



ESTIMATING COST EFFECTIVENESS OF RESIDENTIAL YARD TREES FOR IMPROVING AIR QUALITY IN SACRAMENTO, CALIFORNIA, USING EXISTING MODELS

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Abstract—The Sacramento Municipal Utility District's (SMUD) shade tree program will result in the planting of 500,000 trees and has been found to produce net benefits from air conditioning savings. In this study we assume three scenarios (base, highest, and lowest benefits) based on the SMUD program and apply Best Available Control Technology (BACT) cost analysis to determine if shade trees planted in residential yards can be a cost effective means to improve air quality. Planting and maintenance costs, pollutant deposition, and biogenic hydrocarbon emissions are estimated annually for 30 years with existing deterministic models. For the base case, the average annual dollar benefit of pollutant uptake was \$895 and the cost of biogenic hydrocarbon emissions was \$512, for a net pollutant uptake benefit of \$383 per 100 trees planted. The uniform annual payment necessary to repay planting and maintenance costs with a 10% rate of interest was \$749. When high biogenic hydrocarbon emitting tree species were replaced with low-emitters, the base case benefit-cost ratio (BCR) increased from 0.5:1 to 0.9:1. The BCR for the "highest" and "lowest" benefit cases were 2.2:1 and –0.8:1, respectively. Although SMUD plantings produce cost effective energy savings, our application of the BACT analysis does not suggest convincing evidence that there is cost savings when only air quality benefits are considered. Published by Elsevier Science Ltd.

Key word index: Air pollution mitigation, urban forestry, pollutant deposition, hydrocarbon emission, natural resource valuation.

1. INTRODUCTION

Trees absorb gaseous pollutants through leaf stomata and can bind or dissolve water soluble pollutants onto moist leaf surfaces. Tree canopies also intercept particulates and reduce local air temperatures. Although there is a large literature on the deposition of air pollutants to vegetation (Smith, 1981; Davidson and Wu, 1990), there are few studies on the effects of urban forests on air quality. DeSanto *et al.* (1976a, b) estimated pollutant uptake by street trees in the St. Louis, MO area. Using algorithms from the Urban Airshed Model (UAM), Nowak (1994) estimated air pollutant removal by trees in the Chicago region during 1991 as 5575 t with an implied value of \$9.2 million.

Urban trees may reduce ambient air ozone concentrations, either by direct absorption of ozone or other pollutants such as NO₂, or by reducing air temperatures, which reduces hydrocarbon emission and ozone formation rates (Cardelino and Chameides, 1990). Sailor (1995) found diminished development of the summertime urban heat island when surface albedo and urban vegetation were increased in a meso-scale climate model of the Los Angeles basin. On the other hand, biogenic hydrocarbon emissions from

trees can play a role in ozone formation (Winer *et al.*, 1983; Chameides *et al.*, 1988).

The role of trees in air quality has become coupled with concern over the costs and benefits of large-scale urban tree planting programs (Corchnoy *et al.*, 1992). For example, Sacramento Shade, a partnership between the Sacramento Municipal Utility District (SMUD) and the non-profit Sacramento Tree Foundation, annually plants approximately 50,000 trees near homes to reduce air conditioning demand. It is one of SMUD's most cost effective energy efficiency programs (Weedall, 1995). However, Sacramento Shade's 10 year goal of planting 500,000 new trees has air quality managers concerned about possible impacts of biogenic hydrocarbon emissions on ozone levels in the country's fifth smoggiest region.

Trees planted to conserve energy in Sacramento have secondary benefits. They indirectly influence air quality by reducing CO₂, SO₂ and NO_x emissions from power plants. Trees also remove atmospheric CO₂ and store it as woody biomass. The value of different benefits and costs were reported for a hypothetical planting of 25,000 trees in yards by Chicago residents (McPherson, 1994). The projected net present value was \$585 per tree planted (3.5:1 benefit-cost ratio) assuming a 30 yr analysis period and

7% discount rate. Most benefits were attributed to "other intangible benefits" (e.g. scenic quality, recreation and relaxation opportunities, wildlife habitat), while energy savings and uptake of CO₂ and other air pollutants by the trees accounted for 39 and 3% of the total present value of benefits, respectively.

Two approaches are being used to evaluate the feasibility of urban forestry for urban air quality improvement. In one approach, scientists in New York are using the UAM (Morris *et al.*, 1992) to determine if urban vegetation has an effect on predicted ozone concentrations in metropolitan areas of the Northeast (Boston, New York, Philadelphia, Baltimore). If modeling experiments indicate that canopy cover can decrease ozone concentrations near the ground, then urban forest management may eventually qualify as a directionally sound strategy in state implementation plans, provided other regulatory criteria are met. Directionally sound strategies are defined as having positive effects on air quality, although their effects may be difficult to measure or verify.

This paper describes a second approach that is concerned with the economic feasibility of urban forestry as a local air quality improvement measure. It recognizes that local air quality management districts in California enforce regulations to promote use of the best available pollution control technologies when industries install new equipment. Best Available Control Technology (BACT) manuals are consulted to determine if new controls or processes are technically feasible and cost effective. Typically BACT analysis is applied to stationary sources, but we apply it here to determine if a large-scale urban tree planting like that in Sacramento can be a cost effective means to improve air quality. Both the UAM modeling and BACT analysis estimate pollutant deposition and biogenic hydrocarbon emissions. Our approach estimates annual pollutant deposition and biogenic hydrocarbon emissions over a 30 yr period, while the UAM predicts hourly air pollutant concentrations (primarily ozone) over the course of several days. Because our interest is to determine the return on investment as trees grow and die, our approach is necessarily long term. Our estimates of pollutant uptake and biogenic hydrocarbon emissions are first order approximations since they lack the precision associated with photochemical modeling. We recognize that cost effectiveness analysis may not be used in a regulatory environment, particularly when the net effects of urban trees on ozone are currently unknown, and describe the limitations of our approach and findings later in this paper.

2. METHODS

For this analysis we assume a hypothetical planting of 100 trees in a typical residential neighborhood in Sacramento, California. Sacramento Shade's average tree planting density of 4 trees for every third property

is applied. Hence, the neighborhood consists of 75 total residential lots, with 25 lots each receiving 4 trees. Assuming typical lot dimensions of 21 × 37 m (70 × 120 ft) and a 15 m (50 ft) street right-of-way, the total neighborhood area is 7 ha (17 acres). The deterministic models annually "grow" and "kill" trees; estimate air pollutant uptake by tree canopy projection area (CP) using algorithms for dry deposition; and project costs associated with their planting and care during a 30 yr period.

