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A modeling study of the impact of urban trees on ozone

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Abstract

Modeling the effects of increased urban tree cover on ozone concentrations (July 13–15, 1995) from Washington, DC, to central Massachusetts reveals that urban trees generally reduce ozone concentrations in cities, but tend to increase average ozone concentrations in the overall modeling domain. During the daytime, average ozone reductions in urban areas (1 ppb) were greater than the average ozone increase (0.26 ppb) for the model domain. Interactions of the effects of trees on meteorology, dry deposition, volatile organic compound (VOC) emissions, and anthropogenic emissions demonstrate that trees can cause changes in dry deposition and meteorology, particularly air temperatures, wind fields, and boundary layer heights, which, in turn, affect ozone concentrations. Changes in urban tree species composition had no detectable effect on ozone concentrations. Increasing urban tree cover from 20 to 40% led to an average decrease in hourly ozone concentrations in urban areas during daylight hours of 1 ppb (2.4%) with a peak decrease of 2.4 ppb (4.1%). However, nighttime (20:00–1:00 EST) ozone concentrations increased due to reduced wind speeds and loss of NO_x scavenging of ozone from increased deposition of NO_x. Overall, 8-hour average ozone concentration in urban areas dropped by 0.5 ppb (1%) throughout the day.

Keywords: Urban forestry; Air quality modeling; Photochemistry; Urban meteorology; Biogenic hydrocarbons; Dry deposition

1. Introduction

Vegetation in cities, particularly trees, can directly and indirectly affect urban air quality. Tree transpiration and tree canopies affect: (a) meteorology (air temperature, radiation absorption and heat storage, wind speed, relative humidity, turbulence, surface albedo, surface roughness and consequently the evolution of the mixing-layer height) (e.g., Heisler et al., 1995; Berman et al., 1997); (b) dry deposition of gases to the earth's surface (deposition velocity) (e.g., Baldocchi et al., 1987); (c) emission of volatile organic compounds (VOCs) that can contribute to the formation of ozone (O_3) and carbon monoxide (CO) (Brasseur and Chatfield, 1991); and (d) anthropogenic emissions through reduced energy use due to lower air temperature and shading of buildings. While lower pollutant emissions generally improve air quality, lower nitrogen oxide (NO_x) emissions, particularly ground-level emissions, may lead to a local increase in O_3 concentrations under certain conditions due to reduced NO_x scavenging of O_3 (Rao and Sistla, 1993; Rao and Mount, 1994). These four impacts of vegetation, which lead to changes in atmospheric chemistry and physics, all interact to affect ozone concentrations.

Research integrating the cumulative effects of urban vegetation on air quality, particularly ozone, is very limited. Cardelino and Chameides (1990) modeled vegetation effects on ozone concentrations in the Atlanta region using the OZIPM4 model. The study's primary focus was on the interaction of VOC emissions and altered air temperatures, and revealed that a 20% loss in the area's forest due to urbanization could have led to a 14% increase in O_3 concentrations for the modeled

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day. Although there were fewer trees to emit VOCs, an increase in Atlanta's air temperatures due to the urban heat island, which occurred concomitantly with tree loss, increased VOC emissions from the remaining trees and anthropogenic sources, and altered O_3 chemistry such that concentrations of O_3 increased.

A model simulation of California's South Coast Air Basin suggests that the air quality impacts of increased urban tree cover may be locally positive or negative with respect to ozone. The net basin-wide effect of increased urban vegetation is a decrease in ozone concentrations if the additional trees are low VOC emitters (Taha, 1996). This study used the Colorado State University Mesoscale Model (CSUMM) and Urban Airshed Model (UAM-IV), and accounted for vegetation effects on meteorology and emissions. The most significant meteorological impacts were temperature reductions, which altered chemical reaction rates, reduced temperature-dependent biogenic VOC emissions, and changed the depth of the mixed layer. The study also accounted for increased pollution deposition and possible increased VOC emissions due to increased vegetative cover.

McPherson et al. (1998) estimated the cost effectiveness of residential yards trees for improving air quality in Sacramento, CA, using Best Available Control Cost Technology cost analysis. The benefit–cost ratio over 30 yr for planting 500,000 trees was 0.5. However, the findings of this study are debatable as it omitted the quantification of the net effect of urban trees on air quality (Nowak et al., 1998a).

This paper details findings of a study that investigates the cumulative and interactive effects of increased urban tree cover on urban and regional ozone concentrations in the Northeastern United States. The study uses meteorological, emission, and air quality models and field data on urban vegetation to assess the impacts of altered meteorology, dry deposition, biogenic VOC emissions, and anthropogenic emissions, due to changes in urban tree cover, on ozone concentrations from July 13 (11:00 EST) to July 15 (14:00 EST), 1995. This time period was chosen due to the availability of model input data and used clean initial and boundary layer conditions to isolate the effects of urban tree cover on ozone concentrations. The purpose of these simulations were not to validate model performance but to compare model outputs against model inputs as related to urban tree cover.

2. Methodology

The study domain encompasses an area of 153 north–south by 111 east–west 4-km grids that contain many major urban areas of the eastern seaboard from Washington, DC, to central Massachusetts (Fig. 1). Of the total domain's 16,983 grid cells, 473 are designated as urban in the meteorological model (MM5 – version 1).

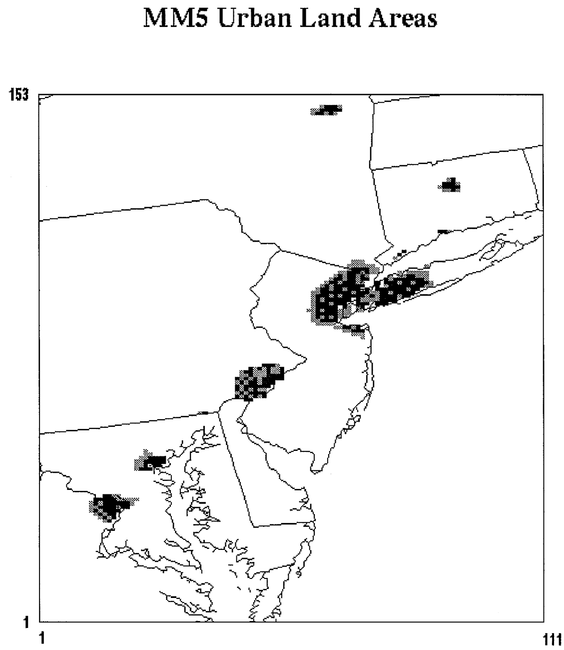

Fig. 1. Map of modeled urban area (dark and hatched cells) and total model domain in the northeastern United States (Dark = urban, hatched = urban changed to deciduous forest).

