Achieving accurate simulations of urban impacts on ozone at high resolution

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Abstract

The effects of urbanization on ozone levels have been widely investigated over cities primarily located in temperate and/or humid regions. In this study, nested WRF-Chem simulations with a finest grid resolution of 1 km are conducted to investigate ozone concentrations [O₃] due to urbanization within cities in arid/semi-arid environments. First, a method based on a shape preserving Monotonic Cubic Interpolation (MCI) is developed and used to downscale anthropogenic emissions from the 4 km resolution 2005 National Emissions Inventory (NEI05) to the finest model resolution of 1 km. Using the rapidly expanding Phoenix metropolitan region as the area of focus, we demonstrate the proposed MCI method achieves ozone simulation results with appreciably improved correspondence to observations relative to the default interpolation method of the WRF-Chem system. Next, two additional sets of experiments are conducted, with the recommended MCI approach, to examine impacts of urbanization on ozone production: (1) the urban land cover is included (i.e., urbanization experiments) and, (2) the urban land cover is replaced with the region’s native shrubland. Impacts due to the presence of the built environment on [O₃] are highly heterogeneous across the metropolitan area. Increased near surface [O₃] due to urbanization of 10–20 ppb is predominantly a nighttime phenomenon while simulated impacts during daytime are negligible. Urbanization narrows the daily [O₃] range (by virtue of increasing nighttime minima), an impact largely due to the region’s urban heat island. Our results demonstrate the importance of the MCI method for accurate representation of the diurnal profile of ozone, and highlight its utility for high-resolution air quality simulations for urban areas.

Keywords: WRF-Chem, urbanization, ozone change, urban heat island, arid city, air quality, modeling

1. Introduction

Lower tropospheric ozone is mainly generated by chemical reactions of primary pollutants such as nitrogen oxides (NOₓ) and volatile organic compounds (VOCs) in the presence of sunlight (Duncan et al 2010). Concentrations of ground-level ozone continue to be a critical issue in major cities around the world (Banta et al 2005) since urban areas are often characterized by large emissions of NOₓ and VOCs (Wood et al 2009). These emissions are the primary pollutants, or precursors of ozone. The spatio-temporal distribution of ozone concentrations [O₃], including high-level ozone episodes, is the result of complex processes involving photochemical reactions, meteorology, and emissions of primary pollutants (Ryu et al 2013). In other words, any changes in...
chemical reaction processes (e.g., Henderson et al. 2011, Yegorova et al. 2011), meteorological/climate conditions (e.g., Mao and Talbot 2004, Zhang et al. 2007), and/or emissions (e.g., Chen et al. 2013, Huang et al. 2013) can affect [O$_3$].

It is well known that land cover and land use change (LCLUC) modifies local meteorological/climate conditions (e.g., Georgescu et al. 2011, 2014, Pielke et al. 2011, Boisier et al. 2012). Consequently, much work has also been performed examining LCLUC impact on air quality (e.g., Cheng et al. 2008, Tao et al. 2013). One of the major LCLUCs in the past decades is worldwide urbanization and, based on World Bank data, urban expansion is continuing (http://data.worldbank.org/indicator/SP.URB.TOTL.IN.ZS). Urban expansion does not only result in modified land surface/near-surface thermal features, but also increases local emissions of air pollutants due to the intensification of traffic, industry, and energy consumption resulting from population rise.

Therefore, considerable research has been performed to investigate effects of urbanization on air quality, especially ozone production. For example, Ryu et al. (2013) used Weather Research and Forecasting (WRF)/CMAQ in one-way coupling at 1 km resolution to assess impacts of urban land-surface forcing on [O$_3$] in the Seoul metropolitan area. They found that urbanization modifies both boundary layer properties and local circulation, and increases ozone-mixing ratio by 16 ppb at nighttime and 13 ppb during daytime hours. Tao et al. (2013) used a modified WRF-Chem (NASA Unified WRF model) at 20 km resolution and multiple US land cover datasets to investigate the effects of land use and land cover change on air quality. Using the same anthropogenic and biogenic emissions, their case studies indicate that [O$_3$] was reduced by 2 ppb when MODIS land cover data are used in comparison with data from USGS. Since MODIS data represent urban cover areas better than USGS data, they suggest the 2 ppb difference in ozone is due to increasing continental US urbanization.