There are many sources of uncertainty associated with modeling pollutant uptake by trees, as well as biogenic hydrocarbon emissions (sometimes called biogenic volatile organic compounds, BVOCs) from trees. We calculate benefit-cost ratios for three scenarios that encompass some of the uncertainties associated with estimates of pollutant deposition and emission of biogenic hydrocarbons. The scenarios reflect the highest and lowest net air quality benefits that the trees are likely to produce.

2.1. Tree growth and mortality

The simulated planting consists of identical "typical" trees with characteristics representative of the tree species planted by Sacramento Shade from 1990 to 1993. To determine tree canopy projection area CP (area under a typical tree's canopy) 30 yr after planting, we calculated a weighted average CP from Sacramento Shade data based on the number of each tree species planted and its mature crown spread (Table 1). Therefore, this weighted average CP reflects the numbers of each species planted by Sacramento Shade and their mature sizes. We assumed initial growth of the "typical" tree to match data provided by Sacramento Shade on tree growth measured 3 yr after planting for 20 species (R. Tretheway, pers. comm., November 1995).

Mortality rates have been reported for street trees, but not residential yard trees (Dawson and Khawaja, 1985; Miller and Miller, 1991). We assumed that 100 trees were planted, 21 died during the first five year establishment period, and one tree died every other year thereafter. Thus, 67 trees survived 30 yr after planting.

2.2. Deposition

The hourly pollutant dry deposition to urban trees is expressed as the product of a deposition velocity $V_d = 1/(R_a + R_b + R_c)$, a pollutant concentration C , a canopy projection area CP, and a time step. Hourly deposition velocities for each pollutant were calculated using estimates for the resistances R_a , R_b , and R_c . R_a and R_b were estimated for each hour throughout a "base year" (1990) using formulations from the Urban Airshed Model (Killus *et al.*, 1984; Nowak, 1994). Hourly meteorological data for wind speed, solar radiation and precipitation from a California Department of Water Resources monitoring site located in Sacramento County, together with nighttime three-hourly cloud cover data from a local

Table 1. Species, numbers and dimensions of trees planted by Sacramento Shade from 1990–1993. Biogenic hydrocarbon emission factors ($\mu\text{g-C g}^{-1} \text{h}^{-1}$) at 30°C and PAR flux of 1000 $\mu\text{mol m}^{-2} \text{s}^{-1}$ (from Geron *et al.*, 1994, except where noted)

Common name Botanical name	Number planted	% Total planted	Mature spread (m)	Total CP (m^2)	% Total CP	Isoprene	Monoterpene
Maples							
<i>Acer spp.</i>	16,652	24.6	10.0 ^a	1,313,548	15.9	0.1	1.6
Oaks							
<i>Quercus spp.</i>	11,568	17.1	17.5 ^b	2,782,244	33.7	11.9 ^b	0.005 ^b
Chinese pistache							
<i>Pistacia chinensis</i>	7,182	10.6	12.2	838,438	10.2	0.0 ^c	12.5 ^c
Chinese hackberry							
<i>Celtis sinensis</i>	5,279	7.8	15.2	962,935	11.7	0.1	0.2
Tallow							
<i>Sapium sebiferum</i>	5,205	7.7	10.7	465,224	5.6	0.0	0.0
London plane							
<i>Platanus acerifolia</i>	3,970	5.9	12.2	463,464	5.6	35.0	0.1
Ginkgo							
<i>Ginkgo biloba</i>	3,319	4.9	12.2	387,465	4.7	0.0	3.0
Tupelo							
<i>Nyssa sylvatica</i>	2,560	3.8	7.0	98,810	1.2	14.0	0.6
Goldenrain tree							
<i>Koelreuteria paniculata</i>	2,141	3.2	10.7	191,363	2.3	49.0 ^d	0.0 ^d
Saucer magnolia							
<i>Magnolia soulangiana</i>	1,932	2.9	6.1	56,386	0.7	0.1	3.0
Hornbeam							
<i>Carpinus spp.</i>	1,887	2.8	10.2 ^e	156,638	1.9	0.1	1.6
American linden							
<i>Tilia americana</i>	1,670	2.5	16.8	368,593	4.5	0.0	0.0
European white birch							
<i>Betula pendula</i>	1,420	2.1	5.5	33,569	0.4	0.1	0.2
Other species	2,877	4.3	7.6	130,859	1.0	0–0.1	0–0.1
Totals	67,662	100.0		8,249,536	100.0	7.4 ^f	1.8 ^f

^aCalculated as a weighted average for the following *Acer* species: *rubrum*, *palmatum*, *buergerianum*, *ginnala*, *campestre*, *rubrum*, *platanoides*.

^bCalculated as a weighted average for the following *Quercus* species: *rubra*, *lobata*, *coccinea*, *douglasii* (Tanner *et al.*, 1992).

^cSurrogate: *P. vera* (Tanner *et al.*, 1992).

^dSurrogate: *Cupaniopsis anacardioides* (Corchnoy *et al.*, 1992).

^eCalculated as a weighted average for the following *Carpinus* species: *betulus*, *betulus fastigiata*.

^fCalculated as a weighted average.

airport were used as input data. The aerodynamic resistance R_a is expressed as

$$R_a = \frac{u(z)}{u_*^2}$$

where $u(z)$ is the wind speed (m s^{-1}) at height z (10 m) and u_* is the frictional velocity (m s^{-1}). The quasi-boundary layer resistance R_b is expressed as

$$R_b = B^{-1} u_*^{-1}$$

where $B^{-1} = 2.2 u_*^{-1/3}$. The frictional velocity u_* is expressed in terms of the dimensionless Ψ_m stability function for momentum

$$u_* = \frac{ku(z-d)}{\ln \left[\frac{(z-d)}{z_0} \right] - \Psi_m \left[\frac{(z-d)}{L} \right] + \Psi_m \left[\frac{z_0}{L} \right]}$$

where k is the von Karman's constant (0.4), d the displacement length (8 m) and z_0 the roughness length

(0.5 m) for "suburban" areas (Stull, 1989), and L the Monin–Obukhov length (m) (van Ulden and Holtslag, 1985). L was estimated by classifying hourly meteorological data using an automated table look-up stability classification scheme to estimate $1/L$ as a function of Pasquill classes and z_0 (Gifford, 1976; Golder, 1972). Nine stability classes were used.