Within three urban areas (New York, NY; Philadelphia, PA; and Baltimore, MD), approximately 210 stratified (by land use) randomly located 0.04 ha field plots were measured per city to quantify urban tree characteristics, particularly leaf biomass composition by species (Nowak and Crane, in press). In addition, 1994 aerial photography was randomly sampled using a random dot-grid (New York: $n = 13{,}856$ grid points; Philadelphia: $n = 8{,}068$) and 1992 Landsat TM leaf-on and leaf-off data were analyzed for Baltimore (Grove, 1996) to determine tree and other surface cover characteristics of the city. These field and photo data were used to obtain better estimates of urban tree species leaf biomass distributions needed for quantification of biogenic VOC emissions.

2.1. Urban tree cover model scenarios

The intent of these hypothetical simulations is to assess changes in pollutant levels as a result of changes in urban tree cover. Five sets of model scenarios were analyzed in a stepwise fashion to assess the effect of changes in urban tree cover, anthropogenic and biogenic emissions, and species composition on ozone concentrations (Table 1): (1) Base case (BASE) – base urban conditions; (2) High tree (HITRE) – 40% urban tree cover; no change in utility emissions; (3) High tree – energy (HITRE-LOE) – 40% urban tree cover, a 5% reduction in utility point source emissions due to energy conserving effect of

Table 1
Meteorology, dry deposition, and anthropogenic and biogenic emission inputs for SAQM simulations

SAQM case[a]	Meteorology[b]	Deposition[c]	Anthro. emissions[d]	Biogenic emissions[e]
BASE	Base	Base	Base	Base
HITRE	High Tree	High Tree	Base	High Tree
HITRE-LOE	High Tree	High Tree	High Tree	High Tree
SPPΔ	High Tree	High Tree	High Tree	High Tree/Low Emitters
LANDΔ	High Tree	High Tree	Land Use Change	High Tree
VdVOCΔ	Base	Dep. Change	Base	Reduced Tree

Note: Changes to inputs only occurred within cells designated as urban.

[a]BASE – base urban conditions for MM5 and EMS-95; field data inputs and 20% tree cover in BEIS-2 and UFORE.
HITRE – increase from BASE to 40% urban tree cover.
HITRE-LOE – 40% urban tree cover and consequent anthropogenic emission effects.
SPPΔ – 40% urban tree cover and change in species composition to low emitters.
LANDΔ – 40% urban tree cover and removal of anthropogenic emissions in urban cells changed to forest.
VdVOCΔ – removal of urban trees with no change in meteorology or anthropogenic emissions.

[b]Meteorological variables from MM5.
Base – original land use distribution (Pielke, 1984), but with urban cells dominated by anthropogenc features these cells inherently contain some vegetation of an unknown amount.
High Tree – 40% of urban cells changed to deciduous forest.

[c]Dry deposition velocities from UFORE.
Base – 20% tree cover, 30% grass cover, 50% artificial surface cover.
High Tree – 40% tree cover, 10% grass cover, 50% artificial surface cover.
Dep. Change – 0% tree cover, 50% grass cover, 50% artificial surface cover.

[d]Anthropogenic emissions from EMS-95.
Base – emissions based on base temperatures from MM5.
High Tree – motor vehicle emissions based on high tree MM5 temperatures; 5% reduction in utility point source emissions.
Land Use Change – anthropogenic emissions removed in 40% of urban cells (cells changed to deciduous forest).

[e]Biogenic emissions from SMOKE-BEIS2.
Base – base MM5 temperatures; base emissions with leaf biomass altered in 83 urban cells based on field data.
High Tree – emissions in 40% of urban cells scaled up to 100% tree cover; high tree MM5 temperatures.
High Tree/Low Emitters – emissions in 40% of urban cells scaled up to 100% tree cover with all species converted to *Prunus* (low-emitting genera); high tree MM5 temperatures.
Reduced Tree – all trees in urban cells replaced with 'open forest' emissions (low hydrocarbon and NO emissions); base MM5 temperatures.

increased tree cover (e.g., McPherson, 1994), and motor vehicle emissions based on new air temperatures; (4) Species change (SPPΔ) – same as (3) but all trees changed to low VOC emitters (*Prunus* spp.); (5) Land use change (LANDΔ) – same as (3) plus removal of anthropogenic emissions in 40% of the urban cells (a reduction of about 10–15% of total anthropogenic emissions of CO, NO_x and VOC); and (6) deposition velocity and VOC change (VdVOCΔ) – trees removed from the urban areas to test the impact of changes in deposition velocity and biogenic VOCs on ozone concentrations without meteorological changes.

Urban areas in the model domain currently have approximately 20% tree cover (based on field and photo data). However, the 40% tree cover for the high tree scenarios is hypothetical to illustrate potential maximum effects from large-scale tree planting efforts. Five models were used to determine the cumulative effect of urban vegetation change on ozone concentrations: (a) Mesoscale Model (MM5) for meteorology; (b) Urban Forest Effects (UFORE) model for dry deposition; (c) Emissions Modeling System (EMS-95) for anthropogenic emissions; (d) Biogenic Emissions Inventory System (SMOKE-BEIS2) for biogenic emissions; and (e) SARMAP Air Quality Model (SAQM) for integrative air quality modeling. UFORE and SMOKE-BEIS2 models are specifically designed to quantify the impacts of vegetation; the other models are based on urban land use classes and incorporate some level of vegetation structure inherent within the land use classes, but the amount vegetation within these classes are unknown.

2.2. Meteorological modeling

Meteorological modeling was performed with the Penn State University/National Center for Atmospheric Research (NCAR) Mesoscale Model, commonly referred to as MM5 (Dudhia, 1993). It is a nonhydrostatic, prognostic model with a terrain-following vertical coordinate. The model incorporates a high resolution terrain and land use database, with 13 land use categories used to define the surface albedo, moisture availability, longwave emissivity (9 μm), roughness length, thermal inertia, and surface heat capacity per unit volume (Grell et al., 1993). The model was set up according to Seaman (1996) to cover the Northeast model domain with 32 vertical layers (lowest level $\sim$18 m). The model version that was used employed the Blackadar planetary boundary layer parameterization (Zhang and Anthes, 1982). July 12 (7:00 EST) to July 13 (11:00 EST) was used to initialize meteorological conditions.

Two sets of MM5 simulations were performed: (1) base case with land use unchanged; and (2) high tree with 40% of the urban cells changed to deciduous forest (Civerolo et al., 2000). Both model simulations used the same initial boundary conditions (Seaman, 1996). The selection of the 40% of the urban cells (189 grid cells) changed to deciduous forest was based on: (a) cells not containing airports or anthropogenic emissions from stacks > 65 m, and (b) cells along the periphery of metropolitan areas being changed more frequently than inner-city/downtown cells.