Using a European coupled meteorology-chemistry model and Town Energy Balance model at 3 km resolution, Sarrat et al. (2006) concluded that the built environment in Paris caused changes in both daytime and nighttime spatial distribution of primary and secondary pollutants, mainly due to modified turbulence. Civerolo et al. (2007) reported from simulations using MM5/CMAQ at 4 km resolution that urbanization in New York could result in ozone increases of 1–5 ppb for episode-averages as well as up to 6 ppb for 8 h episode-maxima for large parts of the city. They also found that the spatial pattern of ozone change is heterogeneous and can exhibit decreases for some portions of the urban area. Wang et al. (2007) used WRF-Chem at 4 km resolution to study the effects of urbanization for the Pearl River Delta region in South China on air quality. They concluded that urbanization could decrease wind speed and increase [O$_3$] by 10 ppb during daytime and 14 ppb during nighttime. All of the above studies took place in temperate and/or humid climate regimes.

The effect of urbanization on [O$_3$] over arid/semi-arid regions has received little attention despite greater global population growth rates for dryland areas relative to any other ecological zone (www.un.org/en/events/desertification_decade/whynow.shtml). These regions include major cities of global importance: Lima (Peru), Cairo (Egypt), Marrakesh (Morocco), and Dubai (United Arab Emirates), to name just a few. Investigation of urbanization impacts within these populated regions is practically meaningful as urban areas, globally, aim to enhance living comfort levels within their environments and scientifically essential as the community continues development of accurate simulation tools to address concerns raised by policymakers, stakeholders, and the populace at large.

There are about 56 million people living in the arid/semi-arid Southwestern United States and this number is projected to increase to 94 million by 2050 (Garfin et al. 2013). According to a survey conducted by the Environmental Protection Agency (EPA) from 2005 to 2008, all or parts of 48 counties in the Southwest did not attain the 8 h ozone National Ambient Air Quality Standard (NAAQS) (Garfin et al. 2013), despite emission controls that have been in place for years (Pusede and Cogen 2012). Thirty-six (of 58) counties in California, eight (of 64) in Colorado, two (of 16) counties in Arizona, and two (of 29) counties in Utah were deemed as non-attainment (Garfin et al. 2013). The majority of these counties are located in Colorado, Utah, Arizona, and California. High near surface [O$_3$] adversely affects human health, agricultural productivity, and other components of the ecosystem. Additionally, lower tropospheric ozone is one of the six EPA-regulated pollutants in the United States. Using reliable numerical modeling of ozone to facilitate attainment status can improve our understanding of dispersion processes and spatio-temporal distribution of ozone. Importantly, such an approach can shed light on which emission control strategies would lead to attainment.

Most prior modeling studies focusing on ozone and usage of WRF-Chem were based on domains with grid-spacing ranging from a few kilometers (e.g., Wang et al. 2007, Chen et al. 2013, Ritter et al. 2013) to tens of kilometers (Yegorova et al. 2011, Tao et al. 2013). Few studies using WRF-Chem at higher resolution have been reported (e.g., Joe et al. 2014, which was focused on particulate matter). Tie et al. (2010) have suggested that higher model resolution can produce more reliable results for air quality studies, due to better representation of topography, land use features, and emissions. Although recent simulations have demonstrated the importance of built environments within arid/semi-arid regions on local weather and climate (Grossman-Clarke et al. 2010, Georgescu et al. 2011, 2012, Salamanca et al. 2013), there has been comparatively less work focusing on air-quality impacts. Here we examine the variations of [O$_3$] and its precursors that are caused by urbanization, at 1 km resolution using WRF-Chem, within the rapidly expanding semi-arid Phoenix metropolitan area (Georgescu et al. 2013). This is a topic that has received limited research attention to date (Ellis et al. 2000, Lee et al. 2003), but requires consideration due to the metropolitan area’s frequent non-attainment ozone status. Section 2 presents the methodology, the data utilized and the
model setup. Results and discussion are presented in section 3, followed by concluding remarks in section 4.