When atmospheric conditions are unstable ($L < 0$), the function Ψ_m (van Ulden and Holtslag, 1985) is

$$L < 0: \Psi_m = 2 \ln \left[\frac{(1+X)}{2} \right] + \ln \left[\frac{(1+X^2)}{2} \right] - 2 \tan^{-1}(X) + \frac{\pi}{2}$$

where the dimensionless factor X is (Dyer and Bradley, 1985)

$$X = \left(1 - 28 \frac{z}{L} \right)^{0.25}$$

When conditions are stable ($L > 0$), the function Ψ_m (van Ulden and Holtstag, 1985) is

$$L > 0: \Psi_m = -17 \left[1 - \exp \left(-0.29 \frac{(z-d)}{L} \right) \right].$$

Canopy resistances (R_c) for individual pollutants were based on averages taken from literature values for both forest and individual trees (Nowak, 1994). Hourly R_c estimates were categorized into in-leaf season (15 March through 15 November, based on local foliation periods), daytime, and nighttime. Canopy resistance for particles (PM_{10}) was estimated based upon an average deposition velocity minus an average R_a and R_b for Sacramento. Average daytime and nighttime canopy resistance values ($s\ cm^{-1}$) were: NO_2 (3.01, 7.54), O_3 (1.74, 17.2), SO_2 (1.87, 9.54), PM_{10} (0.74, 0.74). These canopy resistance values reflect the leaf surface area of the forest stand or tree that was measured.

The hourly canopy resistances were combined with R_a and R_b to estimate hourly deposition velocities for each pollutant. Deposition velocities were set equal to zero during periods of precipitation. Fifty percent of the deposited particulate matter was assumed to be resuspended in the base value scenario. Hourly pollutant depositions were then summed to get monthly and annual pollutant depositions for each year.

2.3. Pollutant concentrations

Hourly concentrations for NO_2 , O_3 , SO_2 (ppm) and PM_{10} ($\mu g\ m^{-3}$) were obtained from the USEPA AIRS data base for 1990 for a monitoring station located in a residential neighborhood in Sacramento. The station monitors for air pollutant concentrations representative of areas of high population density, at spatial scales of up to 4 km (ARB, 1994).

Pollutant concentrations in Sacramento in 1990 were typical of concentrations for the period 1989–1992. For the in-leaf season, the O_3 concentration between 3:00 and 4:00 p.m. averaged 0.061 ppm, while higher average hourly concentrations occurred in July (0.079 ppm), August, and September (0.071 ppm) during the hour between 3:00 and 4:00 p.m. The lowest average hourly afternoon concentrations occurred in March (0.048 ppm) and November (0.045 ppm). In Sacramento County the 0.12 ppm National Ambient Air Quality Standard (NAAQS) for ozone was exceeded 15 times between June and September (ARB, 1991). The in-leaf season NO_2 concentrations averaged 0.017 ppm in the morning hours and 0.023 ppm in the afternoon. The average hourly concentrations were highest in October (0.030, 0.045 ppm) and November (0.029, 0.044 ppm), occurring between the hours 7:00–9:00 am and 5:00–7:00 pm, respectively. The in-leaf season average hourly SO_2 concentration was 0.001 ppm.

2.4. Biogenic hydrocarbon emissions

Biogenic hydrocarbon emissions (sometimes called biogenic volatile organic compounds, BVOCs) were

estimated for the tree canopy using the algorithms of Guenther *et al.* (1991, 1993). The emission of carbon as isoprene is expressed as a product of a base emission factor adjusted for sunlight and temperature ($\mu g\text{-C}\ g^{-1}$ dry foliar biomass h^{-1}), a leaf biomass factor (g dry foliar biomass m^{-2} canopy projection) and a canopy projection area (m^2). Hourly emissions were summed to get monthly and annual emissions. The isoprene emission rate adjusted for sunlight and temperature is expressed as

$$I = I_s \cdot C_L \cdot C_T$$

where I is the isoprene emission rate at temperature T and photosynthetically active radiation flux PAR and I_s is the isoprene base emission factor at a standard temperature of 30°C and PAR flux of 1000 $\mu mol\ m^{-2}\ s^{-1}$. The light correction factor C_L is estimated as

$$C_L = \frac{\alpha C_{L1} L}{(1 + \alpha^2 L^2)^{1/2}}$$

where L is the PAR flux and α ($= 0.0027$) and C_{L1} ($= 1.066$) are empirical coefficients derived from emission rate measurements by Guenther *et al.* (1993). The leaf temperature correction factor C_T is given by

$$C_T = \frac{\exp[c_{T1}(T - T_s)/RT_s T]}{1 + \exp[c_{T2}(T - T_m)/RT_s T]}$$

where R is the ideal gas constant ($8.314\ J\ K^{-1}\ mol^{-1}$), T (K) is leaf temperature, T_s ($= 303\ K$) is the standard temperature and T_m ($= 314\ K$), C_{T1} ($= 95,000\ J\ mol^{-1}$) and C_{T2} ($= 230,000\ J\ mol^{-1}$) are empirical coefficients.

Monoterpene emissions are estimated as

$$M = M_s \exp[\beta(T - T_s)]$$

where M is the monoterpene emission rate at leaf temperature T (K), M_s is the base emission factor at T_s ($= 303\ K$) and β ($= 0.09$) is an empirical coefficient.

In natural forest stands, PAR and temperature profiles vary through the depth of the canopy layer. However, profiles of PAR and leaf temperature within urban forest canopies are not well known. We assume that leaf temperatures are equivalent to ambient air temperatures above the urban forest canopy, as a result of transpirational cooling (Geron *et al.*, 1994) and that the canopy leaf biomass is in full sunlight with hourly varying PAR during the day. PAR is estimated as a fraction of the hourly measured solar radiation ($W\ m^{-2}$) and converted to $\mu mol\ m^{-2}\ s^{-1}$ (Woodward and Sheehy, 1983).

Published emission factors (standardized at 30°C and 1000 $\mu mol\ m^{-2}\ s^{-1}$) for individual species and genera were used to estimate a canopy projection area-weighted average base emission factor that was representative of both the species composition and size distribution of trees represented by the Sacramento Shade planting list (Table 1). Where species-specific emission factors were unavailable, values were assigned

from other species of the same genus. Emission rates for species within a genus have been estimated to fall within $\pm 50\%$ (Geron *et al.*, 1994). When emission rates for the genera were lacking for two tree species, we assigned surrogates from other genera within their families (Horie *et al.*, 1991): *Malus* for *Pyrus* (family *Roseaceae*) and *Cupaniopsis anacardioides* for *Koeleria paniculata* (family *Sapindaceae*). The estimated area-weighted average base emission factor for isoprene and monoterpene were 7.4 and $1.8 \mu\text{g-C g}^{-1}$ dry foliar biomass h^{-1} at 30°C and $1000 \mu\text{mol m}^{-2} \text{s}^{-1}$, respectively (Table 1).

