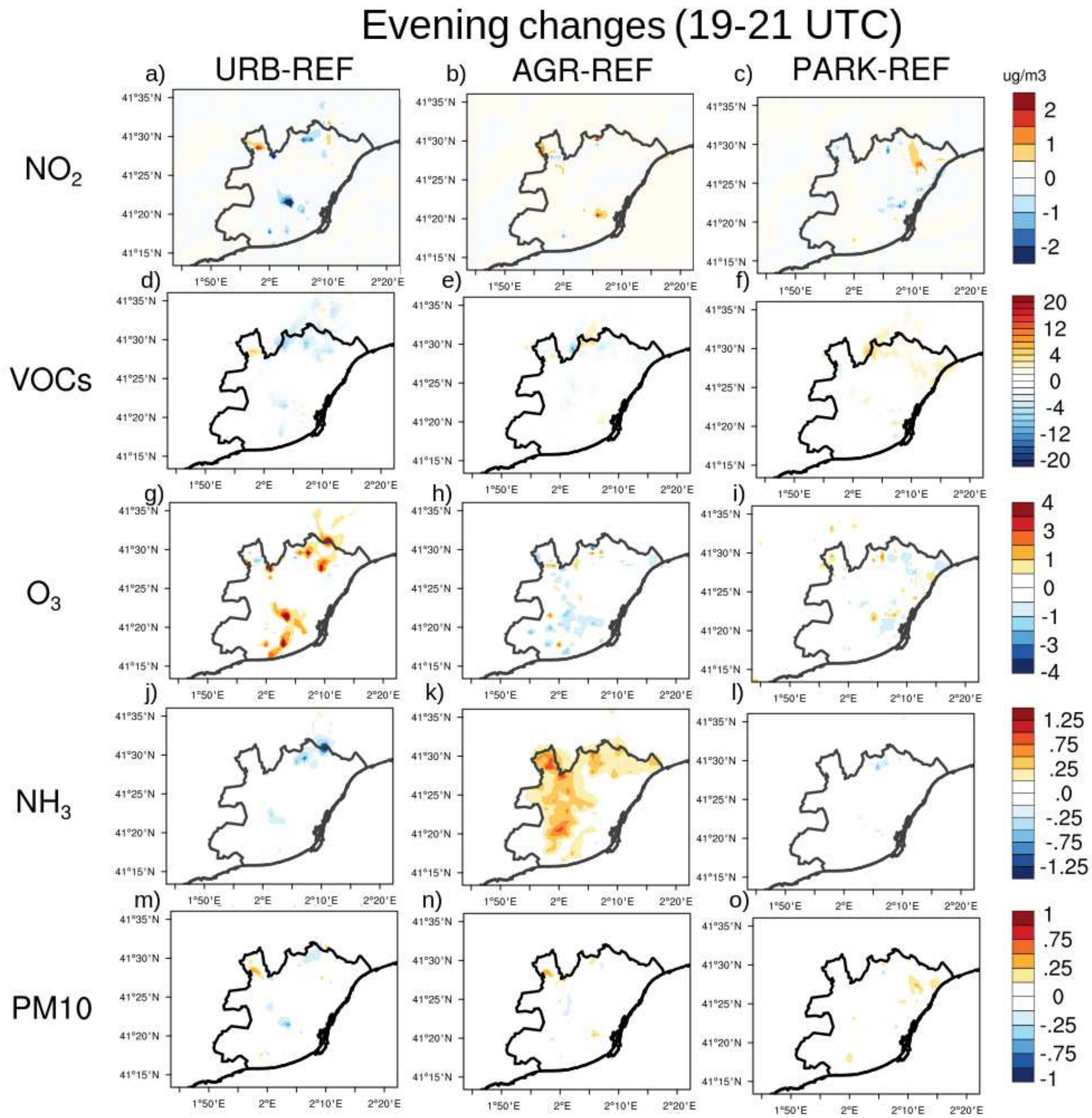

Figure 5. Evening (19-21 UTC) air quality average changes (in µg m^{-3}) due to the increase of urbanization (URB-REF), agriculture (AGR-REF) and urban parks (PARK-REF).

to 105%, third quartile) and a decrease in biogenic isoprene emissions (up to 50%, third quartile) where the forest category (in REF) has been replaced by cropland (in AGR) (Figs. 3b,e and 6e). This result is consistent with the fact that agricultural crops where large quantities of fertilizers are used, produce significant quantities of nitrogenous gases(80; 81). Anthropogenic NO and NH_3 emissions (Fig. S6 in the Supplementary), which come from agricultural crop machinery and fertilizers, increased significantly (up to 35% and 340%, respectively).