2. Methods

The complex terrain characterizing the Phoenix area—with a population approaching 4.5 million people located within more than two dozen distinct cities—combined with its arid and hot summer climate contributes to the production of elevated [O₃]. Mountain-valley circulations are regarded as key factors affecting advection and diffusion of air pollutants (Fast et al 2000, Ellis et al 2000, Kleinnan et al 2003, Lee et al 2003, 2007, Atkinson-Palombo et al 2006, Lee and Fernando 2013). Meteorological models such as the Regional Atmospheric Model System (Pielke et al 1992) and PENN State/NCAR Mesoscale Meteorological Model 5 (Grell et al 1994), which are coupled in one-way mode with a chemical model, have been used in previous studies to simulate [O₃] in Phoenix (e.g., Fast et al 2000, Lee et al 2007, Lee and Fernando 2013). The WRF model is coupled with a Chemistry model (Chem) in two-way mode (Grell et al 2005) and has been widely used to simulate air-quality and atmospheric physics—chemistry interactions (e.g., Zhang and Dubey 2009, Tie et al 2010, 2009, Yegorova et al 2011, Zhao et al 2013, Chen et al 2013, Tao et al 2013). The focus of this research is on the prospective utility of high-resolution WRF-Chem simulations, which have not been used for this area to examine air quality impacts.

2.1. WRF-Chem setup

We use version 3.5.1 of WRF-Chem as the integration model. The WRF model (Skamarock et al 2008) is employed to resolve atmospheric physics and dynamical processes, while the chemistry (Chem) model is used to simulate chemical processes such as gaseous and aqueous chemical reactions, dispersion, and deposition. The WRF-Chem setup includes Lin’s double-moment cloud microphysics parameterization (Lin et al 1983), the RRTM radiation scheme (Mlawer et al 1997), the Noah land surface model with single layer urban canopy model (Chen and Dudhia 2001, Ek et al 2003, Chen et al 2011), the Grell–Devenyi ensemble cumulus scheme (Grell and Devenyi 2002) that allows subsidence and spreading at high-resolution, revised MM5 surface layer, and the BouLac planetary boundary layer (PBL) schemes. Land cover and land use data from MODIS 1 km resolution dataset (Friedl et al 2002) are combined with the 2006 National Land Cover Database (NLCD) 3-class urban covers to better represent the urban landscape. The second generation regional acid deposition model (RADM2, Stockwell et al 1990, Gross and Stockwell 2003) is used for gas-phase chemical reactions. The aerosol scheme is based on the MADE/SORGAM with GOCART, functioning as a background aerosol scheme that accounts for surface wind speed, soil moisture, and soil erodibility (Ginoux et al 2001). The other selected chemistry schemes are based on the recommendations provided in the WRF-Chem users’ guide (Peckam et al 2013).

Four nested domains are used (figure S1 in supplementary materials, available at stacks.iop.org/ERL/9/114019/mmedia). The first (domain 1) has 36 km grid spacing and covers the Western and Central US, Eastern Pacific, Northern and Central Mexico, the Gulf of California, and the Western Gulf of Mexico. Nested domains 2, 3, and 4 use a grid spacing of 12 km, 4 km, and 1 km, respectively. The innermost domain (1 km grid spacing) encompasses the urbanized portion of Phoenix and surrounding desert areas, including mountains, to better represent the complex terrain and land cover features (figure 1). The observation sites (including ozone and meteorological observations) used for validation and urban coverage are also superimposed in figure 1.

2.2. Downscaling of anthropogenic emissions

The anthropogenic emissions used in this study are obtained from NEI05 data provided by the US EPA (www.epa.gov/ttnchie1/net/2005inventory.html). These data are distributed on a 4 km grid covering the US and surrounding land areas. A method that can be applied to interpolate the 4 km grid spacing NEI05 data to any resolution one wishes to use for WRF-Chem simulations is provided with the WRF-Chem system (www.acd.ucar.edu/wrf-chem/). Each WRF-Chem model grid point data is based on averaging from those NEI05 grid points that fall within a distance less than the WRF-Chem model resolution (here, we refer to this method as the ‘default method’). The method works well when WRF-Chem grid spacing is coarser than 4 km. The method misrepresents emissions, however, when the model resolution is greater than the NEI05 grid. To overcome this issue, we use a Monotonic Cubic Interpolation (hereafter ‘MCI’) to downscale the 4 km resolution NEI05 data to a 1 km resolution grid (the finest model grid spacing of our WRF-Chem simulations). The
Merit of MCI is that it preserves the shape of emissions and does not introduce negative mass values compared with non-MCI methods. Details on emissions and model output evaluation can be found in the following sections and supplementary section 2, and figures S2–S5.