We assumed a leaf biomass factor of $375 \text{ g dry foliar biomass m}^{-2}$ canopy projection. The estimated leaf area index (LAI) for a "typical" open-grown 15-yr old Sacramento Shade program tree with a canopy projection of 49.6 m^2 and an average dry weight to leaf area ratio of 87.6 g m^{-2} leaf area (P. Peper, pers. comm., February 1996) is approximately 4.3.

2.5. Uncertainty estimates for deposition and emission

Pollutant uptake and BVOC emission estimates are sensitive to large uncertainties associated with canopy resistance and emission factors. Our use of average daytime and nighttime canopy resistances neglects their hourly variation. As a result, uptake during transition periods such as morning and evening may be overestimated, while midday values may be underestimated. Our daytime canopy resistance for ozone (1.74 s cm^{-1}) is greater than a summer daytime canopy resistance estimated by the Regional Acid Deposition Model (RADM) ($\sim 1.13 \text{ s cm}^{-1}$) for eastern deciduous forests (Wesley, 1989). Therefore, ozone uptake may be underestimated by a factor of 1.5–2.0. On the other hand, our daytime canopy resistance for ozone is 2–3 times less than measured over crops (Massman *et al.*, 1995), suggesting an overestimation of ozone uptake. Our in-leaf season daytime canopy resistance for NO_2 (3.01 s cm^{-1}) is higher than the RADM summer daytime average for eastern deciduous forests ($\sim 1.37 \text{ s cm}^{-1}$), so our model may underestimate daytime NO_2 uptake by a factor of 2. For this analysis we assume a factor 2 uncertainty for estimated ozone, NO_2 and SO_2 uptakes.

Factors that influence particulate dry deposition include atmospheric characteristics (e.g. turbulence), surface properties (e.g. roughness, albedo, wetness, chemical reactivity) and properties of the depositing particle species (size, mass, chemical composition). Particle resuspension is also influenced by these factors (Davidson and Wu, 1990). Based on limited literature for open-grown urban tree canopies (Dochinger, 1980; Nowak, 1994) we assumed a 50% resuspension rate for the base case, with values of 20% and 80% for the highest and lowest benefit scenarios. Deposition velocities for particles between 5 and $10 \mu\text{m}$ in diameter ranged from approximately 0.5 and 1.5 cm s^{-1} for a eucalyptus forest (Davidson and Wu, 1990). Thus, we assume a $\pm 50\%$ variation in the deposition velocity.

For the tree genera and species represented in our model tree population we assume that there is a $\pm 50\%$ uncertainty in the isoprene and monoterpene base emission factors and a $\pm 30\%$ uncertainty for the leaf biomass estimates (Horie *et al.*, 1990; Tanner *et al.*, 1992; Geron *et al.*, 1994).

2.6. Cost effectiveness analysis procedure

A pollution control technology is cost effective if the cost of controlling one ton of air pollutant is less than the cost specified in the BACT policy document. For this analysis we assume one cost scenario. Control costs for Sacramento are listed per kg as: $\$19.29$ for BVOC ($\$17,500 \text{ t}^{-1}$), $\$27.01$ for NO_2 ($\$24,500 \text{ t}^{-1}$), $\$11.68$ for PM_{10} ($\$10,600 \text{ t}^{-1}$), and $\$20.17$ for SO_2 ($\$18,300 \text{ t}^{-1}$) (SMAQMD, 1993). No cost is listed for ozone because it is formed in the atmosphere and not directly emitted. Because programs to reduce ozone are increasingly aimed at reducing NO_2 , the control price for NO_2 was applied to ozone. If the maximum control cost (MCC) is greater than the total annual control cost (TAC) a control is cost effective. MCC is estimated as

$$\text{MCC} = \frac{\sum_{k=1}^n \sum_{j=1}^5 (c_j f_{c_j})}{n}$$

where c_j is the annual amount of pollution deposition or emission for pollutant species j , f_{c_j} is the control cost per pollutant species, and pollutant uptake or emission is calculated for each year k to year n (30). TAC is estimated as

$$\text{TAC} = \text{CRC} + \text{MRC}$$

The capital recovery cost (CRC) is the uniform end-of-year payment necessary to repay the investment (P) in n years (30) with interest rate I (10%) where

$$\text{CRC} = \text{CRF} \times P$$

and the cost recovery factor (CRF) is

$$\text{CRF} = \frac{i(1+i)^n}{(1+i)^n - 1}$$

The maintenance recovery cost (MRC) is the uniform annual payment necessary to repay the present value of 30-yr tree maintenance costs (M) with an interest rate I (10%) where

$$\text{MRC} = \text{CRF} \times M \quad \text{and} \quad M = \sum_{k=1}^n \frac{m_k}{(1+i)^k}$$

where m_k is the future value of total annual tree care costs for each year k to year n (30).

Tree planting and care costs were obtained from a survey of local garden centers and arborists. Costs were estimated for planting, removal of dead trees and stumps, pruning, and irrigation water. Although additional costs could accrue (e.g. pest and disease control), they are likely to be small if trees are judiciously selected and located.

A planting cost of \$45 per tree (19 ℓ [5 gal] size) for 100 trees was assumed. This price is midway between the likely range of \$25 to \$65. Twenty-one trees dying during the 5-year establishment period were assumed to be removed by the resident at no cost. Based on a survey of local arborists, removal costs for the remaining 12 trees were \$7.87 cm⁻¹ d.b.h. (\$20 in.⁻¹) and ranged from \$142 to \$360 per tree for trees between 18 and 45 cm (7–18 in) d.b.h. Stumps of every other dead tree were removed at a price of \$1.97 cm⁻¹ d.b.h. (\$5 in.⁻¹). Pruning of young trees less than 7.5 m (24 ft) was assumed to be done by the residents. We conservatively estimated pruning costs at \$3.94 cm⁻¹ d.b.h. (\$10 in.⁻¹) per tree for 50% of the trees projected to be alive at 15 and 25 years after planting. Hence, we assumed that half of the live trees would be pruned by a professional arborist once they grew too large for pole pruning by residents.

