

## Spatial heterogeneity and air pollution removal by an urban forest

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### ABSTRACT

Estimates of air pollution removal by the urban forest have mostly been based on mean values of forest structure variables for an entire city. However, the urban forest is not uniformly distributed across a city because of biophysical and social factors. Consequently, air pollution removal function by urban vegetation should vary because of this spatial heterogeneity. This paper presents a different approach to evaluate how the spatial heterogeneity of the urban forest influences air pollution removal at the socioeconomic subregion scale. Air pollution removal for July 1997 to June 1998 and July 2000 to June 2001 were estimated using measured urban forest structure data from three socioeconomic subregions in Santiago, Chile. Dry deposition was estimated using hourly climate, mixing height, and pollutant concentration data. Pollution removal rates among the three socioeconomic subregions were different because of heterogeneous urban forest structure and pollution concentrations. Air pollution removal per square meter of tree cover was greatest in the low socioeconomic subregion. Pollution removal during 1997–1998 was different from 2000 to 2001 due to pollution concentration differences. Seasonal air quality improvement also differed among the subregions. Results can be used to design management alternatives at finer administrative scales such as districts and neighborhoods that maximize the pollution removal rates by the urban forest in a subregion. Policies that affect the functionality of urban forest structure must consider spatial heterogeneity and scale when making region-wide recommendations. Similarly, when modeling the functionality of the urban forest, models must capture this spatial heterogeneity for inter-city comparisons.

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### 1. Introduction

Air pollution in urban areas is a significant environmental problem as it affects human health and well-being. The effects of exposure to air pollution can vary from premature mortality to many chronic effects such as reduced physical ability and capacity, coughing, airway problems, permanent damage to lungs, and emphysema (Eskeland, 1997; Powe and Willis, 2004; Samet et al., 2000; World Bank, 1997). The effects are manifested in costs to society in lost workdays, increased medical bills, and subsequent losses in productivity. Air pollution also reduces visibility, damages infrastructure, and can kill vegetation (Chameides et al., 1988; Eskeland, 1997; Kozlowski, 1980; Romero et al., 1999; World Bank, 1994). We will not review the extensive body of literature on the effects of air pollution on vegetation (see Beckett et al., 1998, 2000; de Bauer and Krupa, 1990; Kozlowski et al., 1991; Powe and Willis, 2004).

Urban forests can affect local and regional air quality by removing atmospheric pollutants, emissions of atmospheric chemicals from the vegetation and its maintenance, altering urban microclimates by lowering temperatures through shading and evapotranspiration, changing wind patterns, modifying boundary layer heights, and reducing building energy use and consequent emissions from power plants (Beckett et al., 1998, 2000; Chameides et al., 1988; McPherson et al., 1998, 1999; Scott et al., 1998; Nowak et al., 1998a,b, 2006; Sharkey and Singsaas, 1995; Yang et al., 2005). Urban forests also influence global climate change through direct removal of greenhouse gases and by affecting emissions from energy production (McPherson, 1994; McPherson et al., 1999).

Several studies have quantified the amount of air pollution removal by urban forests (Free-Smith et al., 1997; McPherson et al., 1999; Yang et al., 2005). For example, Nowak et al. (2006) studied air pollution removal and air quality improvement by urban forests for several cities in the United States. Using assumed urban forest structure values such as leaf area index, estimated mean removal of PM<sub>10</sub> by trees in Los Angeles, United States was 8.0 g m<sup>-2</sup>. Yang et al. (2005) discuss the role of urban forests on air quality in Beijing and found that pollution removal rates by its urban forest were greater than those for cities in the United States. Freer-Smith et al. (1997),

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Beckett et al. (1998, 2000), and Powe and Willis (2004) discuss particulate matter removal by woody vegetation in Britain. McPherson et al. (1999) analyzed the benefits and costs of managing municipal urban forests for multiple benefits and services, including air quality.

Studies on the cost-effectiveness of the use of urban forests to improve air quality present mixed results. For example, McPherson et al. (1998) analyzed the cost-effectiveness of yard trees for air quality improvement and found that limited tree plantings were not-cost effective. The results and the approach of this study were questioned by Nowak et al. (1998a). Escobedo et al. (2008) determined that urban forest management programs and policies aimed at particulate matter reduction in Chile were cost-effective. Additional literature on the effects of urban forests on air quality can be found in Scott et al. (1998) and Smith (1990).

Most examples from the literature model air pollution removal by an urban forest as a function of pollution concentration and dry deposition (Smith, 1990). With few exceptions, these pollution removal estimates are based on mean forest structure parameters (e.g., leaf area, leaf area index, and biomass) over an entire modeling region, often the entire city. Some studies have stratified modeling regions and quantified urban forest structure by land use but the air pollution removal function was not quantified (McPherson et al., 1998, 1999; Nowak and Crane, 2000; Yang et al., 2005). However, the urban forest is not uniformly distributed across a city because of different biophysical and anthropogenic factors such as land use, soils, socioeconomic variables, and human values (Escobedo et al., 2006; Heynen and Lindsey, 2003; Zipperer et al., 1997). Because of the importance of trees and their influence on atmospheric processes, Zipperer et al. (1997) recommends defining different patches of tree cover by land use to capture the spatial heterogeneity of vegetation cover. Other finer scale evaluations may include socio-economic factors such as census blocks.

However, analyzing the effects of the spatial, temporal, and social variability of the urban environment on environmental quality can also present several problems. For example, Grimmond et al. (2002) found measurements of CO<sub>2</sub> fluxes in urban environments to vary according to spatial scales (micro-, local- and meso-). Air pollution dynamics within an urban region at the meso-scale (10<sup>4</sup>–10<sup>5</sup> m) are complex and removal rates by vegetation can vary as a result of differences in vegetation structure, pollution concentration, and atmospheric conditions (Grimmond et al., 2002; Nowak et al., 2006). Escobedo et al. (2008) found that particulate matter pollution removal by trees, shrub and grasses varied according to socioeconomic strata of an urban region. Unfortunately, there are few known studies that analyze differences in urban forest structure and air pollution removal in subregions of a city and there are even fewer studies that link a city's urban forest structure and socioeconomic activity with site-specific pollution dynamics through time.