Changes in NO_2 concentrations (-1 to 3 µg m^{-3}, -5 to 45%) are observed over new agricultural areas

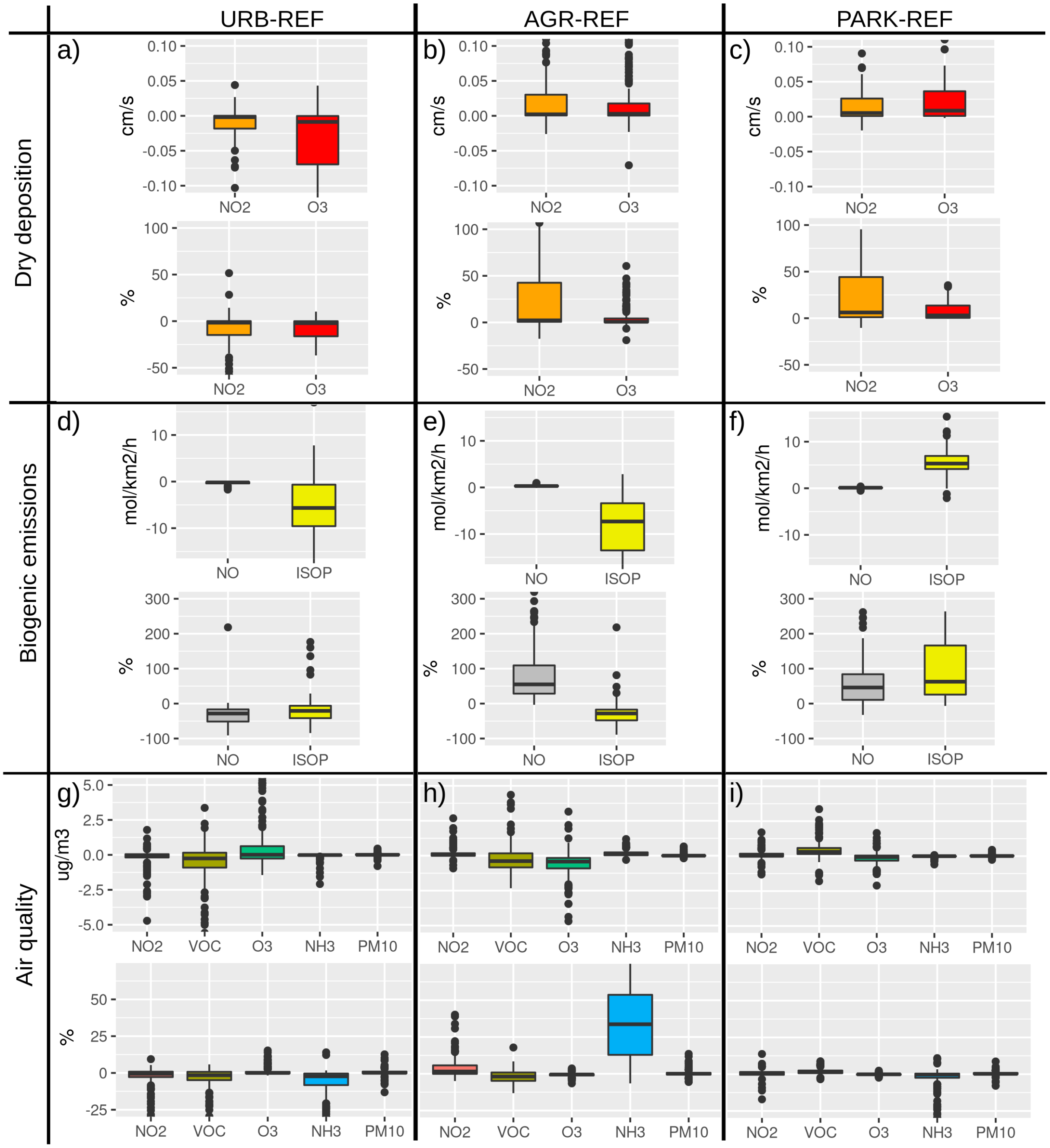

Figure 6. Dry deposition, biogenic emissions, and air quality changes due to urbanization (URB-REF, grids that increase more than 10% in urban areas), agriculture (AGR-REF, grids that increase more than 10% in agricultural areas) and urban parks (PARK-REF, grids that increase more than 10% in urban parks). Bars indicate the 25th–75th quartile interval and points indicate outliers for model hourly data. For the dry deposition and biogenic emissions analysis the 6-21 UTC has been chosen. For the air quality analysis the morning (6-8 UTC) and evening (19-21 UTC) hours are considered.

(Figs. 6, 4 and 5), with the enhancements mainly due to the increase in agricultural emissions. We found a general decrease in VOC concentrations (up to 2.5 μg m^{-3}, 12%) resulting from agriculture replacing forested areas that reduce biogenic VOC emissions. Low levels of VOC (O_3 precursors), together with a decrease in the surface temperature, decrease O_3 formation and its concentrations throughout the day (up to 5 µg m^{-3}, 10%) over new agricultural lands. Over the same agricultural areas, the results of the simulations revealed an increase in NH_3 concentrations (up to 1.5 µg m^{-3}, 90%) that further enhanced the production of particulates (such as ammonium nitrate NH_4+), and, consequently, we found increases in PM_{10} and $PM_{2.5}$ concentrations of up to 0.75 and 1 µg m^{-3} (10 and 12%), respectively, in the morning (see Fig. S7 in the Supplementary). Fig. S8 in the Supplementary provides a detailed meteorological analysis.