Sacramento water is priced with a flat rate schedule. Because the amount residents pay for water is independent of their consumption, additional water used to irrigate new trees will not increase their payments. However, there are social costs associated with increased water consumption that may be reflected in current water pricing. To capture these social costs we conservatively include irrigation water costs for the first 5 years, assuming that thereafter trees will not increase landscape water demand. Most trees are likely to be planted in turf and other irrigated areas where they can harvest surplus water and their shade can reduce the irrigation demand of nearby understory plants. Annual irrigation water use IWU (m³) was estimated as

$$IWU = (ET_0 - P) \times \left(K \times CP \times \frac{1}{H} \right)$$

where ET_0 is the reference evapotranspiration (m), P the precipitation (m), K the crop coefficient, CP the canopy projection (m²), and H the irrigation efficiency (%).

Net evaporative demand in Sacramento was calculated using historic precipitation and ET_0 data (NOAA, 1990; Snyder *et al.*, undated). Values of K and H were 0.5 and 0.7, respectively. These values are characteristic of tree species requiring moderate amounts of water in a turf landscape that is efficiently irrigated with automatic sprinklers (McPherson *et al.*, 1991). The City of Sacramento charged commercial customers \$0.19 m⁻³ (\$0.54 100 ft⁻³) for water delivered in 1995 (pers. comm., Phillip McAvoy, Department of Utilities, 1995).

3. RESULTS

3.1. Tree growth, mortality, and canopy projection area

The typical tree's weighted average canopy projection area (CP) was 122 m² (1312 ft²) and its average

crown diameter was 12.5 m (41 ft) at 30 yr from planting. Tree height and trunk diameter at breast height (d.b.h.) were estimated as 13.7 m (45 ft) and 48 cm (18 in.) at year 30, respectively. Total CP, obtained yearly as the product of live trees and typical tree CP, followed an S-shaped curve: slow at first, most rapid during the middle years, and slowing as the trees reach mature size 30 yr after planting (Fig. 1). At year 30 the CP area covered 11.6% of the neighborhood (8181 m²). The CP area associated with benefits used in the BACT calculation is the 30-yr annual average for all trees, or 4096 m² (5.8% land cover). At this time the typical tree's CP is approximately 50 m² (538 ft²) and crown spread is 8 m (26 ft).

3.2. Deposition and emission

Throughout the study period, pollutant removal was greatest for ozone and PM₁₀ (Fig. 2). Assuming base uptake and emission values, the estimated 30 yr annual average deposition per 100 trees planted was 22.2 kg for ozone; 13.5 kg for PM₁₀; 4.5 kg for NO₂ and 0.8 kg for SO₂ (Table 2). The base case 30 yr annual average BVOC emissions was estimated at 26.5 kg-C (Table 2).

For a live tree 30 yr after planting, the estimated annual uptake for ozone was 0.66 kg; 0.4 kg for PM₁₀; 0.13 kg for NO₂ and 0.02 kg for SO₂. These estimates are in reasonable agreement with values simulated for a tree in the Chicago area (Nowak, 1994). Annual biogenic hydrocarbon emission for the same tree was estimated at 0.56 kg-C for isoprene and 0.23 kg-C for monoterpenes. Peak total monthly deposition for ozone occurred in July (0.1 kg) and August (0.09 kg); for NO₂ in August (0.02 kg) and October (0.03 kg); for SO₂ in April (0.02 kg) and for PM₁₀ in September (0.06 kg) and October (0.08 kg). Peak total monthly BVOC emissions occurred in July and August (0.11 kg-C for isoprene, 0.04 kg-C for monoterpene). Minimum total monthly depositions for ozone occurred in March and November (0.04 kg and 0.02 kg, when

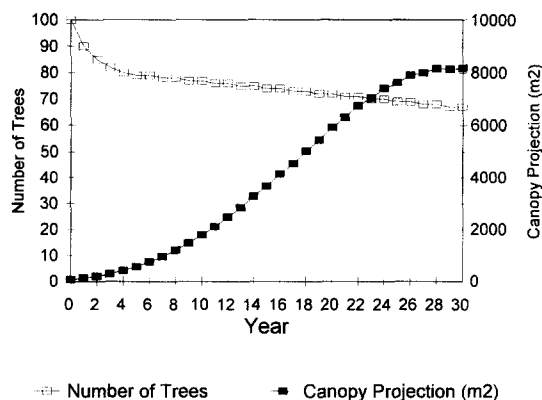


Fig. 1. Number of live trees and canopy projection over study period.

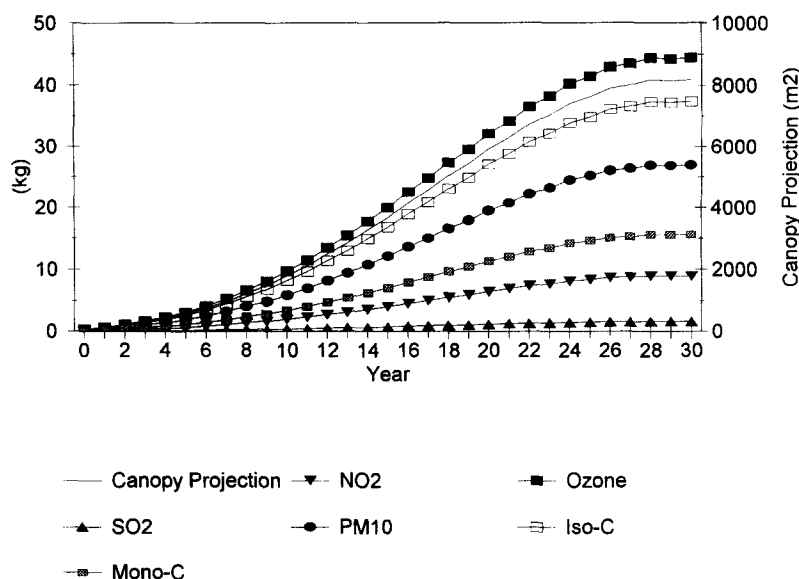


Fig. 2. Estimated annual pollutant uptake and emission for the model tree population assuming base values.