2.3. Data used for model initialization and evaluation

The data used for WRF-Chem initialization include biogenic emissions with 1 km resolution obtained from the Model of Emissions of Gases and Aerosols from Nature (MEGAN, Guenther et al 2006). We use the North American Regional Reanalysis (NARR; Mesinger et al 2006) product, a reanalysis dataset for regional meteorological fields, for initial and boundary conditions. These data are distributed on a 32 km grid with a 3 h temporal frequency. The atmospheric chemical boundary and initial conditions are from the global air quality forecast model called MOZART-4/GEOS-5 (www2.acd.ucar.edu/acresp/forecasts), available on a 1.9 × 2.5° grid (Emmons et al 2010).
The data used for model evaluation include measurements of hourly 2 m temperature (eight sites within urban locations) and 10 m wind speed and direction (seven sites within urban locations). These meteorological data are obtained from the Maricopa County Air Quality Department (MCAQD) and the Flood Control Department of Maricopa County (FCDMC). In addition, we use hourly observations of \([O_3]\) from 24 stations and the hourly NO\(_x\), CO from two stations, courtesy of EPA (www.epa.gov/ttn/airs/airsaqs/). In the Phoenix area, the VOC observations are irregularly available and have not been found for the selected episodes.

### 3. Results

Two episodes (14 May 2012 and 09 June 2011) are selected as case studies. The criterion for selection of these episodes required observed daily maximum 8 h \([O_3]\) to exceed 80 ppb for at least ten of the reporting stations. Event selection also relied on availability of CO and NO\(_x\) observations, necessary for detailed evaluation of model simulations against ozone precursors. The model is initialized three days prior to each episode (i.e., output from these days is used for spin-up and discarded).

#### 3.1. Model evaluation

In this section, we focus our discussion on \([O_3]\) simulated by the innermost WRF-Chem domain (i.e., domain 4 with 1 km resolution). Given that our focus is on air quality, the simulated meteorological conditions for the present case studies are only briefly summarized (see supplemental section 3 for details). The comparison between the observed wind fields averaged over eight sites (from MCAQD) and the model grid-points closest to the locations of these sites shows good agreement (figure S6 and table S2). As in previous studies (e.g., Lee and Fernando 2013), the simulations overestimated daytime wind speed. Overall, the model reproduces the late afternoon and nighttime urban heat island (UHI) (figure S7 and table S3).

The comparison between simulated and observed CO and NO\(_x\) for available reporting stations is presented in figure 2. In comparison with ozone observations, the WRF-Chem Urban experiment (MCI-labeled in figure 2) better captured correspondence to observations relative to the Default method. Additional site-by-site evaluation can be found in the supplementary materials (figures S3–S5).

The model generally overestimates observed magnitudes of CO although simulations reproduced the diurnal variability. NO\(_x\) diurnal comparison between model output and available observations are also shown in figure 2. As with CO, WRF-Chem reproduced the observed diurnal variations, although time shifts between the modeled NO\(_x\) and observed NO\(_x\) peak are apparent. Previous studies have detected a similar temporal shift (e.g., Lee and Fernando 2013). NO\(_x\) concentrations show two peaks daily (early morning and late afternoon) while \([O_3]\) show one peak in the early afternoon. NO\(_x\) diurnal variations are out of phase with \([O_3]\) and in phase

#### Table 1. Statistical results of hourly ozone concentrations of WRF-Chem simulations when observation is greater than 40 ppb with anthropogenic emissions being down scaled using MCI method.

|                | 09 June 2011 | 14 May 2012 |
|----------------|-------------|-------------|
| MB (ppb)       | −1.69       | −1.50       |
| RMSE (ppb)     | 14.70       | 14.75       |
| NMB (%)        | −6.32       | −6.50       |
| NME (%)        | 15.32       | 14.43       |
| MB (%)         | −5.59       | −5.60       |
| MNGE (%)       | 15.70       | 15.76       |
| IA             | 0.84        | 0.81        |
| R              | 0.75        | 0.74        |
| Sample #s      | 670         | 667         |

MB: mean bias  
RMSE: root mean square error  
NMB: normalized mean bias  
NME: normalized mean error  
MB: mean normalized bias  
MNGE: mean normalized gross error  
IA: index of agreement  
R: correlation coefficient

The data used for model evaluation include measurements of hourly 2 m temperature (eight sites within urban locations) and 10 m wind speed and direction (seven sites within urban locations). These meteorological data are obtained from the Maricopa County Air Quality Department (MCAQD) and the Flood Control Department of Maricopa County (FCDMC). In addition, we use hourly observations of \([O_3]\) from 24 stations and the hourly NO\(_x\), CO from two stations, courtesy of EPA (www.epa.gov/ttn/airs/airsaqs/). In the Phoenix area, the VOC observations are irregularly available and have not been found for the selected episodes.
with CO diurnal variations, relevant to emissions and photochemical reactions.