2.3. Dry deposition modeling

To estimate hourly deposition velocities (V_d) in urban areas, hourly tree-canopy and grass/soil surface resistances (r_c) for O_3, sulfur dioxide (SO_2), and nitrogen dioxide (NO_2), as well as aerodynamic resistance (r_a) and quasi-laminar boundary-layer resistance (r_b) were calculated based on a hybrid of big-leaf and multi-layer canopy deposition models (Baldocchi et al., 1987; Baldocchi, 1988) as part of the Urban Forest Effects (UFORE) model (Nowak et al., 1998b). Resistance (r_c) to artificial surfaces (e.g., buildings, roads) was based on Wesely (1989). Ground area under tree canopies was estimated to be 94% grass/soil and 6% artificial surfaces (Nowak, unpubl. data). Total resistance (r_c) in urban areas was estimated based on parallel resistances of tree, grass/soil and artificial surfaces for: (1) base case – 20% tree cover, 30% grass cover, and 50% artificial surface cover; (2) high tree case – 40% tree cover, 10% grass cover, and 50% artificial cover; and (3) deposition change case – 50% grass cover and 50% artificial cover. Total $V_d = (r_a + r_b + r_c)^{-1}$ (Baldocchi et al., 1987). Deposition velocities calculated by the UFORE model for each case were substituted in place of SAQM default values.

To estimate the effect of change in V_d on total change in hourly ozone concentration (ΔC in ppb h^{-1}), the following equation was used:

$$\Delta C = V_d C \, \mathrm{BL}^{-1} \tag{1}$$

where C is ozone concentration in lowest layer (ppb), BL is boundary layer height (m) and V_d is in m h^{-1}.

2.4. Anthropogenic emissions modeling

Anthropogenic emissions were calculated with the SARMAP/LMOS Emissions Modeling System, commonly referred to as EMS-95 (Wilkinson et al., 1994). EMS-95 consists of three major components: low-level area (e.g., gas stations, shopping centers, and homes), low-level mobile (e.g., motor vehicles and boats), and point sources (e.g., power plant smoke stacks).

Both area and point sources are based on state-supplied specific emission factors to calculate hourly gridded emissions of NO_x, CO, and VOCs typical of representative weekdays and weekends. The mobile source component requires gridded hourly temperatures to calculate mobile source emissions, which were obtained from MM5 simulations.

Three sets of anthropogenic emissions were modeled: (1) base – temperatures from base MM5 simulation; (2) high tree – high tree MM5 temperatures, motor vehicle emissions based on new MM5 temperatures, and a 5% reduction in utility point sources due to reduced demand for air conditioning resulting from increased tree cover; and (3) land use change – anthropogenic emissions were removed in 40% of the urban cells (cells changed to deciduous forest cover in MM5 simulation).

2.5. Biogenic emission modeling

To estimate biogenic emissions, the MCNC Sparse Matrix Operator Kernel Emissions – Biogenic Emissions Inventory System (SMOKE-BEIS2) was used (Geron et al., 1994; Houyoux et al., 1996; Williams et al., 1992). This model requires land cover/land use, species-specific emission factors, temperatures, and photosynthetically active radiation (PAR) to generate hourly gridded estimates of aldehydes, isoprene, nitric oxide, paraffins, and olefins. Temperature data were obtained from two MM5 simulations. PAR was calculated with gridded surface pressure, cloud height, and cloud fractional coverage also from the MM5 simulations (Iqbal, 1983).

Four sets of biogenic emissions were developed: (1) base emissions, with base MM5 temperatures reflecting new land cover/land use information in 83 urban cells (New York, Philadelphia, Baltimore) based on field data; (2) high tree emissions, with high tree MM5 temperatures and biogenic emissions in the 189 modified urban cells (40%) scaled up to 100% tree cover; (3) same as (2) but with base temperatures (i.e., no temperature change); (4) high tree/low emitter emissions, identical to (2) except that all trees in the urban cells were replaced with low ($<$0.1 μg C per g dry foliar mass per hour) VOC emissions (*Prunus* spp), and (5) reduced tree, with base MM5 temperatures, and all trees in the 473 urban cells replaced with 'open forest', a category with grass-like (low) hydrocarbon emissions and low NO emissions (Pierce et al., 1998).

2.6. Air quality modeling

Air quality modeling was performed using the SARMAP Air Quality Model (SAQM) (Lu and Chang, 1998). SAQM calculates gridded hourly concentrations of 29 species, using the Carbon Bond IV mechanism. The model was initialized using clean boundaries (e.g. $[O_3] = 40$ ppbv) and simulated one day using the first day meteorology and emissions. Subsequently, fields from the last hour were extracted as the new initial and boundary conditions and the first day was rerun. This process was repeated again such that three ramp-up days were used to smooth out initial conditions before model output began for July 13 (11:00 EST). SAQM simulations had 16 vertical levels to match the structure in MM5. All concentrations reported correspond to the lowest model layer, ~ 18 m above the surface.

For the BASE model scenario, MM5 (an input into SAQM) model inputs were based on urban land use data. BEIS-2 and UFORE inputs were based on field data and 20% urban tree cover. For the high tree scenarios (HITRE and HITRE-LOE), MM5, BEIS-2, and UFORE were increased to 40% urban tree cover. For MM5, 40% of the urban cells were converted to forest; for BEIS-2 and UFORE, canopy cover was directly increased in these vegetation models. Thus, the high tree scenarios are essentially the effect of increasing canopy cover in urban areas, as anthropogenic emissions either remained the same (HITRE) or were altered due to changes in energy use and air temperature from trees (HITRE-LOE). However, the meteorological modeling (MM5) was based on altering the land use from urban to forest.

The appropriateness of the models used for meteorological and air quality analyses have been examined in other studies (e.g., Dudhia, 1993; Geron et al., 1997; Lu and Chang, 1998; Pierce et al., 1998; Seaman et al., 1995). The objective of this study was not to use the modeling results in an absolute sense, rather, it was to examine the changes in model output relative to model input. The models are more useful when they are used as tools for directional and relative analyses than in an absolute sense (Rao et al., 1996). This study specifically examined response of meteorological and air quality variables in response to changes in urban vegetation inputs.

3. Results

Under the BASE scenario, hourly urban temperatures averaged between 25.2°C and 34.7°C, and hourly ozone concentrations for all urban cells averaged between 25 and 66 ppb (Fig. 2), with a peak ozone level within an urban cell of 108 ppb. BASE urban boundary layer heights ranged between 90 m (6:00 EST) and 1050 m (12:00 EST), while horizontal wind speeds ranged between 4.2 m s^{-1} (5:00 EST) and 6.1 m s^{-1} (15:00 EST).