Table 2. Average annual pollution emission, uptake, and costs

Pollutant	fc ^c (\$ kg ⁻¹)	Base values			Highest benefits ^a			Lowest benefits ^b		
		Amount ^d (lb)	(kg)	Total cost (\$)	Amount (lb)	(kg)	Total cost (\$)	Amount (lb)	(kg)	Total cost (\$)
BVOC emitted	-19	58.5	26.5	-510	20.5	9.3	-180	114.1	51.8	-1,000
O ₃ uptake	27	49.0	22.2	600	98.0	44.5	1,200	24.5	11.1	300
NO ₂ uptake	27	9.9	4.5	120	19.7	9.0	240	4.9	2.2	60
PM ₁₀ uptake	12	29.7	13.5	160	71.4	32.4	380	5.9	2.7	30
SO ₂ uptake	20	1.8	0.8	20	3.5	1.6	30	0.9	0.4	10
Net		31.9	14.5	390	172.2	78.1	1,670	-77.8	-35.3	600

^aAssumes upper bound uptake rates and lower bound BVOC emission rates.

^bAssumes lower bound uptake rates and upper bound BVOC emission rates.

^cMaximum control costs for each pollutant.

^dDivide amounts and total costs by 100 to derive average annual cost per tree planted.

trees were foliated only during portions of these months), in June for NO₂ (less than 5 g) and PM₁₀ (0.02 kg) and in September for SO₂ (less than 1 g).

Figure 3 shows estimated hourly fluxes of pollutant and biogenic hydrocarbons on 10 August 1990 for a healthy tree 30 yr after planting. Peak ozone uptake occurs in the late afternoon. Isoprene emissions follow the variation in PAR and temperature, peaking at noontime, while monoterpene emissions peak during the late afternoon. As light levels and temperatures increase during the late morning, isoprene and monoterpene fluxes approach 3 and 0.7 mg-C m⁻² h⁻¹, respectively. Measured emission rates over southeastern U.S. deciduous forests were recently reported to average between 3 and 4 mg-C m⁻² h⁻¹ for isoprene and 0.5 and 0.7 mg-C m⁻² h⁻¹ for monoterpenes (at 30°C and 1000 μmol m⁻² s⁻¹), with maximum isoprene emission rates ranging between

5 and 6 mg-C m⁻² h⁻¹ at ambient temperature and light levels (Guenther *et al.*, 1996).

3.3. Cost effectiveness

The planting cost for 100 trees was estimated to be \$4500. The cost recovery factor (CRF) and capital recovery cost (CRC) for planting were 10.6% and \$480, respectively. The total present value of tree maintenance costs was estimated as \$2560 and the maintenance recovery cost (MRC) was \$270. Maintenance costs were quite variable. The total future value (costs not discounted) of pruning (\$10,830) and tree removal (\$3330) costs were greatest, while costs were least for irrigation water (\$230) and stump removal (\$430). The total annual cost (TAC), or uniform payment necessary to repay planting and maintenance costs over 30 years with an interest rate of 10%, was estimated as \$750 (CRC + MRC).

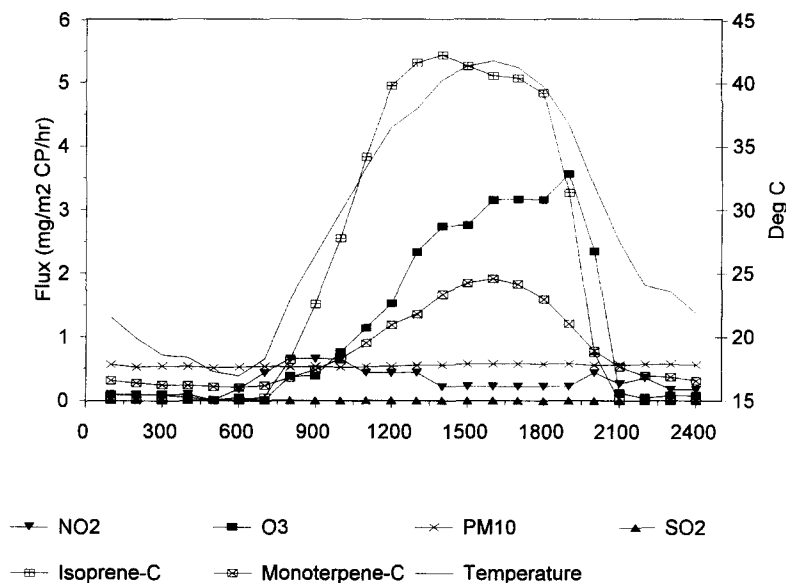


Fig. 3. Estimated hourly pollutant and BVOC fluxes for a healthy tree 30 years after planting, assuming base values (10 August 1990).

For the scenario assuming base case values, deciduous trees were estimated to emit 27 kg (59 lb) of BVOC per year on average, at an annual cost of \$510 (Table 2). Deposition of all other pollutants totaled 41 kg (90 lb), for an annual benefit of \$900. The net pollution mitigation benefit (value of uptake minus emissions), or MCC was estimated as \$380. The hypothetical tree planting is not cost effective because MCC is \$370 less than TAC (\$750). The benefit-cost ratio is 0.5:1 (Table 3).

The scenario depicting highest possible air quality benefits assumes upper bound deposition rates and lower bound biogenic hydrocarbon emission rates. A benefit-cost ratio of 2.2:1 indicates that the planting may be cost effective (Table 3). For this case, the average annual value of pollutant uptake (87 kg) was estimated as \$1670, while the cost of BVOC emissions (9 kg) was \$180 (Table 2). For the scenario depicting lowest possible air quality benefits we assume lower bound deposition and upper bound emission rates. Average annual BVOC emissions (52 kg-C) exceed pollutant uptake (16 kg) and the cost of emissions is \$600 greater than the deposition benefit (Table 2). The benefit-cost ratio of $-0.8:1$ reflects negative net benefits of \$1350 for the project (Table 3).

4. CONCLUSIONS AND LIMITATIONS OF THE STUDY

These findings do not suggest convincing evidence that a large-scale residential yard tree planting program such as Sacramento Shade will produce costs savings based solely on air quality benefits. The results reflect the stringent accounting requirements of BACT analysis. For example, inclusion of long term tree care costs borne by residents rather than the

Table 3. Costs, benefits, and benefit-cost ratios

	Base values	Highest benefits ^a	Lowest benefits ^b
Benefit (MCC)	\$380	\$1,670	\$ - 600
Cost (TAC)	\$750	\$750	\$750
Net Benefit (MCC-TAC)	\$ - 370	\$920	\$ - 1350
Benefit-cost ratio (B/C)	0.5	2.2	- 0.8

^aAssumes upper bound uptake rates and lower bound BVOC emission rates.

^bAssumes lower bound uptake rates and upper bound BVOC emission rates.

hypothetical investing air quality district, as well as the relatively high interest rate combine to diminish the present value of net benefits. Also, characteristics of the Sacramento Shade program, such as the mix of tree species and program delivery costs, limit the extent to which these results can be generalized to other tree planting programs.