Figure 3 presents simulated and observed [O₃] for the pair of cases. The average of the 24 available observation sites and the corresponding model grid cell closest to each observation site is plotted. Error bars represent standard deviations of the 24 sites at each time period. Figure 3 illustrates that model simulations reproduced observed [O₃] with excellent fidelity. Individual station-by-station evaluation is shown in the supplementary section (see line labeled as MCI in figures S3–S4, and note improvement relative to default method). The site-by-site plots also display simulated results comparable with observations in urban as well as rural areas.

Model [O₃] performance is also evaluated against a range of EPA recommended statistical metrics (EPA 1991) and this is presented in table 1. EPA selects the mean normalized bias (MNB) and the mean normalized gross error (MNGE) as metrics when observed values of ozone mixing ratio exceed 40 ppb. These two metrics must have values that fall within ±15% (in magnitude) and ±35% for MNB and MNGE, respectively, based on the USEPA acceptance criteria for model performance. Statistical results indicate that the MNB and MNGE, for the two cases, fall in the acceptance ranges.

The evaluation shown in figures 2–3 (and figures S3–S5) and the statistical analysis presented in table 1 (also table S1) demonstrate that the newly developed MCI method to downscale anthropogenic emissions achieve superior results compared to those obtained using the default interpolation method.

3.2. Impact of urbanization on ozone concentration

Next, we use simulations initialized by the MCI method to investigate impacts of urbanization on [O₃] within the Phoenix metropolitan area. To achieve this goal, we have conducted additional WRF-Chem simulations for the selected cases with the same WRF-Chem setup and MCI-based
Figure 5. Scatter plots depicting ozone concentration differences between the Urban and NO\textsubscript{urban} simulations at daytime for the periods (a) 14 May 2012 and (b) 09 June 2011. (c) and (d) are the same as (a) and (b), but at nighttime.

Figure 6. Simulated comparison of potential temperature profiles (panels (a) and (b)) and Turbulence Kinetic Energy (TKE) profiles (panels (c) and (d)) for Urban run and NO\textsubscript{urban} run: (a) 14 May 2012 episode and, (b) 09 June 2011 episode; (c) and (d) are as (a) and (b), but for TKE. The heights (m) represent heights above ground level (agl). The data in the figure are averaged over the urban areas (indicated in figure 1) during nighttime (the same time periods as figures 5(c) and (d)).
emissions used in the simulations presented and evaluated in section 3.1 (these experiments are referred to as ‘Urban’), except that all urban pixels are replaced by open shrubland (the native vegetation of the semi-arid Sonoran desert). These simulations will be referred to as ‘NO_urban’ experiments.

Figure 4 shows the time series of NO$_x$ (figures 4(a) and (b)) and ozone (figures 4(c) and (d)) concentrations, averaged over the urban areas illustrated in figure 1, for each of the selected episodes. Overall, ozone and NO$_x$ concentrations are out of phase. Urbanization displays diurnally varying impacts on pollutant distribution. During the daytime (15Z–2Z or 08–19MST), differences in NO$_x$ concentrations between simulations with and without urban land cover are relatively small. However, a systematic increase of NO$_x$ is apparent during nighttime (03Z–14Z or 20MST–07MST) and early morning when the urban land cover is not accounted for. NO$_x$ concentrations for the NO_urban simulations are nearly doubled relative to simulations accounting for the urban landscape.

The middle panels of figure 4 (figures 4(c) and (d)) illustrate the diurnal cycle of [O$_3$] for the urban grid cells shown in figure 1. It is clear that [O$_3$] from late afternoon onward are higher when urban land cover is considered, relative to simulations without urban land cover. Maximum differences as high as 15 ppb (see also figure 4(f)) occur during nighttime hours. During daytime, averaged [O$_3$] changes range between −2.0 ppb (for the 09 Jun 2011 case) to +1 ppb (for the case of 14 May 2012).

Figures 4(e) and (f) show time differences between simulations with and without urban land cover for NO$_x$ and O$_3$ concentrations. Urbanization results in a decrease of [O$_3$] of ~1 ppb during daytime and an increase of [O$_3$] by ~5–15 ppb from late afternoon to early evening, coinciding with the region’s developing UHI. Diurnally averaged, we calculate that urbanization will result in [O$_3$] change of 4 ppb daily, less than magnitudes reported previously for temperate and/or humid areas (e.g., Wang et al 2007, Ryu et al 2013).