A spatial analysis based on socioeconomic subregions at the local-scale (10<sup>2</sup>–10<sup>4</sup> m) within a city can account for differences in cultural and urban activities and their inherent pollution emission and concentration dynamics (Bertrand and Romero, 1993; Scarpaci et al., 1988; World Bank, 1997). Socioeconomic subregions also differ in their urban forest structures (Escobedo et al., 2006; Heynen and Lindsey, 2003). More refined estimates of pollution removal at the land use, socioeconomic subregion, or local and micro-scale (10<sup>1</sup>–10<sup>2</sup> m) can improve the understanding of the effects of the urban forest on pollution removal and consequently resident's well-being. By understanding how spatial heterogeneity influences pollution removal by urban forests, analysis and understanding of currently available urban forest function models should improve. Spatial heterogeneity in this analysis is defined as the vari-

ability in urban forest structure across the urban landscape (e.g., cover, leaf area index, evergreen leaf compositions). With the addition of a temporal element, one can begin to evaluate the efficacy of management and policies. Using this approach to account for heterogeneity in the distribution of the urban forest, managers and policy makers can gain a better understanding of the urban forest, factors influencing forest cover, and where urban forest structure needs to be modified to maximize its function.

This analysis builds upon previous studies (McPherson et al., 1999; Nowak et al., 2006; Powe and Willis, 2004; Smith, 1990; Yang et al., 2005) and applies the pollution flux approach, which utilizes dry deposition rates and pollution concentrations to quantify the influence of spatial heterogeneity on air pollution removal for the City of Santiago, Chile. The specific goals of this paper are: 1. To evaluate the effect of the spatial heterogeneity on pollution removal by the urban forest at the subregion scale and 2. To examine the role of scale and spatiotemporal heterogeneity of urban forests in management and policy options for improving urban air quality. To capture the spatial heterogeneity of the urban forest in Santiago, we divided the city into three socio-economic areas.

## 2. Methods

### 2.1. Study site

The study was conducted in the 967.2 km<sup>2</sup> Santiago Metropolitan Region in central Chile (Fig. 1). Land uses range from natural vegetation cover (Andean piedmont shrublands) to agriculture to urban and are characterized by a temperate, semi-arid, Mediterranean climate with an average annual precipitation of 375 mm most of which is concentrated during the winter months (World Bank, 1994; CONAMA, 1997). The Santiago Metropolitan Region, with its population of over 5 million (approximately 45% of the country's population), has among the worst urban air quality problems, particularly particulate matter less than 10 microns (PM<sub>10</sub>) pollution, among major Latin American cities (World Bank, 1997).

Santiago's air pollution problem can be attributed to four general factors: (1) economic growth and its related activity, (2) changing urban dynamics, (3) unique topographic position, and (4) meteorological conditions (CONAMA, 1997). With economic growth there has been an influx of industry and people into the metropolitan area of Santiago. This influx has increased air pollution from an expanding industrial sector and motor vehicle pool (Romero et al., 1999; Castañeda, 1999). Auto emissions have increased because of the number of trips per person per day, increased kilometers driven per vehicle, and a reduced average driving speed (World Bank, 1994, 1997). Air pollution in Santiago is further aggravated by its location between mountain ranges that rise to 2000 m, a condition that contributes to permanent subsidence and thermal inversions. The almost permanent thermal inversions are caused by the Pacific lows that stabilize wind patterns and radiation inputs (CONAMA, 1997).

### 2.2. Modeling

Many studies that analyze air pollution removal by urban forests reported results for an entire city or modeling region (e.g., Nowak, 1994; McPherson et al., 1998, 1999; Nowak and Crane, 2000; Nowak et al., 2002, 2006; Scott et al., 1998; Yang et al., 2005). These estimates were derived using a representative weather monitor, an average data set derived from local pollution concentration monitors and total and mean value of urban forest structure variables for a region. Estimated pollution removal was a function of deposition velocity and pollution concentration (Baldocchi et al., 1987; Davidson and Wu, 1990; Lovett, 1994).



Fig. 1. Map of the Santiago, Chile, modeling region including the three subregions and municipalities.

This study takes a different approach and evaluates pollution removal differences among three different socioeconomic subregions (high, medium, and low socioeconomic status). These three socioeconomic subregions were established based on methods from Escobedo (2004) and are comprised of different population demographics, urban forest structures, and ambient pollution dynamics (Table 1). These subregions resulted from differing land uses, economic development, and cultural activity. Subsequently, pollution emission and concentration trends as well as urban forest structure differed among the subregions (Bertrand and Romero, 1993; Browder et al., 1995; Escobedo, 2004; Eskeland, 1997; Romero et al., 1999; Scarpaci et al., 1988).

The Urban Forest Effects (UFORE) model (e.g., Nowak and Crane, 2000; Nowak et al., 2002, 2006) was used to estimate the amount of air pollution removal by Santiago's three urban forest subregions. Using 2002 field data and meteorological and pollution concentration data from July 2000 to June 2001, urban forest effects on pollutant dry deposition (i.e., pollution removal), were quantified using the UFORE model. In addition, pollution concentration and weather data from July 1997 to June 1998 were used to compare air quality improvement differences for the Santiago Metropolitan Region between analysis years. July through June was used as

the modeling year to account for the southern latitude change of seasons.

### 2.3. Field data

Using field data from 200, 0.04 ha randomly located plots, urban forest structure was quantified for the three subregions for input into the UFORE model. Escobedo (2004) provides specific methods used for allocating plots, field measurements, and site conditions. Since urban tree and shrubs influence pollution removal through leaf surface area and percent evergreen leaf area of the population, these same parameters were quantified using the model based on field measurements. Additional urban forest population variables affecting air quality such as leaf periodicity were obtained from literature sources cited in Nowak et al. (2002) and personnel with the University of Chile Herbarium (personal communications with Maria Teresa Serna; Botanist University of Chile).