The findings should not be interpreted to mean that the costs of planting and caring for trees in Sacramento to exceed their benefits. Rather, since the Sacramento Municipal Utility District has found that their shade tree program produces net benefits from air conditioning savings alone, additional benefits from air quality improvement can be viewed as increasing their ratepayers return on investment (Hildebrandt, 1996). A more comprehensive accounting method that includes benefits from air conditioning savings, atmospheric carbon dioxide removal, and property value increases might result in benefits exceeding costs.

However, as the benefit-cost ratios for highest and lowest benefit scenarios of 2.2:1 and $-0.8:1$ indicate, our ability to estimate the extent to which shade trees

produce air quality benefits is impaired by large uncertainties regarding rates of pollutant deposition and, especially, biogenic hydrocarbon emissions. Thirty-year annual average BVOC emissions ranged between a lower bound of 9.3 kg-C and an upper bound of 51.8 kg-C per 100 trees planted. To make the modeling effort more accurate, biometric measurements are needed for common urban tree species so as to better estimate foliar biomass and BVOC emission factors. The range of total annual pollutant deposition rates was similarly large: 16.5–87.4 kg. We used canopy resistance values for rural forests because data are lacking for urban trees. We expect urban trees might have lower canopy resistance values than trees in rural forests due to lower levels of water stress and therefore, higher gas exchange rates.

The benefit-cost results also are sensitive to species composition. Both mature size and biogenic hydrocarbon emission factors influence total emissions. Relatively high overall BVOC emissions are due in part to the relative abundance of large-growing oaks with high emission factors in the model tree population. For instance, if oaks were replaced by maples in the model population, emissions would decrease by approximately 25% and the benefit-cost ratio for the base value scenario would increase from 0.5 to 0.7. If London plane, goldenrain, and sweetgum were also replaced by maple, emissions would decrease by approximately 60%, and the benefit-cost ratio for the base value scenario would increase from 0.5 to 0.9.

There are other limitations to this analysis. These limitations include little data on property owner investment in urban tree care services; rates of tree growth and survival; extrapolation of work in chambers and forest stands to relatively open-grown urban trees; lack of consideration of tree configuration and local meteorology; and little knowledge about the relative ability of different tree species to intercept particulates and absorb gaseous pollutants, or their tolerance to air pollution stress. Moreover, our analysis occurs at a small scale without consideration of regional photochemical and meteorological processes. Current research will address questions regarding the net impact of urban forests on regional climate and air quality.

In our analysis we applied the NO₂ control cost to the value of ozone uptake by the trees. This is problematic because this treats ozone as a primary pollutant and it neglects non-linear photochemical reactions involved in ozone formation. An alternative valuation scheme could involve adjusting BVOC costs with a correction factor representing a ratio of ozone uptake to ozone produced as a result of BVOC emissions. A theoretical estimate of the potential ozone formation due to BVOC emissions could be made using maximum incremental reactivity factors (Carter, 1994) and a methodology recently developed by Benjamin and Winer (1996). Incremental reactivity is defined as the change in ozone concentration caused by adding small amounts of a test volatile

organic compound (VOC) to an initial mix of reactants or emissions in a given episode (e.g. a summer day), divided by the amount of test volatile organic compound added (Carter, 1994). Incremental reactivities also depend on the availability of NO_x, expressed as a ratio of total emissions of reactive organic gases (in this case BVOCs) to NO_x. Knowing the initial local atmospheric and photochemical conditions for a given episode, the methodology could be used as a basis to extend the modeling approach.

Despite these limitations, this research builds on previous work by estimating the economic value of vegetation impacts on air quality. It is one of the first studies to attempt to account for both biogenic hydrocarbon emissions and ozone uptake by trees. It considers a 30 year stream of benefits and costs associated with an aging urban forest planting, whereas previous studies have been limited to periods of a year or less. Finally, by applying the BACT analysis it allows air resource managers and other potential investors to assess the implied value of air quality benefits from the urban forest.

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REFERENCES

- Air Resources Board (1991) California Air Quality Data. Summary of 1990 Air Quality Data: Gaseous and Particulate Pollutants. TSD-91-006, California Environmental Protection Agency, Sacramento, CA.
- Air Resources Board (1994) California state and local air monitoring network plan. TSD-94-001, California Environmental Protection Agency, Sacramento, CA.
- Benjamin, M. T. and Winer, A. M. (1997) Estimating the ozone-forming potential of urban trees and shrubs. *Atmospheric Environment*, Part B: Urban Atmospheres, in press, this issue.
- Cardelino, C.A. and Chameides, W. L. (1990) Natural hydrocarbons, urbanization and urban ozone. *Journal of Geophysical Research* **95**, 13,971–13,979.
- Carter, W. P. L. (1994) Development of ozone reactivity scales for volatile organic compounds. *Journal of the Air and Waste Management Association* **44**, 881–899.
- Chameides, W. L., Lindsay, R. W., Richardson, J. and Kiang, C. S. (1988) The role of biogenic hydrocarbons in urban photochemical smog: Atlanta as a case study. *Science* **241**, 1473–1475.
- Corchnoy, S. B., Arey, J. and Atkinson, R. (1992) Hydrocarbon emissions from twelve urban shade trees of the Los Angeles, California, air basin. *Atmospheric Environment* **26B**, 339–348.
- Davidson, C. I. and Wu, Y.-L. (1990) Dry deposition of particles and vapors. In *Acidic Precipitation*, eds. Lindberg, S. E., Page, A. L. and Norton S. A., Vol. 3: Sources, Deposition and Canopy Interactions, pp. 103–216. Springer, New York.
- Dawson, J. O. and Khawaja, M. A. (1985) Change in street-tree composition of two Urbana, Illinois neighborhoods