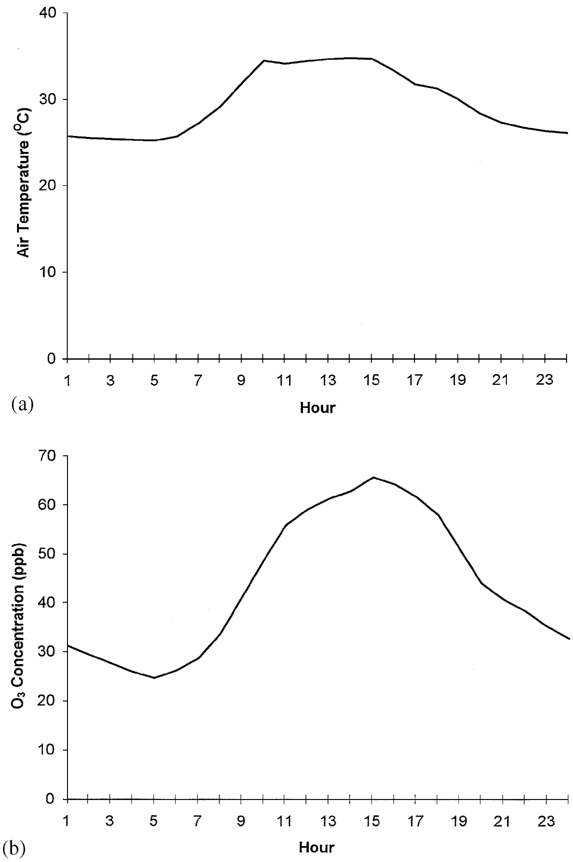

Fig. 2. Average hourly (a) air temperature and (b) ozone concentration in urban cells for July 13 (11:00 EST) to July 15 (14:00 EST), 1995.

Changes in VOC emissions and deposition velocities (V_d) in the urban cells of BASE case from default BEIS2 and MM5/SAQM model values (i.e., model values prior to changing urban tree leaf biomass and deposition velocities based on field data and canopy models) were negligible for VOC emissions and most significant for V_d. Average O_3 V_d remained unchanged during the day (0.27 cm s^{-1}), but dropped from 0.19 cm s^{-1} (MM5/SAQM default) to 0.09 cm s^{-1} (BASE) at night; average NO_2 V_d increased from 0.12 cm s^{-1} (MM5/SAQM default) to 0.19 cm s^{-1} (BASE) during the day, and from 0.04 to 0.07 cm s^{-1} (BASE) at night. Base VOC emissions in urban areas during the daylight hours averaged 1130 mol h^{-1} (BEIS2).

Daily anthropogenic hydrocarbon emissions across the model domain were approximately 2.14×10^6 mol for the BASE case, while domain-wide biogenic hydrocarbon emissions were approximately 2.11×10^6 mol (49.6% of total domain hydrocarbon emissions). Isoprene comprised about 78% of the total biogenic hydrocarbon emissions.

3.1. High tree scenario (40% tree cover)

An increase in urban tree cover from 20% (BASE) to 40% (HITRE) led to an increase in average hourly NO_x concentrations during most of the day and a decrease in concentrations during most of the night (Fig. 3a); a decrease in average hourly O_3 concentrations during the day (except 13:00 EST) and an increase in concentrations during most of the night (Fig. 3b); and a decrease in 8-hour average O_3 concentrations (forward average assigned to first hour) from 23:00 EST to 18:00 EST (Fig. 3c). Urban biogenic hourly VOC emissions increased an average 75% throughout the day, 70% (720 mol h^{-1}) during daylight hours, with a minimum increase of 61% at 10:00 EST. Average hourly ozone concentrations decreased due to increased tree cover (Table 2). Average V_d increased 0.14 cm s^{-1} for O_3 and 0.07 cm s^{-1} for NO_2 during the day, and 0.02 cm s^{-1} for O_3 and 0.01 cm s^{-1} for NO_2 during the night. Change in ozone concentration was partially predicted by the change in ozone deposition velocities from Eq. (1) (Fig. 4a).

Increased tree cover also led to a decrease in air temperature (Fig. 5a), horizontal wind speed (Fig. 5b), and boundary layer height (Fig. 5c) throughout the day, with boundary layer height decreases most significant during daylight hours.

If temperature reductions were not accounted for in biogenic VOC emissions, these emissions would have been higher throughout the day, particularly during the daylight hours when emissions would have increased an average 140 mol h^{-1} (7.1%) with a peak increase of 290 mol h^{-1} (11.4%) at 10:00 EST. Increased urban tree cover lead to a decrease in anthropogenic emissions of carbon monoxide (20,000 mol d^{-1}; 0.3%), nitrogen oxides (30,000 mol d^{-1}; 2.5%), and VOCs (60,000 mol d^{-1}; 2.8%).

Increased urban tree cover also had an impact on surrounding ozone concentrations, particularly in downwind areas to the north and/or east of the urban area (Civerolo et al., 2000). In these areas, ozone concentrations were increased in some localities (30% of total cells) and decreased in others (20% of total cells), with the largest proportion of cells with ozone changes occurring at night. For the overall modeling domain, daytime ozone increased an average of 0.26 ppb (0.5%), with concentrations decreasing from 4:00 to 10:00 EST and a maximum average increase of 0.7 ppb (1.2%) at 13:00 EST.

For the entire model domain, biogenic VOC emissions increased throughout the day, averaging about 10 mol h^{-1} at night with a maximum average hourly increase of 30 mol h^{-1} (2.2%) at 13:00 EST. Temperature decreased for all hours except 22:00–23:00 EST, with an average daily decrease of 0.04°C and 44% of the cells decreasing in temperature and 35% exhibiting a temperature increase. Horizontal wind speeds decreased throughout the day an average of 0.01 m s^{-1} (0.2%). NO_x concentrations dropped an average of 0.1 ppb between 20:00 and 8:00 EST and had a peak increase of 0.3 ppb (14.6%) at 11:00 EST. Average boundary layer