### 2.4. Leaf area and leaf area index

Leaf area index (LAI) is the total amount of one-sided leaf area of woody plant foliage per unit area of ground. For UFORE, this value

Table 1

Average annual incomes and population differences in Santiago.

| Subregion | City area (km <sup>2</sup> ) | Average annual income (USD, 2004) | 1995 urban density (pop/km <sup>2</sup> ) | Urban population (1995) <sup>a</sup> | Population (2000) <sup>b</sup> |
|-----------|------------------------------|-----------------------------------|---|--------------------------------------|--------------------------------|
| High      | 164.9                        | 10,000                            | 4308                                      | 710,373                              | 773,633                        |
| Medium    | 370.3                        | 4,000                             | 4328                                      | 1,602,795                            | 1,924,767                      |
| Low       | 431.9                        | 1,250                             | 5694                                      | 2,459,189                            | 2,823,864                      |

Source: Escobedo et al., 2006.

<sup>a</sup> Does not include inhabitants living in rural portions of the city.

<sup>b</sup> Includes both rural and urban inhabitants.

is calculated for only trees and shrubs. This index influences the amount of pollution removed and plant productivity (Kozlowski et al., 1991; Scurlock et al., 2001). Amount of canopy cover and its associated LAI are the main urban forest variables used to estimate the amount of dry deposition by urban vegetation (Nowak and Crane, 2000). The UFORE model estimates leaf area using regression equations (Nowak, 1996; Nowak et al., 2002, 2006) with field data measurements as input variables. Shrub leaf area was estimated from leaf biomass estimates using measured species conversion ratios.

### 2.5. Meteorological data

Several weather stations located in each of the three subregions did not measure data required for the UFORE model. Hence, hourly meteorological data from the *La Platina* weather station located in the south, central area of the modeling region was used to estimate dry deposition velocities for both modeling periods and for all three subregions. Meteorological data included air temperature ( $^{\circ}\text{C}$ ), precipitation (cm), wind speed ( $\text{m s}^{-1}$ ), barometric pressure (mmHg), relative humidity (%), and measured hourly total solar radiation ( $\text{Whr}^{-1} \text{m}^{-2}$ ). Hourly mixing heights (m) were used in calculating air quality improvement. Mixing heights were determined by the Chilean National Center for the Environment (CENMA) from direct hourly wind profiler and Radio Acoustic Sounding System (RASS) measurements from this same weather station using methods found in Seibert et al. (2000).

### 2.6. Pollution concentration data

The MACAM-2 (*Red de Monitoreo Automática de Contaminantes Atmosféricos*) monitoring station network hourly pollution data were used to measure atmospheric pollution concentrations in the three modeling subregions for the periods July 1997 to June 1998 and July 2000 and June 2001. Hourly pollutant concentrations for the High income modeling subregion were obtained from two monitors: the *Las Condes* and *Providencia* MACAM-2 monitoring stations. Medium income modeling subregion used pollution concentration data obtained from three monitors: *La Florida*, *La Paz*, and the *Parque O'Higgins* monitors. Low income modeling subregion used pollution concentrations from three monitoring stations: *Pudahuel*, *Cerrillos* and *El Bosque*. Because of their effects on human health, dry deposition of the five criteria air pollutants in Santiago (CONAMA, 1997) were quantified: particulate matter less than 10 microns ( $\text{PM}_{10}$ ), sulfur dioxide ( $\text{SO}_2$ ), carbon monoxide (CO), nitrogen dioxide ( $\text{NO}_2$ ), and ozone ( $\text{O}_3$ ). The MACAM-2 network measured  $\text{NO}_2$ ,  $\text{O}_3$ , and  $\text{SO}_2$  in units of parts per billion (ppb), while CO was measured in parts per million (ppm). Pollutants measured in ppb were converted to ppm and then to  $\mu\text{g m}^{-3}$  based on measured atmospheric temperature and pressure (Seinfeld, 1986). Hourly  $\text{PM}_{10}$  concentrations ( $\mu\text{g m}^{-3}$ ) were also obtained from the MACAM-2 monitors. Missing hourly meteorological or pollution-concentration data were estimated using the monthly average for the specific hour. In some locations, an entire month of pollution-concentration data might have been missing and are estimated based on interpolations from existing data.

### 2.7. Air pollution removal and improvement

Dry deposition for the periods of July 1997 to June 1998 and July 2000 to June 2001 were estimated using measured urban forest cover and estimated leaf area data from Santiago's three socioeconomic subregions, hourly data from the *La Platina* weather station, and hourly pollutant concentration data stratified by socioeconomic subregion. Downward pollutant flux, or removal ( $F$ ;

$\text{g m}^{-2} \text{s}^{-1}$ ) was calculated by the UFORE model as the product of dry deposition velocity ( $V_d$ ;  $\text{m s}^{-1}$ ) and hourly pollutant concentration ( $C$ ;  $\text{g m}^{-3}$ ) ( $F = V_d C$ ). Deposition velocities of the gaseous pollutants for the in-leaf season are estimated by the model using a series of resistance formulas (Pederson et al., 1995) where  $V_d$  is the reciprocal of the sum of the aerodynamic ( $R_a$ ), quasi-laminar boundary layer ( $R_b$ ) and canopy resistances ( $R_c$ ) (Baldocchi et al., 1987). Hourly estimates of  $R_a$  and  $R_b$  were calculated using hourly weather data from the *La Platina* weather station.  $R_a$  and  $R_b$  effects were relatively small compared to  $R_c$  effects.