- after fifty years: 1932–1982. *Journal of Arboriculture* **11**, 344–348.
- DeSanto, R., Glase, R., McMillan, W., MacGregor, K. and Miller, J. (1976a) Open space as an air resource management measure. Vol. II: Design criterion. EPA-450/3-76-028b. U.S. Environmental Protection Agency, Research Triangle Park, NC.
- DeSanto, R., MacGregor, K., McMillan, W. and Glase, R. (1976b) Open space as an air resource management measure. Vol. III: Demonstration plan (St. Louis, MO). EPA-450/3-76-028c. U.S. Environmental Protection Agency, Research Triangle Park, NC.
- Dochinger, L. S. (1980) Interception of airborne particulates by tree plantings. *Journal of Environmental Quality* **9**, 265–268.
- Dyer, A. J. and Bradley, C. F. (1982) An alternative analysis of flux gradient relationships. *Boundary-Layer Meteorology* **22**, 3–19.
- Geron, C. D., Guenther, A. B. and Pierce, T. E. (1994) An improved model for estimating emissions of volatile organic compounds from forests in the eastern United States. *Journal of Geophysical Research* **99**, 12,773–12,791.
- Gifford, F. A. (1976) Turbulent diffusion typing schemes: a review. *Nuclear Safety* **17**, 71.
- Golder, D. (1972) Relations among stability parameters in the surface layer. *Boundary-Layer Meteorology* **3**, 47–58.
- Guenther, A. B., Monson, R. K. and Fall, R. (1991) Isoprene and monoterpene emission rate variability: observations with eucalyptus and emission rate algorithm development. *Journal of Geophysical Research* **96**, 10,799–10,808.
- Guenther, A. B., Zimmermann, P. R., Harley, P. C., Monson, R. K. and Fall, R. (1993) Isoprene and monoterpene emission rate variability: model evaluations and sensitivity analyses. *Journal of Geophysical Research* **98**, 12,609–12,617.
- Guenther, A. B., Zimmerman, P. R. and Wildermuth, M. (1994) Natural volatile organic compound emission rate estimates for U.S. woodland landscapes. *Atmospheric Environment*, **28**, 1197–1210.
- Guenther, A. B., Zimmerman, P., Klinger, L., Greenberg, J., Ennis, C., Davis, K., Pollock, W., Westberg, H., Allwine, G. and Geron, C. (1996) Estimates of regional natural volatile organic compound fluxes from enclosure and ambient measurements. *Journal of Geophysical Research* **101**, 1345–1359.
- Hildebrandt, E. (1996) *Maximizing the energy benefits of urban forestation*. Proceedings of the ACEEE 1996 Summer Study on Energy Efficiency in Buildings, Asilomar, California.
- Horie, Y., Sidawi, S. and Ellefsen, R. (1991) Inventory of leaf biomass and emission factors for vegetation in California's South Coast Air Basin. Final Technical Report III-C. South Coast Air Quality Management District.
- Killus, J. P., Meyer, J., Durran, D., Anderson, G., Jerskey, T., Reynolds, S. and Ames, J. (1984) Continued research in mesoscale air pollution simulation modeling. Vol. 5: Refinements in numerical analysis, transport, chemistry and pollutant removal. EPA/600/3-84/095a, U.S. Environmental Protection Agency, Research Triangle Park, NC.
- Massman, W. J., Pederson, J., Delany, A., Grantz, A., den Hartog, G., Neumann, H. H., Oncley, S. P., Pearson, R., Jr, Shaw, R. H. (1994) An evaluation of the regional acid deposition model surface module for ozone uptake at three sites in the San Joaquin Valley of California. *Journal of Geophysical Research* **99** (D4), 8281–8294.
- McPherson, E. G., Minor, C. and Guertin, D. (1991) A prototype model of residential irrigation water use. Preprints, Tenth Conference on Biometeorology and Aerobiology and the Special Session on Hydrometeorology, Salt Lake City, UT, American Meteorological Society, pp. 63–68.
- McPherson, E. G. (1994) Benefits and costs of tree planting and care in Chicago. In *Chicago's Urban Forest Ecosystem: Results of the Chicago Urban Forest Climate Project*, (eds. E. G. McPherson, D. J. Nowak, and R. A. Rowntree), pp. 115–133. General Technical Report No. NE-186, U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station, Radnor, PA.
- Miller, R. H. and Miller, R. W. (1991) Planting survival of selected street tree taxa. *Journal of Arboriculture* **17**, 185–191.
- Morris, R. E., Yocke, M. A., Meyers, T. C. and Mirabella, V. (1992) *Overview of the variable-grid Urban Airshed Model (UAM-V)*. Proceedings of the 85th annual meeting of the Air and Waste Management Association, Kansas City, MO, 92-86.10.
- Nowak, D. J. (1994) Air pollution removal by Chicago's urban forest. In *Chicago's Urban Forest Ecosystem: Results of the Chicago Urban Forest Climate Project* (Eds. McPherson, E. G., Nowak, D. J. and Rowntree, R. A.), pp. 63–82. General Technical Report No. NE-186, U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station, Radnor, PA.
- NOAA (1990) Local climatological data, annual summary with comparative data. Sacramento, CA. National Climatic Data Center, Asheville, NC.
- SMAQMD, Sacramento Metropolitan Air Quality Management District (1993) BACT cost analysis policy document. SMAQMD, Sacramento, CA.
- Sailor, D. J. (1995) Simulated urban climate response to modifications in surface albedo and vegetative cover. *Journal of Applied Meteorology* **34**, 1694–1704.
- Smith, W. H. (1981) *Air Pollution and Forests: Interactions Between Air Contaminants and Forest Ecosystems*. Springer, New York.
- Snyder, R. L., Pruitt, W. O. and Shaw, D. A. undated. Determining daily reference evapotranspiration. Leaflet 21426. Cooperative Extension-University of California, Division of Agriculture and Natural Resources, Oakland, CA.
- Stull, R. B. (1989) *An Introduction to Boundary Layer Meteorology*. Kluwer Academic Publishers, Boston.
- Tanner, R. L., Minor, T., Hatzell, J., Jackson, J., Rose, M. and Zielinska, B. (1992) Emissions data collection and inventory development. Work element 2: Development of a natural source emissions inventory. DRI Final Report No. 8303-009.FR1. Desert Research Institute, Reno, NV.
- van Ulden, A. P. and Holtslag, A. A. M. (1985) Estimation of atmospheric boundary layer parameters for diffusion application. *Journal of Climate and Applied Meteorology*, **24**, 1196–1207.
- Weedall, M. (1995) What we know about shade trees and energy use in Sacramento. In *Benefits of the Urban Forest*, pp. 5–6. Sacramento, CA: Sacramento Municipal Utility District.
- Wesley, M. L. (1989) Parameterization of surface resistances to gaseous dry deposition in regional-scale numerical models. *Atmospheric Environment* **23**, 1293–1304.
- Winer, A. M., Fitz, D. R. and Miller, P. R. (1983) Investigation of the role of natural hydrocarbons in photochemical smog formation in California. Final Report, Contract No. AO-056-32. Sacramento, CA: California Air Resources Board.
- Woodward, F. I. and Sheehy, J. E. (1983) *Principles and Measurements in Environmental Biology*. Butterworth, London.