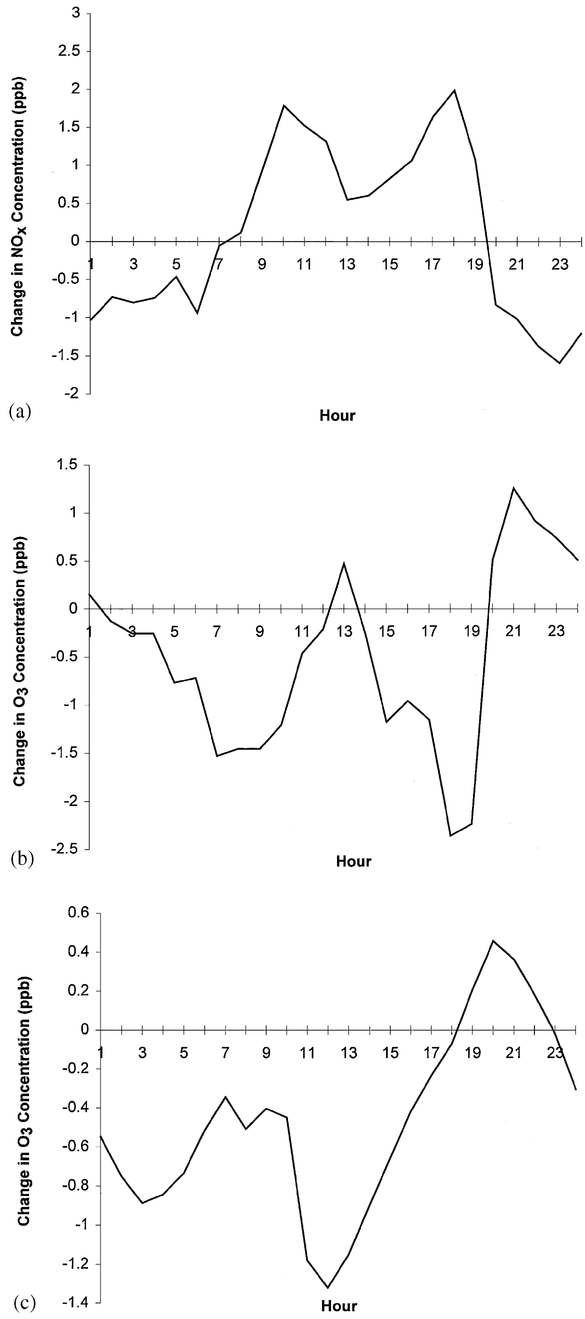

Fig. 3. Change in (a) average hourly NO_x concentration, (b) average hourly O_3 concentration and (c) eight-hour average O_3 concentration in urban cells from BASE to HITRE scenario (HITRE – BASE).

Table 2
Average and peak change in hourly O_3 concentrations (ppb), and biogenic VOC emissions ($mol\ h^{-1}$) in urban cells for SAQM simulations

	Daylight hours		Entire day			
SAQM	Avg. Δ O_3	Avg Δ VOC	Avg. Δ O_3	Peak Δ O_3	8-h avg Δ O_3	8-h peak Δ O_3
HITRE[a]	− 1 (2.4%)	+ 720 (70%)	− 0.5 (1.2%)	− 2.4 (4.1%)	− 0.5 (1%)	− 1.3 (2.2%)
HITRE-LOE[b]	+ 0.2 (0.6%)	na	+ 0.3 (1%)	+ 0.7 (2.3%)	+ 0.3 (0.9%)	+ 0.6 (2.1%)
SPPΔ[c]	0 (0%)	− 1,380 (69%)	0 (0%)	0 (0%)	0 (0%)	0 (0%)
LANDΔ[d]	+ 0.6 (2.3%)	na	+ 1.6 (5.2%)	+ 4.2 (15.8%)	+ 1.7 (4.8%)	+ 3.5 (11.7%)
VdVOCΔ[e]	+ 0.4 (1.1%)	− 1,010 (88%)	+ 0.3 (1%)	+ 0.9 (3.7%)	+ 0.3 (0.9%)	+ 0.5 (1.7%)

[a] HITRE vs. BASE: adding urban trees
Average daylight temperature change = − 0.6°C (Fig. 5a).
Average daylight wind speed change = − 0.27 $m\ s^{-1}$ (Fig. 5b).
Average daylight boundary layer height change = − 99 m (Fig. 5c).
[b] HITRE-LOE vs. HITRE: utility and auto emission reductions due to urban trees.
[c] Change tree species in HITRE to low emitters (*Prunus* spp.).
[d] Remove anthropogenic emissions from HITRE in 40% of urban cells.
[e] Remove trees from BASE with no change in meteorology or anthropogenic emissions.
na – not applicable; SAQM case did not include any change that would affect biogenic VOC emissions.

heights across the domain were only slightly decreased (<1 m; <0.1%) by increased urban tree cover.

3.2. Energy conservation effect

Reduction of utility emissions by 5% and motor vehicle emissions (HITRE-LOE) from high tree scenario (HITRE) lead to a decrease in NO_x concentrations within urban areas throughout the day, but particularly at night (Fig. 6a); an increase in hourly average O_3 concentrations from 20:00–13:00 EST (Fig. 6b); and an increase in 8-hour average O_3 concentration for all hours except 11:00–14:00 EST (Fig. 6c). The highest ozone increases occurred at night. Average hourly ozone concentrations increased due to reduced utility and motor vehicle emissions from increased urban tree cover (Table 2). For the entire domain, average hourly ozone concentrations dropped between 13:00 and 21:00 EST with a peak decrease of 0.2 ppb (0.3%) at 12:00 EST. The peak average increase was at 4:00 EST (0.2 ppb; 0.6%).

3.3. Species composition effect

Changing the existing urban tree species composition (SPPΔ) to all low VOC emitting species (*Prunus* spp.) decreased hourly VOC emissions from urban trees an average 60% throughout the day, 69% (1380 $mol\ h^{-1}$) during daylight hours, with the highest percent decrease of 76% at 10:00–12:00 EST. These changes in VOC emissions, which were greater than the VOC changes in the VdVOCΔ and HITRE cases, had no impact (<1 ppb) on modeled ozone concentrations (Table 2).

3.4. Land use change effect

Removing anthropogenic emissions (LANDΔ) from 40% of the urban cells in the HITRE-LOE scenarios led to a similar pattern as exhibited by the energy conservation effect (Fig. 6) but to a greater degree. Hourly ozone concentrations increased due to reduced anthropogenic emissions (Table 2), particularly during the night (3.2 ppb; 9.9%). For the entire domain, average hourly ozone concentrations dropped between 9:00 and 19:00 EST with a peak decrease of 0.3 ppb (0.5%) at 12:00 EST. The peak average increase was at 4:00 EST (0.3 ppb; 0.8%).