Canopy resistances for  $\text{O}_3$ ,  $\text{SO}_2$ , and  $\text{NO}_2$  are based on the big-leaf and multi-layer canopy deposition models (Baldocchi et al., 1987; Baldocchi, 1988) and have three components: stomatal resistance ( $r_s$ ), mesophyll resistance ( $r_m$ ), and cuticular resistance ( $r_t$ ), such that:  $1/R_c = 1/(r_s + r_m) + 1/r_t$ . Mesophyll resistance was set to zero  $\text{m s}^{-1}$  for  $\text{SO}_2$  and  $10 \text{ m s}^{-1}$  for  $\text{O}_3$ . Mesophyll resistance was set to  $100 \text{ m s}^{-1}$  for  $\text{NO}_2$  to account for the difference between transport of water and  $\text{NO}_2$  in the leaf interior, and to bring the computed deposition velocities in the range typically exhibited for  $\text{NO}_2$  (Lovett, 1994). Base cuticular resistances were set at  $8000 \text{ m s}^{-1}$  for  $\text{SO}_2$ ,  $10,000 \text{ m s}^{-1}$  for  $\text{O}_3$ , and  $20,000 \text{ m s}^{-1}$  for  $\text{NO}_2$  to account for the typical variation in  $r_t$  exhibited among the pollutants (Lovett, 1994). Since CO and  $\text{PM}_{10}$  do not directly depend on transpiration and photosynthesis, canopy resistance for CO was set to a constant for in-leaf season ( $50,000 \text{ m s}^{-1}$ ) and leaf-off season ( $1,000,000 \text{ m s}^{-1}$ ) (Bidwell and Fraser, 1972).  $\text{PM}_{10}$  was set to a median deposition velocity (Lovett, 1994) of  $0.064 \text{ m s}^{-1}$  based on a LAI of 6 and a 50% resuspension rate of particles back to the atmosphere (Zinke, 1967) and then adjusted to actual LAI. Hourly deposition values for all pollutants were set to 0 during periods of measured precipitation, fog or mist. Annual pollutant flux or abatement was multiplied by urban forest cover ( $\text{m}^2$ ) to estimate annual pollutant removal, in  $\text{g m}^{-2}$  and metric tons.

The relative effect of urban forest cover in reducing monthly pollutant concentrations in the atmosphere, or air quality improvement ( $E$ ; %) were estimated ( $E = R/(R+A)$ ) using the amount of pollution in the atmosphere ( $A$ ; kg), which was then contrasted with the amount of average monthly dry deposition as estimated by the model ( $R$ ; kg) (Nowak et al., 2006). The mean hourly mixing heights ( $m$ ) per month as measured by the wind profiler and RASS of the CENMA were used along with mean hourly pollution concentrations per month to calculate the amount of pollutants within the mixing layer ( $\mu\text{g m}^{-2}$ ), assuming a well mixed ground-boundary layer. Mixing heights provided by CENMA were obtained from methods using Seibert et al. (2000) (Personal communication Pablo Ulricksen CENMA). Air quality improvement assumes a continuous urban forest cover of 100%.  $E$  is therefore adjusted to the actual urban tree and shrub cover for each subregion and for the city.

## 3. Results

### 3.1. Urban demographics and urban forest structure

Urban demographics in Santiago differed by socioeconomic subregions within the city as exhibited in Table 1. Santiago's high income subregion occupied a proportionally larger area than many other high income sectors in major Latin American cities (Browder et al., 1995; Scarpaci et al., 1988). The middle income subregion and the Cerro San Cristobal, a high prominent feature running along the northwest axis of the city, separate Santiago's high income subregion from the low income subregion. The high income subregion had a greater leaf area than the low and medium income subregions (Escobedo et al., 2006). However, the low and medium income strata encompassed nearly 80% of the study area (Table 2). LAI and

**Table 2**  
Mean estimates of Santiago's urban tree and shrub cover characteristics by socioeconomic subregion.

| Subregion | N plots | D <sup>a</sup> (trees/ha) | % cover tree <sup>a</sup> (shrub) | % leaf area evergreen composition <sup>a</sup> tree (shrub) | LAI tree <sup>a,b</sup> (shrub) | Tree leaf area <sup>a</sup> km <sup>2</sup> (SE) | Shrub leaf area <sup>a</sup> km <sup>2</sup> (SE) |
|-----------|---------|---------------------------|-----------------------------------|---|---------------------------------|--|---|
| High      | 74      | 100.8                     | 25.9 (15.9)                       | 35 (72)   | 4.1 (4.3)                       | 175 (50)   | 113 (15)  |
| Medium    | 62      | 58.2                      | 12.2 (9.8)                        | 24 (53)   | 2.6 (1.3)                       | 119 (32)   | 47 (20)   |
| Low       | 64      | 55.6                      | 13.7 (8.0)                        | 35 (42)   | 2.5 (1.2)                       | 149 (36)   | 43 (20)   |
| Region    | 200     | 64.3                      | 15.1 (10.0)                       | 32 (61)   | 3.0 (2.0)                       | 443 (65)   | 200 (30)  |

N, number of plots; D, tree density; SE, standard error.

<sup>a</sup> Source: Escobedo et al. (2006, 2008).

<sup>b</sup> Leaf area index is per tree and shrub canopy and is a function of cover and leaf area.

overall tree and shrub cover per subregion were greatest in the high income area. Santiago's low tree and shrub LAI values with respect to those reported in Nowak et al. (2006) might reflect tree cover and conditions characteristic of a semi arid climate. Additional urban forest structure results for these subregions are presented in Escobedo et al. (2006).

### 3.2. Air pollution concentrations

The low income sector had the highest PM<sub>10</sub> concentrations in the city (Table 3) and a low tree density (Table 2). This corresponds with other findings that indicate a relationship among Santiago's socioeconomics and amount of bare soils and unpaved roads with ensuing PM<sub>10</sub> emissions (Bertrand and Romero, 1993; CONAMA, 1997). Further, the northwestern part of the study area is also composed of pumice-derived soils that can become easily suspended. This combination of unpaved roads, pumice soils, and the lack of vegetation cover might be contributing to increased PM<sub>10</sub> concentrations in this subregion (Romero et al., 1999). Nitrogen dioxide had the lowest concentration in the study area probably due to lower vehicle ownership and traffic associated with agricultural and residential areas (World Bank, 1997).