Impacts of urban parks

Dry deposition by vegetation is an important process for the removal of air pollutants. Consequently, doubling urban parks from 21.18 km^2 to 41.43 km^2 significantly increases dry deposition velocities for NO_2 (up to 48% for the third quartile) and O_3 (18% for the third quartile) during the day (Figs. 2c,f and 6c), contributing to the removal of 0.4 kg/day and 1.3 kg/day for NO_2 and O_3, respectively, during the heatwave period over new urban parks. As expected, there was a clear increase in biogenic emissions from the vegetation of the new urban parks, especially for isoprene (up to 160% for the third quartile) and NO (up to 90% for the third quartile), as shown in Figure 3c,f and 6f. Compared with that in the reference scenario, the increase in biogenic NO in the PARK scenario (up to 90%, third quartile) was smaller than in the AGR scenario (up to 105%, third quartile).

With respect to primary pollutants such as NO_2, we observe a decrease in the morning (1.5 µg m^{-3}, 3%), due to increased deposition, and changes in the evening (-1 to 2.5 µg m^{-3}, -3 to 10%), when photochemical activity ceases. In the case of VOCs we observed a general increase (up to 4 µg m^{-3}, 10%) over the entire AMB domain, explained by the significant increase in biogenic VOC emissions with urban areas replaced by vegetation. Ozone concentrations slightly increase all day over the AMB domain (up to 2 µg m^{-3}, 5%), which coincides with the increase in its production due to more biogenic VOC sources from new vegetation. However, in the evening, O_3 decreases (up to 2 m^{-3}, 5%) in some areas because of a reduction in surface temperature (see Fig. S9) and increased NO titration. Note that in the days leading up to the heatwave, when temperatures are 3-4 degrees lower than during the heatwave, ozone levels are lower in the PARKS scenario compared to the REF (see Figures S1 and S2 in the Supplementary). This suggests that the removal of ozone by the deposition process dominates over the ozone production from biogenic emissions in the newly vegetated areas during that period.

More BVOC emissions as a result of increased vegetation biomass, increase the production of SOA (up to 0.75 µg m^{-3}, as shown in Fig S10. in the Supplementary), and consequently, we found increases in the PM10 and PM2.5 concentrations (1 and 0.75 µg m^{-3}, 8 and 14%, respectively) over the northern part of the AMB in areas where many croplands (up to 60% in some REF cells) have been replaced by urban parks (in PARK) during the morning (Figs. 4 and S11 in the Supplementary). However, we also found PM reductions during the day (see Fig. 6, up to 0.75 µg m^{-3},10%), mainly due to more deposition over new vegetation.

Impacts of irrigation

Irrigation has direct effects on meteorological conditions such as soil moisture, surface temperature and wind speed (see Fig S12 in the Supplementary) and, consequently, has indirect effects on air quality. Here, we focus on the AGR scenario with and without irrigation. The results for the other scenarios are found in the Supplementary (Fig. S13). During the morning the soil regulates the temperature due to the increase in evapotranspiration and heat capacity, which results in cooler ground surfaces (up to 0.8ºC) over irrigated areas, which first decreases instability and turbulence (PBL decreases up to 70m), leading to a weakening of the vertical mixing of primary pollutants. At night, wet soil has a greater heat capacity than dry soil does, resulting in a slight increase in temperature (0.3ºC), as reported by other authors(82). Wind speed also experiences little changes due to irrigation with a general tendency to decrease, particularly in the morning and afternoon, and increase during the evening.

Irrigation enhances surface wetness and favours deposition at the surface of plants(83). Consequently, the dry deposition velocities for NO_2 and O_3 increase during the day, with an increase of more than 15% in the case of NO_2 and more than 40% in the case of O_3 over several grids (see Fig. 7, third quartile). In the case of biogenic emissions, following a reduction in surface temperature, we found a slight decrease during the day, with reductions increasing to 3.5% in the case of NO and up to 3% in the case of isoprene. Other changes in the biogenic emissions may be linked to variations in the wind speed and PBLH (see Fig