3.5. Reduced deposition velocity and VOC emission scenario

Reductions in deposition velocity and VOC emissions without meteorological and anthropogenic emission changes (VdVOCΔ) from the BASE case led to an increase in average hourly NO_x and O_3 concentrations (Fig. 7). Urban biogenic VOC emissions dropped an average 84% throughout the day, 88% (1010 $mol\ h^{-1}$) during daylight hours, and peaked at 90% from 8:00–16:00 EST. Hourly ozone concentrations increased due to the loss of urban tree cover (Table 2). Average V_d decreased 0.11 $cm\ s^{-1}$ for O_3 and 0.07 $cm\ s^{-1}$ for NO_2 during the day, and 0.02 $cm\ s^{-1}$ for O_3 and 0.01 $cm\ s^{-1}$ for NO_2 during the night. Because urban tree VOC emissions had no detectable impact on ozone concentrations, the changes in ozone concentrations in this scenario are attributed to changes in deposition velocity. The change in ozone concentration was closely predicted by the change

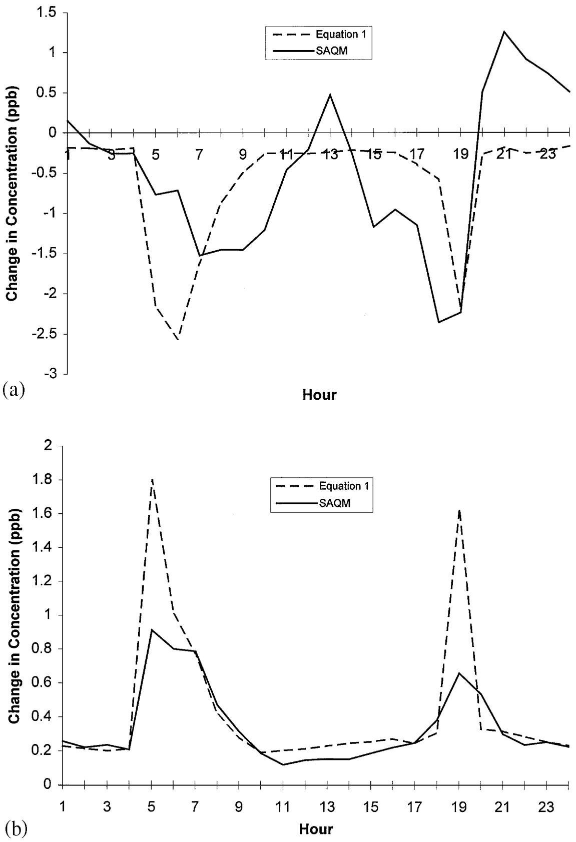

Fig. 4. Change in O_3 concentration in urban cells due to (a) HITRE scenario and (b) VdVOCΔ scenario (SAQM) versus predicted change in concentration from equation (1) based on deposition velocity and boundary layer height.

in ozone deposition velocities from Eq. (1), except for early morning and late afternoon periods (Fig. 4b).

4. Discussion

Though often regarded as a significant factor in ozone formation, VOC emissions from additional urban trees had no detectable effect (<1 ppb) on ozone concentrations in this study. This lack of effect is most likely due to the area being mostly NO_x limited and the relatively large amount of total anthropogenic (2.14×10^6 mol d^{-1}) and biogenic (2.11×10^6 mol d^{-1}) VOC emissions in the model domain, compared with the increase in urban tree VOC emissions (1.38×10^4 mol d^{-1}). Although urban trees VOC emissions are likely to have some effect on ozone formation, the model may not be sensitive enough to detect these relatively minor changes (<1 ppb). Other physical factors of the trees (e.g., changes in V_d) were detected by the model, indicating the relative difference in these physical effects compared to changes in urban tree VOC emissions. Although species composition had no effect on modeled ozone concentrations in this study, selection of low VOC-emitting urban trees can impact ozone concentrations in some cities (Taha, 1996).

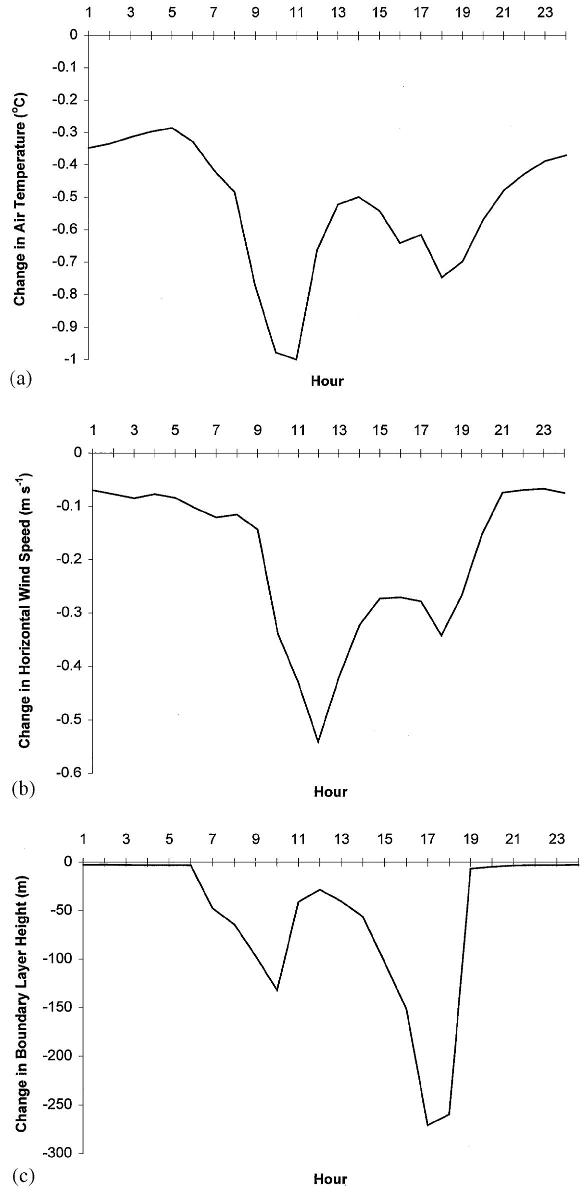

Fig. 5. Change in (a) average hourly air temperature, (b) average hourly horizontal wind speed and (c) average hourly boundary layer height in urban cells from BASE to HITRE scenario (HITRE-BASE).

Some of the most significant effects of urban trees on ozone concentration in this study are due to changes in the atmospheric physical environment, particularly dry deposition and reduced air temperature, wind speeds and boundary layer heights. The interaction between