The medium income sector had the highest O<sub>3</sub> and CO concentrations in the study area (Table 3). CO levels might be attributed to the increased vehicle traffic running through the city center (CONAMA, 1997). NO<sub>2</sub> concentration measurements were not available for this sector, but the same factors creating high CO levels might create high NO<sub>2</sub> concentrations. Additionally increased vehicle traffic will result increase NO<sub>2</sub> concentrations. This and wind direction might be subsequently reducing O<sub>3</sub> concentration in the High income subregion as a result of ozone scavenging (Jacob and Wofsy, 1988; World Bank, 1997). The high ambient O<sub>3</sub> concentrations are likely a result of transport by prevailing local winds from the western areas of Santiago towards the central and eastern piedmont areas (Romero et al., 1999). Monitored SO<sub>2</sub> concentrations were slightly higher in this sector and might possibly be related to the greater density of industrial and commercial areas in this subregion.

The high income subregion had a high mean annual income, greater vegetation cover, and an increased ambient air pollution

**Table 3**  
Mean annual air ambient pollution concentrations from 8 MACAM-2 monitors.

| Modeling domain       | PM10 | O3   | CO     | NO2  | SO2  |
|-----------------------|------|------|--------|------|------|
| Subregion (2000–2001) |      |      |        |      |      |
| High                  | 59.1 | 29.9 | 1118.9 | 49.4 | 10.5 |
| Medium                | 78.9 | 34.8 | 1327.5 | n/a  | 13.5 |
| Low                   | 84.4 | 29.8 | 1150.6 | 30.7 | 12.9 |
| Region (2000–2001)    | 74.1 | 31.5 | 1199.0 | 40.1 | 12.3 |
| Region (1997–1998)    | 94.7 | 30.2 | 1512.7 | 43.2 | 18.1 |

**Table 4**  
Annual pollution removal totals in Santiago's three subregions from July 2000 to June 2001. Ranges (metric tons) report high and low deposition velocities from the literature; no range estimated for CO (Nowak et al., 2002).

|                  | Subregion (metric tons) |                |                |
|------------------|-------------------------|----------------|----------------|
|                  | Low                     | Medium         | High           |
| PM <sub>10</sub> | 672 (263–1050)          | 538 (210–840)  | 544 (213–850)  |
| O <sub>3</sub>   | 197 (70–467)            | 203 (72–483)   | 228 (58–394)   |
| CO               | 34                      | 33             | 27             |
| NO <sub>2</sub>  | 63 (48–141)             | Not measured   | 116 (65–190)   |
| SO <sub>2</sub>  | 50 (38–147)             | 46 (35–139)    | 44 (26–98)     |
| Total            | 1015 (418–1805)         | 820 (317–1462) | 959 (361–1532) |

concentration of NO<sub>2</sub> (Table 3). A possible explanation for the increased NO<sub>2</sub> concentration in this sector is increased vehicle ownership and longer commuting distances (Romero et al., 1999; World Bank, 1997). CO concentrations were the lowest at this subregion possibly due to the influence of the *Las Condes* monitor which measured a very low mean annual concentration of 839 μg as opposed to 1398 μg measured at the nearby *Providencia* monitor. Wind patterns and enhanced vegetation cover may be contributing to lower PM<sub>10</sub> concentrations in this subregion as well.

### 3.3. Pollution removal totals by the Metropolitan Region's urban forest

Pollution removal, excluding NO<sub>2</sub> which was not measured in the medium income subregions, in the modeling region by trees and shrubs was 2,790 metric tons during July 2000 to June 2001 (Table 4). NO<sub>2</sub> was not monitored in the medium and high income subregion during the July 1997 to June 1998 season. However, assuming similar NO<sub>2</sub> removal rates per area of tree and shrub cover for the low and medium subregion, 3,500 metric tons were removed by Santiago's urban forest during the 2000–2001 analysis period. Approximately 60% of the total pollution removed was by trees during this analysis period. The sum of the individual pollution removal rates for the 3 socioeconomic subregions during 2000–2001, yields a weighted (by cover) annual removal rate of 12.5 and 12.3 g m<sup>-2</sup> of tree and shrub cover, respectively, across the city (Table 5).

**Table 5**

Annual pollution removal rates for Santiago's three subregions from July 2000 to June 2001. Ranges ( $\text{g m}^{-2}$ ) report high and low deposition velocities from the literature; no range estimated for CO (Nowak et al., 2002).

|                  | Subregion ( $\text{g m}^{-2}$ ) |                |                 |                |                 |                 |
|------------------|---------------------------------|----------------|-----------------|----------------|-----------------|-----------------|
|                  | Low                             |                | Medium          |                | High            |                 |
|                  | Trees                           | Shrubs         | Trees           | Shrubs         | Trees           | Shrubs          |
| PM <sub>10</sub> |                                 | 5.8 (2.3–9.1)  | 7.4 (2.9–11.5)  | 5.7 (2.2–8.8)  | 7.5 (2.9–11.7)  | 8.5 (3.3–13.3)  |
| O <sub>3</sub>   | 2.4 (0.8–5.0)                   | 1.7 (0.7–5.0)  | 2.8 (0.9–5.9)   | 2 (0.9–5.9)    | 3.3 (0.8–5.7)   | 3.3 (0.9–5.7)   |
| CO               | 0.4                             | 0.4            | 0.4             | 0.4            | 0.4             | 0.4             |
| NO <sub>2</sub>  | 0.7 (0.5–1.5)                   | 0.6 (0.5–1.5)  | NA              | NA             | 1.7 (0.9–2.7)   | 1.7 (1.0–2.8)   |
| SO <sub>2</sub>  | 0.6 (0.4–1.6)                   | 0.4 (0.4–1.6)  | 0.6 (0.4–1.7)   | 0.5 (0.4–1.7)  | 0.6 (0.4–1.4)   | 0.7 (0.4–1.4)   |
| Total            | 12.0 (4.8–20.5)                 | 8.8 (3.9–17.1) | 11.9 (4.8–20.5) | 9.2 (3.5–16.4) | 13.4 (5.0–21.5) | 14.6 (5.5–23.2) |

NA, not measured.