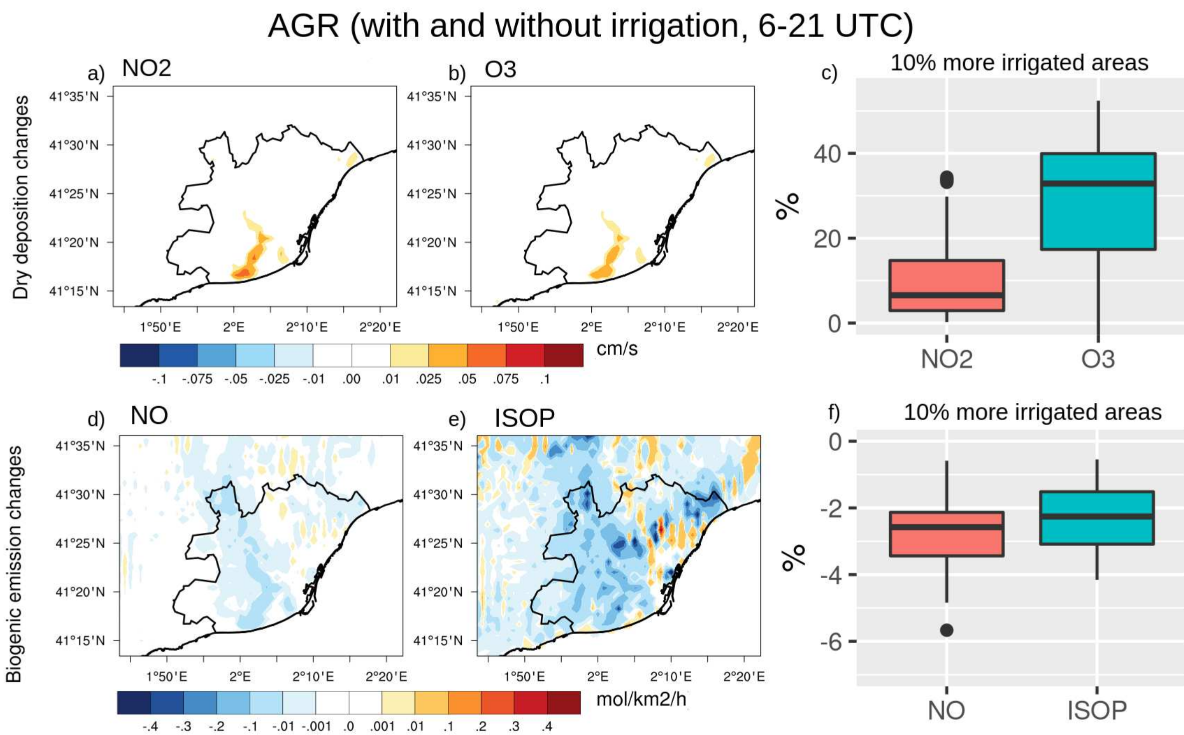

Figure 7. Dry deposition and biogenic emissions changes due to irrigation. For this analysis the 6-21 UTC has been chosen because are the time of the day that these processes are more significant. The third column shows dry deposition and biogenic emissions changes (c, in %) in grids that the irrigation increase more than 10%. Bars indicate the 25th–75th quartile interval.

S13 in the Supplementary).

Although dry deposition velocities increase due to irrigation, surface primary pollutant concentrations within the irrigation zone increase by 2 μg m^{-3} (8%) for NO_2 and 5 μg m^{-3} (9%) for VOCs (Fig. 8) in the morning. This increase in primary pollutants is due to weak vertical mixing, which affects pollutant dilution. In the case of VOCs, a change in the meteorology, changes the distribution of VOCs concentrations over the northern part of the AMB, where more anthropogenic VOC emissions are located. However, we observed a decrease in secondary pollutants such as O_3 up to 5 μg m^{-3} (7%) on irrigated land due to a decrease in O_3 production (cooler surface) and an increase in its dry deposition velocity. We did not observed significant changes in air quality during the remainder of the day. Changes in PM_{10} and $PM_{2.5}$ are not significant and therefore, are not shown here.

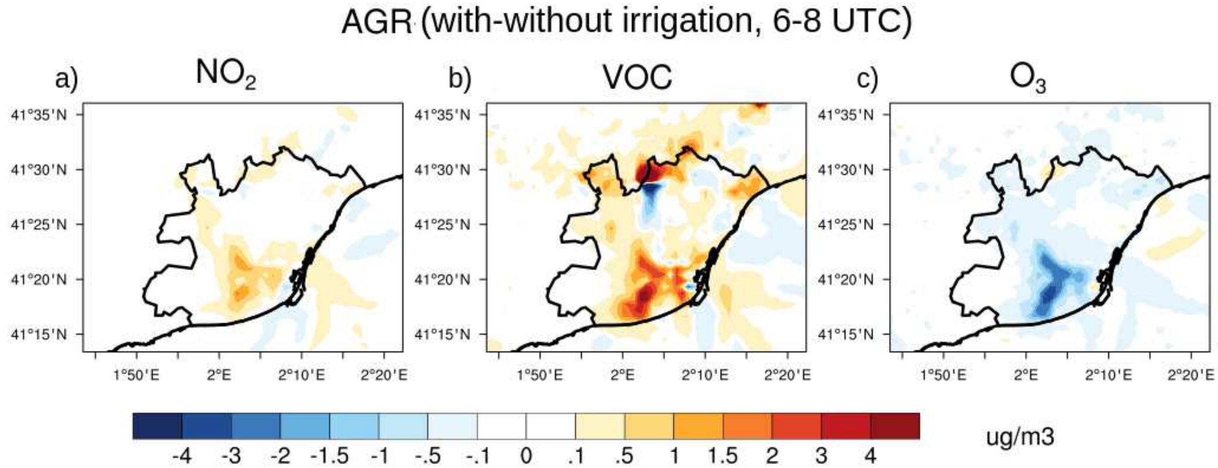

Figure 8. Morning air quality changes (in µg m^{-3}) due to irrigation.