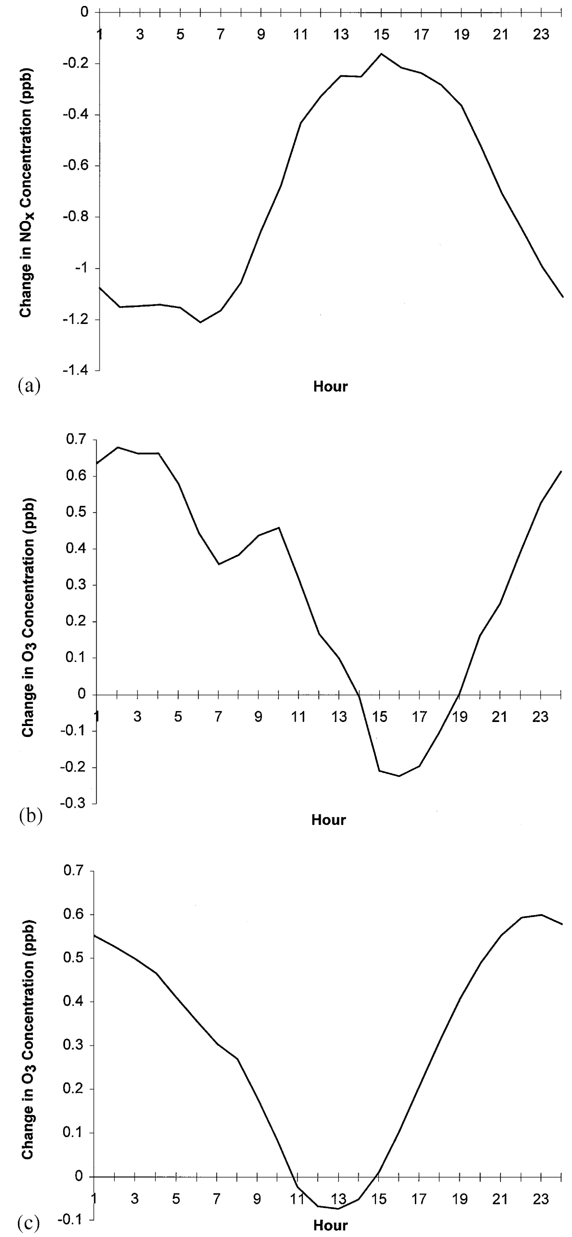

Fig. 6. Change in (a) average hourly NO_x concentration, (b) average hourly O_3 concentration and (c) eight-hour average O_3 concentration in urban cells from HITRE to HITRE-LOE scenario (HITRE-LOE-HITRE).

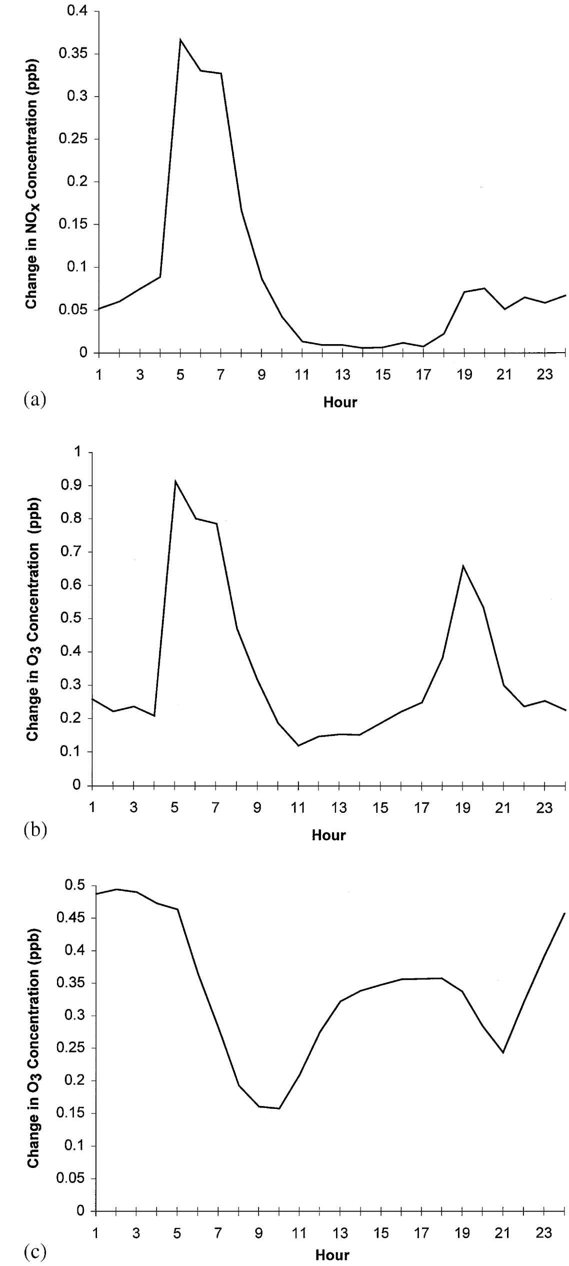

Fig. 7. Change in (a) average hourly NO_x concentration, (b) average hourly O_3 concentration and (c) eight-hour average O_3 concentration in urban cells from BASE to VdVOCΔ scenario (VdVOCΔ-BASE).

deposition velocity and boundary layer height plays a significant role in the overall effect of trees on ozone concentrations. Although deposition velocities increase in the daytime (vs. night), as the boundary layer height increases during the day, the impact of urban tree deposition diminishes (Eq. (1)) (Fig. 4b). Thus, peak effects of urban tree deposition on ozone occur in the early morning and late evening when trees are transpiring and boundary layer heights are relatively low. The difference between the predicted (Eq. (1)) and SAQM model ozone concentrations during the early morning and late evening (Fig. 4b) are likely due to atmospheric factors related to changes in the stability during these times and the relatively high increase in NO_x during these periods (due to a relatively high loss of NO_x deposition when boundary layers are low), which would increase NO_x scavenging of O_3 and thus reduce O_3 concentrations.

In the HITRE scenario, when tree cover was increased to 40% with corresponding meteorological changes, it appears that numerous physical factors are interacting to produce the exhibited changes in O_3 concentrations. During the nighttime, O_3 increases mainly due to reduced horizontal wind speed (reduced dispersion; increased stability) and increased NO_x deposition, which leads to lower NO_x concentrations and reduced NO_x scavenging of ozone. During the early morning, ozone concentrations are reduced at an increasing rate as transpiration increases but boundary layer heights remain relatively low. Eq. (1) over predicts the change in O_3 concentration during this period due to changing atmospheric stability and the relatively high impact of increased NO_x deposition on increasing O_3 concentration during this period (Fig. 4a).

During the remainder of the day, ozone concentrations are generally reduced more than what the deposition process alone would predict. This difference is due to a combination of reduced emissions of carbon monoxide and anthropogenic VOCs (total anthropogenic VOC emissions dropped 6×10^4 mol d^{-1}) due to cooler air temperatures (reduces O_3), reduced air temperatures that reduces ozone formation rate (reduces O_3) (Carter et al., 1979), and reduced horizontal wind speeds, which would reduce pollutant dispersion (increase O_3). The effect of factors that reduce O_3 concentrations is diminished as horizontal wind speed reductions peak toward midday (Fig. 5b). This wind speed reduction counterbalances some of the ozone reductions such that there is a net increase in ozone concentrations at 13:00 EST (Fig. 4a).