During 1997–1998 tree cover removed  $15.3 \text{ g m}^{-2}$  and shrubs removed  $13.6 \text{ g m}^{-2}$  of pollutants across the region. PM<sub>10</sub> pollution removal rate was the greatest in all subregions for both analysis periods.

#### 3.4. Tree and shrub pollutant removal rates by subregion

PM<sub>10</sub> removal per square meter of tree cover was greatest for all pollutants in all subregions in the low socioeconomic subregion (Table 5). The low overall vegetative cover in this subregion might be contributing to higher PM<sub>10</sub> suspension rates, hence greater PM<sub>10</sub> concentrations. These two factors, low vegetative cover (leaf area and tree density) and high rates of pollution removal might indicate a potential for increased trees and shrubs to reduce PM<sub>10</sub> emissions and concentrations in this area. The lower pollutant removal ability of O<sub>3</sub>, NO<sub>2</sub> and SO<sub>2</sub> might be a result of lower LAI and lower ambient pollution concentrations that could possibly affect tree transpiration rates and dry deposition of gases. Differences in meteorological conditions among subregions were not quantified since only 1 weather station was used.

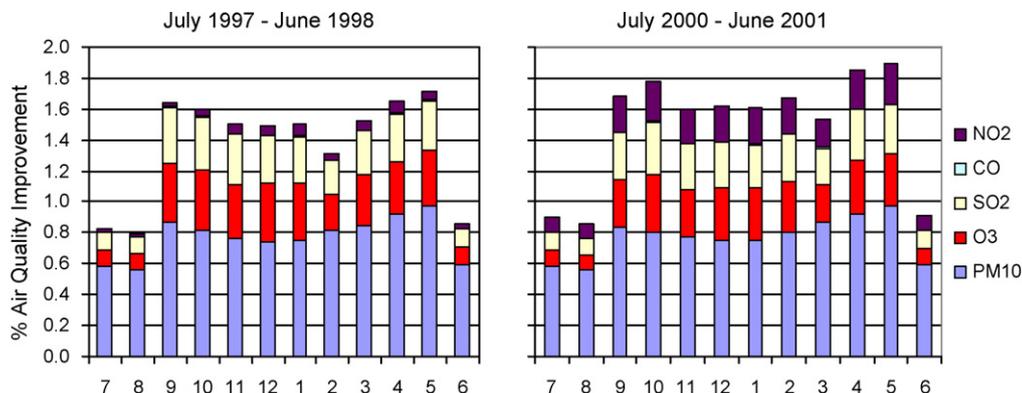
The medium income subregion had the lowest total pollution removal amounts and PM<sub>10</sub> removal rates per area of tree and shrub cover (Table 5). The relatively low pollution removal in this subregion is most likely due to the lowest overall tree and shrub cover and the decreased ambient concentrations for some pollutants. PM<sub>10</sub> removal rate per square meter of shrub cover in the high income subregion was the greatest for any type of vegetation cover (Table 5). Total ozone pollution removal and tree and shrub removal rates per square meter were also the greatest for any strata, as was total NO<sub>2</sub> removal. The relatively high O<sub>3</sub> removal rate may be attributed to this subregion's high LAI and tree cover. An impor-

tant additional factor is that this subregion had the highest percent tree and shrub evergreen leaf area that provides for year round removal of pollutants.

#### 3.5. Percent monthly air quality improvement

Monthly air quality improvement in the modeling region was quantified from July 2000 through June 2001 and July 1997 through June 1998. Average monthly percent air quality improvement due to trees and shrubs is estimated at 1.5 percent for 2000 to 2001 (Fig. 2). Percent monthly improvement during this same analysis period for the month of May was 1.9% and approximately 0.9% from June to August. Average monthly air quality improvement was 1.4% from July 1997 to June 1998. Despite 1997 being an El Niño Southern Oscillation year, percent annual air quality improvement during 1997–1998 was similar to 2000–2001.

Seasonal air quality improvement differed among the subregions. During the 2000–2001 modeling period, the high income subregion had the highest seasonal air quality improvement by trees and shrubs at 2.0%. Annual air quality improvement trends between the medium and low income subregion for the analysis period were different (Table 6). Peak hourly improvements of 10.3%, 7.9%, and 10.4% for PM<sub>10</sub> were estimated for the high, medium and low income subregions, respectively, in areas of 100% tree cover. Inter-annual variations in air quality improvement by urban forests will occur because of meteorological and pollutant concentration conditions. These two conditions might explain the slight seasonal fluctuations observed in the air quality improvement trend. Finally, results are likely due to the relatively low mixing heights present in Santiago when using the percent air quality improvement method used by the UFORE model (Table 6).



**Fig. 2.** Percent mean monthly air quality improvement from trees and shrubs in Santiago, Chile during July 1997 to June 1998 and July 2000 to June 2001. Horizontal axes denote the 12 months of the southern latitude year, 7 represents the month of July, 8 the month of August and so forth. Note that 1997 was an El Niño year and might explain air quality improvement fluctuations during this analysis period.