Our results indicate that urbanization results in substantially increased ozone concentrations and decreased NO$_x$, during nighttime (consistent with previous studies in temperate and/or humid regions). Urbanization results in little change during daytime hours (indicating a potentially unique feature of semi-arid built environments).

Figure 5 shows scatter plots depicting [O$_3$] differences between the Urban and NO$_x$_urban simulations for the urban pixels represented in the control simulations (i.e., the Urban experiments). The differences in the hourly ozone concentration are highly variable with values ranging between −30 and 50 ppb. Our results are consistent with previous studies that also showed inhomogeneity in distribution of ozone concentrations (Sarrat et al 2006, Civerolo et al 2007, Wang et al 2007). During the daytime, differences in [O$_3$] between Urban and NO$_x$_urban cases are small (figures 5(a) and (b)) with an average value close to zero, further indicating a small impact of urbanization. At nighttime (02Z–15Z), these differences are larger with, on average, enhanced [O$_3$] within urban areas (figures 5(c) and (d)).

The principal mechanism responsible for urban-induced [O$_3$] increase is largely due to the region’s UHI (figures S7 and S8(a)), wherein nighttime vertical stability is reduced and turbulence is enhanced (figure 6). Consequently, increased instability enhances turbulent transport for Urban simulations (figures 6(c) and (d)), increasing planetary boundary layer height (PBLH) during the nocturnal period. Destabilization of the lower atmosphere favors the vertical diffusion of air pollutants and results in relatively small ground-level NO$_x$.
concentrations (figures 4–5). This leads to enhanced vertical diffusion of pollutants (such as NOx, CO), reducing the titration rate at ground level, and resulting in increased ground-level [O3] (figure 4). During daytime, due to strong solar radiation and small thermal capacity of the desert area, the 2 m temperature and PBLH over desert areas are as high, or higher, than those in urban areas (figure S8). Small thermal differences between the urban and surrounding desert area result in a slightly negative UHI during daytime, consistent with previous studies (e.g., Georgescu et al 2011).

Finally, figure 7 illustrates the regional distribution of mean ozone changes due to urbanization during daytime and nighttime, for the two episodes. As mentioned previously, ozone changes are small during daytime hours (generally less than 2–4 ppb). During nighttime hours, the increase in ozone ranges between 10 and 20 ppb (more than 12% of the mean ozone concentrations). Differences in ozone fields between urban and nonurban simulations at nighttime are largely limited to urban areas, demonstrating that urbanization is a principal factor responsible for ozone enhancement at night.

4. Conclusion

Here we presented nested WRF-Chem simulations of ozone distribution with a finest grid spacing of 1 km (innermost domain) focused on the Phoenix metropolitan area as a demonstrative case study for urban regions in semi-arid environments. We proposed a method to downscale anthropogenic emissions from the 4 km resolution NEI05 data to the model finest grid resolution of 1 km based on a shape preserving MCI. WRF-Chem experiments using this method considerably improved simulated [O3] relative to results obtained from the default WRF-Chem method.

The impacts of urbanization on [O3] are investigated by conducting a suite of simulations, utilizing the downscaling anthropogenic emissions method, with and without urban land cover. During the daytime, [O3] changes are small but the presence of the built environment increases [O3] by 10–20 ppb during nighttime hours. Hence, urbanization narrows the daily [O3] ranges by raising nighttime minima. The presence of the built environment increases surface air temperatures and instability at night, leading to enhanced turbulence and a greater boundary layer height relative to simulations with the region’s native shrub landscape. This supports enhanced vertical diffusion of NOx and CO, thereby reducing their low-level concentrations. As a result, the decrease in ground-level NOx and corresponding decrease in titration rate, enhances low-level ozone concentrations during nighttime hours.

Understanding regional and temporal distribution of ozone is important for decision makers in deriving strategies to reduce emissions of its precursors and can help urban planners consider strategies to improve air quality associated with future urban expansion (Georgescu et al 2014). The improved technique we have presented to downscale anthropogenic emissions from their native 4 km resolution to finer scales is an important step toward achieving the superior model skill necessary to assist the urban planning community in devising air quality mitigation strategies in the future. Importantly, although we have presented results for the Phoenix metropolitan area, the techniques can be generalized to other locales.

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