DISCUSSION

Replacing urban areas with agricultural areas or parks improves local air quality

Analysing simulations of various land-use scenarios allows us to explore and better understand how changes in land surfaces directly and indirectly affect the urban climate and air quality. Urbanization alters local atmospheric circulation, resulting in a general increase in temperature and a reduction in wind speed. With these meteorological changes and fewer pollutants removed through dry deposition, the concentrations of secondary pollutants such as O_3 increase. This increase in O_3 levels in our study is relevant given that O_3 can be transported inland to the rural area of the Vic Plain, which already records high O_3 concentrations (more information can be found in the section Case Study). These findings are consistent with several studies that have reported increased O_3 concentrations due to urbanization(84; 9; 11; 13), including a 5 ppb rise in Chengdu, China (9), a 6 ppb increase in New York City, USA (11), a 9.5 ppb increase in Beijing, China (13), and increases of 16 ppb at night and 13 ppb during the day in Seoul, South Korea (12).

Replacing urban areas with agriculture areas or parks improves urban air quality through two mechanisms: 1) a decrease in surface temperature due to increased evapotranspiration limits the production of secondary pollutants, and 2) an increase in the capacity for dry deposition. In agriculture, we observe an increase in dry deposition velocity (62 % and 17 %, for NO_2 and O_3, respectively) and a reduction in O_3 (up to 5%) over new agricultural areas. Over new urban parks, we observe an increase in dry deposition velocities (60% and 18% for NO_2 and O_3, respectively) during the day. Additionally, a reduction in primary pollutants such as NO_2 is observed in the morning (3%) because of increased deposition. In the evening, the reduction in surface temperature slightly decreases O_3 production and, consequently, O_3 concentration (2%).

Agriculture enhances ammonia and particulate pollution

While regulatory mechanisms target air pollution from the industrial and traffic sectors, less regulated ammonia (NH_3) emissions from agriculture are significant drivers of fine particulate matter ($PM_{2.5}$) pollution, warranting further attention. Agriculture contributes more than 81% of all global NH_3 emissions(85). Additionally, NH_3 volatilization affects the production of acid rain and acid particulate matter(86) and is indirectly related to N_2O and global warming(87); N_2O is a greenhouse gas (GHG) that depletes the O_3 layer in the atmosphere(88; 89). Our findings indicate that fertilizer use in agricultural practices deteriorates local air quality in terms of NH_3, $PM_{2.5}$, and PM_{10}, with concentrations increasing by up to 90%, 10 and 12%, respectively, over new agricultural areas. The potential increase in these pollutants and the resulting impacts on human health have raised concerns(90; 91).

While expanding urban agriculture reduces the need for agriculture elsewhere and decreases crop imports, the potential increase in pollutants near highly populated areas raises concerns. However, NH_3 emissions from peri-urban agriculture can be significantly reduced through the implementation of nutrient-circularity strategies in cities that substitute for mineral fertilizers. For example, negligible NH_3 emissions were observed in the open-field cultivation of spinach using struvite, a P and N-rich compound resulting from wastewater treatment plants(92). Another study led by Ti et al., (2018)(93) provide global meta-analysis and integrated assessment of measures to reduce NH_3 emission from agricultural systems. This study suggest that selecting the use of ammonium nitrate, controlled-release fertilizers, and deep fertilizer placement can reduce NH_3 emissions by 88.3%, 56.8%, and 48.0%, respectively. Another study show that injected application, with ammonium nitrate or organic fertilizers, reduced NH_3 loss in maize by 75% and 96%, respectively, and in winter wheat by 87% and 98%, compared to surface broadcast(94). Moreover, the use of organic amendments in surface soils, combined with deep-rooting plants in crop rotations and reduced tillage, significantly enhances soil organic matter, improving biochemical processes and extending crop rooting patterns(95). These strategies align with initiatives such as the Revised Waste Framework Directive(96), the European Green Deal(97), a New Circular Economy Plan(98), and the Common Agricultural Policy 2023-2027(99), as there are many benefits associated with avoiding imported crops through nutrient recovery, such as less energy use for transportation and waste management. Indeed, promoting peri-urban and urban agriculture is essential for the development of resilient urban food systems as described by the Farm to Fork initiatives(100); however, they need to be implemented in a way that does not contribute to additional NH_3 in the urban atmosphere.

Biogenic emissions increase the production of secondary pollutants

We find that O_3 concentrations increase throughout the day (up to 4%) when more urban parks are included in the AMB, which is caused by additional BVOC sources from newly emerging vegetation. In addition, the PM concentrations also increase from these additional BVOC sources. These results are in line with other studies. For example, Biagi et al., (2015)(101) found that emissions from green areas can locally increase O_3 concentrations by up to 7.4% during the daytime in the city of Bologna. Similarly, Coen et al., (2014)(102) reported that ozone levels in the city of Tel Aviv were 10 ppb higher in urban parks than on nearby streets during the summer.