**Table 6**  
Annual air quality improvement by pollutant for Santiago's three different subregions during 2000–2001.

| Subregion | Tree             |  | Shrub            |   | Assuming 100% tree cover over a subregion   |
|-----------|------------------|--|------------------|---|---|
|           | Measured % cover | Average % air quality improvement by pollutant   | Measured % cover | Average % air quality improvement by pollutant  | Average % air quality improvement by pollutant  |
| High      | 26%              | PM <sub>10</sub> = 1.6<br>O <sub>3</sub> = 0.5<br>SO <sub>2</sub> = 0.5<br>CO = 0.006<br>NO <sub>2</sub> = 0.4 | 16%              | PM <sub>10</sub> = 1.1<br>O <sub>3</sub> = 0.3<br>SO <sub>2</sub> = 0.3<br>CO = 0.004<br>NO <sub>2</sub> = 0.3  | PM <sub>10</sub> = 6.1<br>O <sub>3</sub> = 2.1<br>SO <sub>2</sub> = 1.9<br>CO = 0.02<br>NO <sub>2</sub> = 1.5 |
| Medium    | 12%              | PM <sub>10</sub> = 0.6<br>O <sub>3</sub> = 0.4<br>SO <sub>2</sub> = 0.1<br>CO = 0.003<br>NO <sub>2</sub> = NA  | 10%              | PM <sub>10</sub> = 0.4<br>O <sub>3</sub> = 0.2<br>SO <sub>2</sub> = 0.1<br>CO = 0.003<br>NO <sub>2</sub> = NA   | PM <sub>10</sub> = 5.4<br>O <sub>3</sub> = 2.8<br>SO <sub>2</sub> = 1.3<br>CO = 0.03<br>NO <sub>2</sub> = NA  |
| Low       | 14%              | PM <sub>10</sub> = 0.7<br>O <sub>3</sub> = 0.2<br>SO <sub>2</sub> = 0.2<br>CO = 0.003<br>NO <sub>2</sub> = 0.2 | 8%               | PM <sub>10</sub> = 0.3<br>O <sub>3</sub> = 0.1<br>SO <sub>2</sub> = 0.1<br>CO = 0.001<br>NO <sub>2</sub> = 0.08 | PM <sub>10</sub> = 4.7<br>O <sub>3</sub> = 1.8<br>SO <sub>2</sub> = 1.6<br>CO = 0.02<br>NO <sub>2</sub> = 1.2 |

NA, not measured.

#### 4. Discussion

Urban forest structure, pollution emission dynamics, and consequent air pollution removal by urban forest cover reflected the existing vegetation and environmental differences among socio-economic subregions in Santiago. Although the gross domestic product in Chile doubled during the 1990s, income inequality has remained the same as in the 1960s (Castañeda, 1999). During the Pinochet regime, lower income neighborhoods and squatters were relocated to the urban fringes of what is now the low income subregion and to a certain extent, the medium income subregion (Scarpaci et al., 1988). Although there has been an increase in average income for many areas in the southwestern-most sectors of the city, low income areas presently occupy the northwestern-most areas, southwestern-central and southeastern subregions of the city. Other urban forest structure analyses reveal the role socio-economics can play in influencing urban forest structure (Escobedo et al., 2006; Heynen and Lindsey, 2003).

The pollutant flux equation ( $F = V_d \times C$ ) illustrates an almost linear relationship between pollution removal, ambient pollution concentrations, and urban forest cover in Santiago. In general, greater pollution concentrations, tree cover, proportion of evergreen tree cover, and LAI will result in more pollutants removed. This relationship is the basis of most urban forest effects on air quality studies (Smith, 1990; McPherson et al., 1998, 1999; Nowak and Crane, 2000; Nowak et al., 2002; Scott et al., 1998; Yang et al., 2005). Although this linear relationship will lead to a point in which very high pollutant concentrations can lead to stomatal closure resulting in cuticular uptake exceeding stomatal uptake, this has as yet not been factored into the UFORE model. These same conditions would however indicate other more severe effects to organisms other than plants (i.e., human populace). Even though pollution removal in the low and medium socioeconomic sectors was only slightly different (Table 5), results indicate the role heterogeneous urban morphology and pollution have across an urban forest landscape. Santiago's meteorological conditions, low mixing heights, and vegetation characteristics also play a role in defining urban forest pollutant removal ability. Using this approach and results from this evaluation, managers could determine the pollution removal effects of increasing urban forest cover in any or all of Santiago's subregions. Likewise, the approach might be applicable to a finer scale of administrative units or other ecological units such as forest patches.

Differences in pollutant removal and air quality improvement between 1997–1998 and 2000–2001 (Fig. 2) can be explained by changes in pollution dynamics within Santiago. For example MACAM monitoring stations exhibited a decrease of 29% in PM<sub>10</sub> concentrations from January 1989 to January 2000, an average reduction of 0.48  $\mu\text{g}/\text{m}^3/\text{year}$  (CONAMA, 1997). Emission concentrations were not monitored in Santiago prior to 1989. Conversion of many industries from wood burning to natural gas usage might have contributed to this reduction in pollutant concentrations (CONAMA, 1997).

Percent air quality improvement by an urban forest is mainly a component of mixing layer height and  $V_d$ . For Santiago, Chile, average percent air quality improvement did not vary between analysis years despite different annual mixing heights and pollution fluxes. Santiago's recurring thermal inversions, stable meteorological conditions during the periods of analysis, and high particulate matter concentrations have likely resulted in Santiago's urban forests having a very high and consistent percent PM<sub>10</sub> removal rate. Further, Santiago had higher removal rates than many other cities previously analyzed (Nowak et al., 2006). Actual field measurements of urban forest structure versus assumed parameters can also affect pollution removal estimates. For example, the tree PM<sub>10</sub> pollution removal rate of Santiago's low income sector is equal to Los Angeles, which has the greatest pollution removal rate per m<sup>2</sup> of tree cover of any city in the United States (Nowak et al., 2006). Los Angeles and Santiago have similar climate and urban characteristics. Los Angeles results, on the other hand, were not based on actual field measurements but on modeling assuming an tree canopy LAI of 6, which is considerably greater than Santiago's measured tree canopy LAI of 3 (Table 2).

Despite a decrease in ambient concentrations of most pollutants in Santiago during the years of analysis, low mixing heights might also have contributed to Santiago and its three subregion's consistent air quality improvement capacity by its urban forests. Note that data used were from one weather and mixing height measurement station. During the 2000–2001 modeling period, mean annual mixing height in Santiago was 260 m while New York City during 1994 had a mean annual mixing height of 866 m. Additional factors possibly contributing to the Santiago's air pollutant removal capacity were a longer growing season, greater percent evergreen urban forest composition, and most importantly a lower boundary layer (Table 2). These are the same factors used to estimate air pollution removal and percent air quality improvement by the UFORE

model. Also the use of subregion-specific or local-scale meteorological data would likely change the  $V_d$  and air quality improvement for all pollutants, except CO and PM<sub>10</sub> which do not depend on transpiration.