The reduction in boundary layer heights throughout the day due to increased tree cover has a potential to increase pollutant concentrations due to limiting vertical mixing of pollutants, particularly during unstable (well-mixed) daytime conditions, and can decrease ozone concentrations due to increased titration of ozone by NO. However, reductions in boundary layer heights greatly enhance the relative magnitude of surface deposition effects in reducing pollutant concentrations (Eq. (1)). As ozone concentrations decrease during daytime periods, when boundary layer heights were reduced the most, it is likely that the enhanced relative magnitude of surface deposition effects, along with other tree effects, compensates for the loss of vertical dispersion of pollutants. Boundary layer height reductions averaged 56 m (9.5%) throughout the day with peak reductions of 271 m (30.8%) at 17:00 EST and 38.2% (260 m) at 18:00 EST. Boundary layer height reductions decreased in mid-afternoon as the boundary layer height reached its maximum.

Increased tree cover also leads to a reduction in urban NO_x concentrations during the night, due to increased NO_x deposition, and an increase in NO_x concentrations during the day (Fig. 3a). The daytime increase is likely due to reduced horizontal wind speeds (Fig. 5b) that would result in reduced pollutant dispersion. The reduced increase in NO_x concentrations during midday (Fig. 3a) is likely due to an interaction with increased ozone formation (Fig. 3b).

The regional increases in ozone concentrations due to increased urban tree cover are likely attributable to the overall reduction in wind speeds, which would reduce pollutant dispersion, and decreased NO_x concentrations at night, which would reduce NO_x scavenging of O_3. The majority of the regional effect, both increases and decreases of ozone in surrounding cells, was concentrated in the downwind areas to the north and/or east of the urban areas. During the daylight hours, average hourly ozone concentrations increased 0.26 ppb. In the urban areas, ozone concentrations dropped an average of 1 ppb during this same period. Thus, the daytime reduction in ozone concentrations in urban areas were three to four times greater than the regional increase in ozone concentrations.

Reduction in anthropogenic emissions due to energy conservation effects of trees (HITRE-LOE) or land use changes (LANDΔ) generally increased ozone concentrations. Urban ozone concentrations were increased throughout the night due to loss of NO_x emissions and NO_x scavenging of O_3 (Fig. 6b). The diurnal pattern of reduced ozone concentrations during the afternoon (Fig. 6b) is likely due to the counterbalancing effect of the loss of anthropogenic VOC emissions and the relatively small reduction in NO_x concentration that occurred during the afternoon. The net overall effect of increased tree cover along with reduction in utility and motor vehicle emissions (comparing HITRE-LOE to BASE) was still positive throughout the daytime (5:00–19:00 EST) with an average decrease in ozone concentrations of 0.9 ppb (1.9%). The magnitude of the changes exhibited in this study are relatively small and reasonable given the relatively small changes to the model inputs, and are fairly comparable to changes exhibited in other studies (e.g., Roselle and Schere, 1995).

Pollutant deposition to trees appears to play an important role in removing atmospheric pollutants and improving air quality in cities. This positive effect and the positive effect of reduced air temperatures, is partially offset by reduced wind speeds and dispersion, and could be offset in some areas of the country by increased biogenic VOC emissions (e.g., Taha, 1996). While urban tree cover reduces ozone concentrations in cities, removal of anthropogenic emissions (LANDΔ) in 40% of the urban cells leads to an overall increase in ozone concentrations of 1.6 ppb (5.2%), primarily due to loss of NO_x scavenging of ozone. Even though reduced anthropogenic and utility NO_x emissions lead to a local, and primarily nighttime, increase in ozone, reduced emissions will likely lead to a regional and long-term improvement in overall air quality (OTAG, 1998).

Existing urban trees in the Northeast appear to reduce urban ozone concentrations as evidenced by the increase in ozone concentrations when urban trees are removed (VdVOCΔ). The estimated ozone reduction by existing trees (0.4 ppb during daylight hours) (Table 2) is conservative as meteorological changes (e.g., increased air temperatures) due to the loss of existing trees were not considered in this scenario.

The relationship between urban trees and ozone indicates that maintaining existing urban tree cover could help prevent ozone concentrations from increasing, and that increased tree cover throughout areas with high ozone concentrations (e.g., ozone non-attainment areas) can be a viable strategy to help reduce ozone in these areas. However, managers need to be aware that this increased tree cover could lead to a slight overall increase in ozone concentrations in surrounding areas.

In addition, the air quality impact of deposition to urban trees would also be positive for other urban pollutants that are readily deposited upon and within urban trees (e.g., nitrogen dioxide, sulfur dioxide, fine particulate matter) (Nowak et al., 1998b). Thus, sustained or increased urban tree cover will reduce concentrations of other pollutants besides ozone. However, reduced wind speeds and dispersion in surrounding areas could again lead to overall regional increases in pollutant concentrations.

An increase to 40% urban tree cover may be unreasonable given the existing morphology of many cities. Urban tree cover in the United States averages 27.1%, with cities in forests averaging 31.1% tree cover, cities in grasslands, 18.9%; and cities in deserts, 9.9% (Nowak et al., 1996). Individual city tree cover has reached as high as 55% in Baton Rouge, LA (Nowak et al., 1996). Thus, urban tree cover of 40% or more are attainable in some circumstances, but surrounding natural environment and city morphology often limit urban tree cover.

Management efforts to sustain or increase urban tree cover will likely help sustain or improve city air quality, particularly in areas immediately around the area of increased tree cover. Location and magnitude of surrounding regional increases in ozone concentrations need to be considered. If surrounding areas have relatively low ozone concentrations, the gain of reduced ozone concentrations in poor air quality areas due to increased tree cover will likely more than offset the relatively low increase in regional ozone concentrations when considering the overall societal impacts of ozone.

5. Summary

From the patterns and results revealed in these analyses, it is apparent that urban trees have a locally positive effect by reducing ozone in urban areas of the Northeast, but tend to increase overall regional ozone concentrations. During the daytime, local ozone reductions (1 ppb) due to increased urban tree cover are greater than the overall regional ozone increases (0.26 ppb). The physical effects of vegetation changes on ozone concentrations also appear to be more important than atmospheric chemical interactions with biogenic VOC emissions from urban trees in the Northeast. Pollutant deposition to trees has a significant impact on reducing ozone levels, but this effect is diminished as the depth of the boundary layer increases. Pollutant deposition of NO_x tends to increase nighttime ozone concentrations due to the loss of NO_x scavenging of ozone. The effect of trees on reducing horizontal wind speed tends to increase ozone concentrations both locally and regionally due to diminished pollutant dispersion in the atmosphere. Shifting to low VOC emitting tree species had little impact on ozone concentration (< 1 ppb), but use of low emitting species could still help reduce ozone levels in some urban areas. Future pollution modeling efforts should further explore the role of vegetation and atmospheric physics on ozone concentrations at both the local and regional levels. Though uncertainty is inherent in all modeling results, it appears that increasing urban tree cover in non-attainment areas of the Northeast is a viable strategy to help reduce local ozone levels, but may also lead to slight overall increases in ozone concentrations in surrounding areas.

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