Furthermore, abiotic stressors such as heat, drought and air pollution could increase BVOC emissions(103; 104; 105), increasing the risk of elevated ozone and aerosol formation. Compared with vegetation in rural regions, urban vegetation is subject to greater temperature stress under the UHI effect, and since BVOC emissions are highly temperature dependent(106), a notable increase in BVOC emissions is expected. It has also been shown that long-term exposure to high O_3 pollution in urban areas impedes the growth and survival of trees(107), leading to the release of BVOCs(108) and increasing the production of O_3. These results and future predictions of higher temperatures and droughts(3) highlight the importance of carefully selecting vegetation for urban parks with relatively low biogenic emission rates(28; 109; 110) to avoid increasing secondary pollutants that are harmful to human health such as O_3 and SOA. Therefore, in urban areas where temperatures often exceed 20°C, species that are high emitters of biogenic volatile organic compounds (bVOCs) should be avoided, especially in large-scale planting projects. Prashant et al., (2019)(111) suggested to avoid species like oaks, poplars, willows, and spruces and identify the woody plant species such as scots pine or stone pine among other tree species, as potentially advantageous for air pollution abatement. Another study led by Owen et al., (2002)(112) suggest that low emitting species in the Mediterranean area include Arbutus unedo, Pinus halepensis, Cistus incanus, Cistus salvifolius, Rosmarinus officinalis and Thymus vulgaris.

Considering that urban areas are making efforts towards reducing NO_x emissions, ozone chemistry, which depends on its precursors (NO_x and VOC levels), will become more sensitive to BVOC emissions(5). Indeed, the increase in urban green spaces (containing higher levels of BVOCs) can shift from a VOC-limited regime to a transitional area between VOC-limited and NOx-limited conditions, with favourable conditions for photochemical ozone formation.

Furthermore, high O_3 pollution often reduces plant biomass and stomatal conductance, which limits subsequent O_3 deposition and can lead to increased atmospheric O_3 concentrations(113; 114). Under urban conditions, a crucial question raised by all of this is whether direct O_3 uptake by urban trees surpasses their indirect generation of O_3 through the release of BVOCs.

Irrigation decreases surface temperatures and secondary pollutants

The simulations show that irrigation decreases surface temperature, especially in the case of increased agriculture, where an increase in surface area dedicated to the cultivation of local crops results in a 0.8 ºC decrease in temperature. In addition, irrigated areas have shown a greater capacity to deposit air pollutants (20-40%) and decrease biogenic emissions that are precursors of secondary pollutants (up to 4%). Consequently, we found a decrease in the concentrations of secondary pollutants such as O_3 (7%). Therefore, one important finding of this study is that irrigation considerably enhances the capacity of vegetation to reduce air pollution. Notably, some studies indicate that pollutant deposition can damage crop yields(115; 116). Analysing the impact of pollutant deposition on crop yields is beyond the scope of this paper, and further investigations are needed. We also found that surface concentrations of primary pollutants (8% for NO_2 and 9% for VOCs) increased within the irrigated areas due to weakened turbulent mixing, which is consistent with other studies(117; 32). Then, combined reduction in anthropogenic emissions from primary pollutants such as NH_3 and NO_x emissions and irrigation is an effective way to improve air quality in urban areas. However, the Mediterranean area is currently experiencing drought episodes. The risk of drought episodes in the future is expected to increase due to climate change(118). Water demands associated with land-use practices such as irrigation, directly affect the supply of freshwater through water withdrawals. Agriculture alone accounts for 85% of the world's consumption of water(119). Therefore, an efficient water management system is needed to cope with future droughts and the increasing demand for new green infrastructure and urban and peri-urban agricultural practices that improve air quality and, in the case of agriculture, increase the food supply locally. Farmers can optimize water use, boost crop yields, and ensure long-term agricultural sustainability by implementing water-saving irrigation methods like drip irrigation, water capture and storage, crop rotation, and organic farming. In particular, Parada et al., (2021)(120) show that organic substrates increased the crop's resilience to water restriction. Futhermore, Sosa et al. (2023)(121) demonstrate that the regulated deficit irrigation regime (conditions of low or moderate stress are maintained during several phenological stages) enhanced the agronomic benefits of compost and nutrient supply, thereby supporting a water-saving strategy.

Limitations of this study and future research

This study has limitations that highlight the need for future research. We consider the most representative Mediterranean vegetation types, namely deciduous broadleaf forest, evergreen broadleaf forest, deciduous needleleaf forest, grassland, cropland, herb, shrub, and brush plant functional types, but more specific urban vegetation and vegetation emission factors under the stress of urban environments are needed. A recent study Maison et al., (2024)(110) evaluated the impact of using a very detailed urban bottom-up tree biogenic emission inventory instead of the MEGAN biogenic emission model, which uses a plant functional type approach on air quality in Paris during a HW event. This study concludes that using a more detailed emission inventory improves air quality performance. However, there are still biases that could be improved if trees and other urban vegetation inventories are set up systematically in more cities and their suburbs. Second, the simulations of this study are at 1km resolution to cover the entire AMB (636 km^2). Future studies should combine regional and microscale modelling to properly assess the impact of urban land-use changes on air quality at a much finer scale. Besides, to further analise the best combinations of land use types that results in the best performance in terms of local air quality, we need to develop innovative strategies. In this context, artificial intelligence, particularly machine learning techniques, shows great potential to tackle this challenge as it is pointed in the recent study Vidal et al., 2024(122).