Results from the UFORE model indicate the influence of spatial heterogeneity of tree-shrub cover, LAI, percent evergreen composition, and scale on urban forest function. Therefore, results from this study seem to indicate a need to incorporate local-scale urban forest structure. At this finer scale (as compared to a city-wide scale), analysis might be suitable for evaluating municipal, district, or neighborhood-wide tree planting programs, their configurations, and their effect on local-scale air quality. A separate analysis using ecological attributes such as forest patches or spatial heterogeneity across several scales, as opposed to socioeconomic classes, might also lead to different pollution removal amounts and rates (Gaucherel, 2007; Zipperer et al., 1997). Accordingly, the distribution of the cover within an area as well as its configuration with respect to weather patterns may yield insights into the effects of urban vegetation on pollution removal rates. This might prove to be valuable in analyzing policies that promote maintaining or increasing tree cover for local-scale air quality improvement around schools, hospitals, residential areas and other strategic areas of public health concern (Powe and Willis, 2004).

Unfortunately, because of the lack of weather data and model limitations, this study could not examine these local-scale effects as well as the effects of microclimate, species configuration and compositions, deposition to wet surfaces, and patch type on pollution removal. However, care needs to be taken when analyzing the micro-scale effects of vegetation on air quality as interaction at this fine scale often do not factor the aggregated effects of vegetation on meteorology and complex atmospheric chemistry and physics processes (Grimmond et al., 2002; Nowak et al., 1998a; Seinfeld, 1986). None the less, additional analyses at the micro-scale are needed to evaluate the overall function and effect of vegetation at the local and meso-scale on pollution and climate processes (McPherson et al., 1998).

## 5. Conclusion

Urban ecosystems are a complex mosaic of climates, land uses, biophysical, and socioeconomic variables. Future studies of urban forests and their role in environmental quality should consider the ecological and socio-economic heterogeneity within the urban ecosystem. Analyses that use a broad meso-scale modeling region approach, often fail to capture the variability of ecosystem functions and their relationship to a diverse human population and urban forest structure. Analysis at different scales could be used for promoting and quantifying the need for increasing vegetation around critical areas and conserving peri-urban forested areas from urbanization and for increasing or maintaining tree cover in urban parks, woodlots and other treed patches (Zipperer et al., 1997).

Santiago's urban forests were most effective at removing PM<sub>10</sub> than any other pollutant across all subregions. This was particularly true for the low income subregion. Amounts of urban forest (i.e., tree and shrub) cover, proportion of evergreen leaf area, LAI, and very high ambient pollution concentrations were the main factors in Santiago's urban forests having an exceedingly high PM<sub>10</sub> pollution removal rate. Coupling these factors with low mixing heights, Santiago had a higher air quality improvement capacity relative to other cities (Nowak et al., 2006). Based on results from this study, the analysis of urban forest structure by socioeconomic characteristics and subregion scales improved our understanding of urban forest function.

Policy applications from this study's results can be used to evaluate the use of different management alternatives and urban forest structure configurations for environmental quality improvement at different scales. Urban forest structure goals (e.g., tree, shrub, and herbaceous cover) can also be integrated into local and regional policies to improve air quality by subregion (Powe and Willis, 2004). Results suggest that finer scale analysis of urban forests enables a better characterization of cover, LAI, and percent evergreen tree composition and how the urban forest can remove air pollutants as opposed to a coarser-scale analysis of the urban forests. Further, quantifying the spatiotemporal distribution across different scales can assist in evaluating policies that maintain or increase vegetation cover in critical areas within urban and peri-urban forests. For example, and depending on management objectives, fine scale patches of urban forests with high LAI and percent evergreen tree composition in highly polluted areas can potentially remove air pollutants as effectively as coarser-scale urban forests with low tree cover, LAI, and high deciduous species composition.

Results from this study warrant further research on the parameterization of species-specific  $V_d$  and micro-scale meteorological data into functional models and the role of location of vegetation on pollution flux and subsequent urban environmental quality and human well being. Research on the role of these factors on air quality in key micro-scale areas of public health concern is sorely needed (Powe and Willis, 2004). Analyzing these factors and their role in urban forest function across micro and local-scales will help to highlight the multi-functionality of the urban forest at the meso-scale. This approach will also improve functionality of current urban forest function models when making city-wide assessments and inter-city comparisons of urban forest functions.

Ecosystem service studies have demonstrated that urban forests directly and indirectly influence ecological processes, environmental quality, and human well-being in cities (Beckett et al., 1998, 2000; McPherson et al., 1998, 1999; Nowak et al., 2002; Powe and Willis, 2004; Ulrich, 1986). Environmentally, urban forests reduce storm water runoff and minimize soil erosion (McPherson et al., 1999). Economically, urban forests can be cost-effective at reducing particulate matter (Escobedo et al., 2008) and they also reduce building energy use, increase real estate values, and reduce recovery time in hospitals (McPherson et al., 1999; Nowak and Crane, 2000; Ulrich, 1986). However, the urban forest might also have detrimental effects on air quality including allergenic effects of pollen and the emissions of volatile organic compounds (VOCs), which can eventually form ozone. Other negative effects include city finances through tree maintenance, removal of litter and debris; tree-related damage to infrastructure, and debris removal after catastrophic storms (Chameides et al., 1988; McPherson et al., 1999; Sharkey and Singsaas, 1995). Policy options that take into consideration the full functionality of urban forests as well as their spatiotemporal heterogeneity, scale, proper species-selection, maintenance, water use, VOC emission rates, allergenic effects, and spatial arrangement, and quantity and quality will maximize the ambient air quality and human well-being of the urban populace.

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