High pollution, typical of urban and suburban areas, can injure plants, altering chemical processes such as dry deposition and biogenic emissions. Knowledge of these pathways is limited and therefore not yet included in widely used models(123). The current understanding of dry deposition is largely based on only a few measurement datasets, many of which cover only short periods. Furthermore, the micrometeorological techniques used to assess the deposition rates of pollutants require large uniform vegetation areas to quantify the vertical flux to the surface, which is not the case for metropolitan areas with small vegetation patches(124). In summary, future urban air quality studies addressing these gaps or uncertainties should include more field studies over cities to refine the input model data (e.g. emissions inventories), improve model parameterizations (e.g. dry deposition and biogenic emission processes) and, subsequently validate any model results.

Our model results, which are designed to assess peri-urban agriculture and urban vegetation strategies, have broad applicability and can be effectively adapted to other cities with Mediterranean climates. By considering the complex interactions between the biosphere (the surface land, including vegetation and soils) and the atmosphere (specifically urban climate and air quality), this approach offers a comprehensive framework for evaluating the potential impacts of various land-use strategies. This approach presents a promising pathway for guiding land-use decisions in the context of urban climate adaptation and mitigation.

METHODS

Scenarios

The AMB's office of urban planning has recently approved the Urban Master Plan (PDU), which determines how the land uses of the AMB will change during the next 50 years, presenting several potential scenarios with various degrees of urbanization, peri-urban agriculture, and greening:

- In scenario URB, urban parks and built-up urban areas increase by 81% (18.56 km^2), and 9% (16.54 km^2), respectively, whereas agricultural land, forests and natural nonforested areas decrease by 21% (10.93 km^2), 9% (12.31 km^2) and 15% (11.82 km^2) respectively.

Table 1. Land-use percentage and irrigation amount ($hm^3 month^{-1}$) for each scenario for June-July.

Scenario	Description	Irrigation	Urban (%)	Agriculture (%)	Vegetation (%)
REF	Reference	5.69	44.89	8.35	46.76
URB	Urban	5.22	48.59	6.57	44.85
AGR	Agriculture	8.35	44.26	19.70	35.99
PARK	Urban parks	5.68	41.97	7.74	50.29

- In the AGR scenario, peri-urban agriculture increases by 136% (64.25 km^2) within the AMB, whereas natural nonforest areas decrease by 34% (26.20 km^2) and forests decrease by 20% (12.32 km^2).
- In the PARK scenario, urban parks and other urban green areas increased by 112% (25.34 km^2), whereas urban areas decreased by 6% (15.02 km^2) and forests increased by 5% (7.88 km^2).

The reference scenario (REF) against which the potential scenarios are compared is based on the land uses present in 2015, which is the same reference year used by the PDU. The scenarios are summarized in Table 1, and further details on the land use changes in each scenario are provided at resolutions of 1 km by 1 km in the Supplementary Figs. S14 and S15. In addition, a recent publication by Segura-Barrero et al. (2024)(47) outlines the effects of these scenarios on the food–water–climate nexus, specifically focusing on the local food supply, freshwater utilization, local summer temperatures, and efforts to mitigate global climate change.

Air quality model

We use the regional chemistry transport model Weather and Research Forecasting Model with Chemistry (WRF-Chem) version 4.3.3, a highly flexible community model for atmospheric research where aerosol–radiation–cloud feedback processes are considered(48). The WRF-Chem model is widely used for simulations of air pollution episodes(49; 50), including those in the AMB(51; 5). The model is coupled with a multilayer layer urban canopy scheme, the Building Effect Parameterization (BEP) with the Building Energy Model (BEP-BEM(52)), to represent the urban areas in our domain, which takes into account the energy consumption of buildings and anthropogenic heat, which has been previously validated for the area under study (53; 54). The Local Climate Zones (LCZ) classification(55) is used for the AMB, which is associated with a specific value for each LCZ for the thermal, radiative and geometric parameters of the buildings and ground, which are used by the BEP-BEM urban canopy scheme to compute the heat and momentum fluxes in the urban areas. Eleven LCZs with specific urban morphologies for each were added to the land-cover map. The land use, urban fraction, and dominant natural vegetation maps were calculated for each scenario (more information can be found in the Supplementary Text S1).

The WRF-Chem gas-phase chemical scheme chosen is the Regional Acid Deposition Model (RADM2)(56), as has been previously used for air quality studies on the AMB(51; 5) and Europe(57; 58; 51). RADM2 is coupled to the MADE/SORGAM aerosol module (59; 60) and the Wesely scheme(61) is used to calculate the gaseous dry deposition velocities. To adapt the model to urban areas, the dominant natural vegetation types and the urban fraction are taken into account to calculate the dry deposition velocity in each grid for urban areas. The meteorological and chemical initial and boundary conditions (IC/BCs) were determined via the ERA5 global model data(62) and WACCM(63), respectively.

We run the model with an irrigation scheme that captures current practices to represent the impact of agricultural and urban park irrigation on local meteorology and air quality. The Noah land surface model irrigation scheme (64) is used with an irrigation option that affects only affecting the soil surface, implying direct evaporation of water from the soil surface. The irrigation takes place from 4 to 5 UTC (6 to 7 local time) at a maximum rate of 6.00 $Lm^{-2}h^{-1}$. The irrigation requirements were derived for each scenario via a water balance model in the case of irrigated agricultural areas determined by Segura et al., (2024)(47) and from official reports on urban park irrigation estimates for urban park (65) (see Figure S16 and Text S2 at the Supplementary).

A one month simulation covering the Iberian Peninsula with a 9km x 9km horizontal resolution (domain 1, D1) was run to provide the initial and boundary conditions (ICs/BCs) for the second domain that covers the Catalonia region (D2, 3km x 3km horizontal resolution). The model is run with the nested domains option for D2 as a parent domain and domain 3, which covers the metropolitan area of Barcelona (D3, 1 km × 1 km horizontal resolution), as the nested domain. Table S1 in the Supplementary describes the main configuration of the model.

The simulation of the reference scenario was evaluated with meteorological observation data from the Catalan Meteorological Service (SMC) and the Spanish Meteorological Agency (AEMET) and air quality observation data publicly available from the Xarxa de Vigilància i Previsió de la Contaminació

Atmosfèrica (XVPCA) of the AMB (see Supplementary Text S3).

Emissions

Biogenic emissions, which depend on meteorological variables and vegetation cover, are computed online with the Model of Emissions of Gases and Aerosols from Nature v2 (MEGAN) (66). The biogenic emission factors for each plant functional type (PFT) have been modified via data from field campaigns over the Mediterranean region conducted by the CREAF(67; 68; 69; 70). In addition, the PFT "broadleaf tree" has been split into two new PFTs: "deciduous broadleaf tree" and "evergreen broadleaf tree" to better represent the vegetation type in the AMB and its surroundings. The leaf area index (LAI) and the Fraction of green Vegetation Cover (FCOVER) maps needed for the biogenic emissions model were taken from the Copernicus Global Land Service(71) with a 300m resolution. These products are generated on a 10-day basis and are averaged to obtain the monthly LAIv (LAI/FCOVER) values for each pixel of each domain. Since the scenarios have diverse land cover maps with varying degrees of agriculture and vegetation, each scenario has different LAIv values and consequently different biogenic emission inventories.

The HERMESv3 global and regional (HERMESv3_GR) preprocessing tool(72) was used to create the anthropogenic emissions inventory for domain D1 from the CAMS-REG-APv3.1 database (73) in year 2016. For the other two innermost domains, D2 and D3, anthropogenic emission inventories from 2015 were provided by the Barcelona Supercomputing Center via the HERMESv3 bottom–up emission model (HERMESv3_BU)(74), which computes bottom–up emissions from point sources, road transport, residential and commercial combustion activities, the use of solvents, shipping in ports, aviation, recreational boats, livestock, and agricultural activities. In terms of agricultural activities, HERMESv3_BU includes crop operations, machinery and fertilizers. Since the scenarios affect only land cover in terms of vegetation changes, all city-related activities are assumed to remain constant; consequently, the anthropogenic emission inventories are the same for all scenarios except for agriculture-related emissions and emissions linked to the domestic and industrial use of solvents, for which the spatial distribution is linked to urban land uses. For the URB scenario, we assume that an increase in the urban fraction means more buildings but that there is no significant impact on traffic activity or change in roads, so transport emissions remain the same as those in the REF scenario. Additional residential combustion emissions due to an increase in buildings do not affect the emission inventory because the study period is during the summer, with no need for heating, and new buildings typically use electricity rather than primary fuels for cooking and cooling. The anthropogenic emission inventories of the PARK scenario consider only variations in ammonia emissions resulting from agricultural activities, based on changes in agricultural areas between each scenario and the REF scenario.

DATA AVAILABILITY

The CAMS emission data are available from Emissions of atmospheric Compounds and Compilation of Ancillary Data (ECCAD) at https://eccad.aeris-data.fr/. The ERA5 global model data are available from the Copernicus Climate Data Store at https://doi.org/10.24381/cds.adbb2d47. The WACCM global data are hosted by the NCAR and are available upon request via the form at https://www2.acom.ucar.edu/wrf-chem/wrf-chem-tools-community (last access: 16 April 2025). The WRF-Chem simulation outputs, underlying Figs. 2–8, Supplementary Figs. S1–S12 and Tables S2-S3, are openly available via the Zenodo repository at https://doi.org/10.5281/zenodo.13838951 The WMS maps underlying Fig. 1 are available in GIS format via the Cartographic and Geological Institute of Catalonia at: https://www.icgc.cat/ca/Administracio-i-empresa/Serveis/Geoinformacio-en-linia-Geoserveis/WMS-i-tessel-lesCartografia-de-referencia/WMS-Mapes-i-ortofotos-vigents.

CODE AVAILABILITY

The WRF-Chem model code is available from https://www2.mmm.ucar.edu/wrf/users/download/get_source.html (last access: 16 April 2025), with the specific code used in this study available from the authors upon request (alba.badia@bsc.es). The $HERMESv3_GR$ model code is available from https://earth.bsc.es/gitlab/es/hermesv3_gr.

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Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

CRediT authorship contribution statement

AB carried out all the model simulations and data analyses, led the interpretation of the results, and prepared the manuscript with contributions from all the co-authors. GV contributed to the interpretation of the results and provided extensive comments on the manuscript. MG ran the HERMES_BU emission model as input data for the simulations. RS and SV prepare urban maps needed as input to run the model.

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