

# Reforestation as a novel abatement and compliance measure for ground-level ozone

Timm Kroeger<sup>a,1</sup>, Francisco J. Escobedo<sup>b</sup>, José L. Hernandez<sup>c,2</sup>, Sebastián Varela<sup>b,3</sup>, Sonia Delphin<sup>b</sup>, Jonathan R. B. Fisher<sup>a</sup>, and Janice Waldron<sup>d</sup>

<sup>a</sup>Central Science Department, Nature Conservancy, Arlington, VA 22203; <sup>b</sup>School of Forest Resources and Conservation, University of Florida, Gainesville, FL 32611; <sup>c</sup>ENV DAT Consulting, Knoxville, TN 37923; and <sup>d</sup>Texas Operations, Dow Chemical Company, Freeport, TX 77541

Edited\* by Peter M. Kareiva, Nature Conservancy, Seattle, WA, and approved August 14, 2014 (received for review May 27, 2014)

High ambient ozone (O<sub>3</sub>) concentrations are a widespread and persistent problem globally. Although studies have documented the role of forests in removing O<sub>3</sub> and one of its precursors, nitrogen dioxide (NO<sub>2</sub>), the cost effectiveness of using peri-urban reforestation for O<sub>3</sub> abatement purposes has not been examined. We develop a methodology that uses available air quality and meteorological data and simplified forest structure growth-mortality and dry deposition models to assess the performance of reforestation for O<sub>3</sub> precursor abatement. We apply this methodology to identify the cost-effective design for a hypothetical 405-ha, peri-urban reforestation project in the Houston–Galveston–Brazoria O<sub>3</sub> nonattainment area in Texas. The project would remove an estimated 310 tons of (t) O<sub>3</sub> and 58 t NO<sub>2</sub> total over 30 y. Given its location in a nitrogen oxide (NO<sub>x</sub>)-limited area, and using the range of Houston area O<sub>3</sub> production efficiencies to convert forest O<sub>3</sub> removal to its NO<sub>x</sub> equivalent, this is equivalent to 127–209 t of the regulated NO<sub>x</sub>. The cost of reforestation per ton of NO<sub>x</sub> abated compares favorably to that of additional conventional controls if no land costs are incurred, especially if carbon offsets are generated. Purchasing agricultural lands for reforestation removes this cost advantage, but this problem could be overcome through cost-share opportunities that exist due to the public and conservation benefits of reforestation. Our findings suggest that peri-urban reforestation should be considered in O<sub>3</sub> control efforts in Houston, other US nonattainment areas, and areas with O<sub>3</sub> pollution problems in other countries, wherever O<sub>3</sub> formation is predominantly NO<sub>x</sub> limited.

air pollution | ecosystem services | natural infrastructure | state implementation plan

Ground-level (tropospheric) ozone (O<sub>3</sub>) is a secondary air pollutant formed through the chemical interaction of nitrogen oxides (collectively referred to as NO<sub>x</sub> and comprising NO and NO<sub>2</sub>) and volatile organic compounds (VOC) in the presence of conducive solar radiation and temperature conditions (1). Ground-level O<sub>3</sub> is considered one of the most pervasive and damaging air pollutants globally, with background concentrations that have more than doubled in the northern hemisphere since the late nineteenth century (2). Despite widespread and often decadeslong control efforts, ambient O<sub>3</sub> concentrations in urban areas in many parts of the world regularly exceed the World Health Organization guideline value of 50 parts per billion (ppb; daily 8-h average concentration) (3).

Despite the highly complex nature of estimating O<sub>3</sub> health effects (4), O<sub>3</sub> has been linked to increased mortality in humans (4–7), with an estimated annual death toll of 28,000 in Europe (8) and 152,000 [95% confidence interval (CI): 52,000–276,000] globally (9), and to reduced worker productivity (10) and increased respiratory and cardiovascular disease (7, 9). In Europe, an estimated 39,000 respiratory hospital admissions per year are attributed to O<sub>3</sub> concentrations above 35 ppb (8). In the United States, an estimated 10.7 (90% CI: 5.5–15.8) million acute respiratory symptoms; 5,300 (90% CI: 0–11,900) respiratory emergency room visits; 4,100 (90% CI: 1,100–7,900) respiratory

hospital admissions; and 3.7 (90% CI: 1.6–5.9) million school loss days could have been avoided per year on average during 2005–2007 if O<sub>3</sub> concentrations in those years had been reduced such that their 8-h averages would not have exceeded 60 ppb anywhere (11). Ozone also has been shown to reduce food crop and forest productivity (12, 13) and is an important greenhouse gas (2).

Efforts to reduce ambient concentrations of O<sub>3</sub> and other pollutants have relied predominantly on engineering-based approaches to reduce emissions from fossil fuel combustion processes, implemented via command-and-control or market-based mechanisms (14). These have included physical dilution of emissions via tall stacks (15); intermittent or permanent, partial, or complete plant shutdowns (16); conversion to lower-emitting combustion processes and fuels (17); and end-of-pipe controls (18).

Despite these control efforts, high ambient O<sub>3</sub> concentrations remain a widespread problem in many areas of the world. In the United States, O<sub>3</sub> is regulated by the Environmental Protection Agency (EPA) as a hazardous air pollutant. In 2013, there were 46 areas with a total population of 123 million that were designated as O<sub>3</sub> nonattainment areas because at least one monitor exceeded the 75 ppb (daily 8-h average) 2008 National Ambient Air Quality Standard (NAAQS) for O<sub>3</sub> 3 times a year (19). States

## Significance

Despite often decadeslong control efforts, in many regions of the world ambient concentrations of ground-level ozone threaten human and ecosystem health. Furthermore, in many places the effects of continuing land use and climate change are expected to counteract ongoing efforts to reduce ozone concentrations. Combined with the rising cost of more stringent conventional technological ozone controls, this creates a need to explore novel approaches to reducing tropospheric ozone pollution. Reforestation of peri-urban areas, which removes ozone and one of its precursors, may be a cost-effective approach to ozone control and can produce important ancillary benefits. We identify key criteria for maximizing the ozone abatement and cost effectiveness of such reforestation and the substantial potential for its application in the United States.

Author contributions: T.K., F.J.E., J.R.B.F., and J.W. designed research; T.K., F.J.E., J.L.H., S.V., S.D., and J.R.B.F. performed research; T.K., F.J.E., J.L.H., and J.R.B.F. analyzed data; J.W. contributed data; and T.K. and F.J.E. wrote the paper.

The authors declare no conflict of interest.

\*This Direct Submission article had a prearranged editor.

Freely available online through the PNAS open access option.

<sup>1</sup>To whom correspondence should be addressed. Email: tkroeger@tnc.org.

<sup>2</sup>Present address: US Department of the Interior, Bureau of Ocean Energy Management – Gulf of Mexico Region, New Orleans, LA 70123.

<sup>3</sup>Present address: Centro de Transporte Sustentable de México-EMBARQ-WRI, México, D.F. C.P. 04000.

This article contains supporting information online at [www.pnas.org/lookup/suppl/doi:10.1073/pnas.1409785111/-DCSupplemental](http://www.pnas.org/lookup/suppl/doi:10.1073/pnas.1409785111/-DCSupplemental).

are required to develop and implement EPA-approved State Implementation Plans (SIP) for each nonattainment area that outline measures deployed to achieve attainment. Because EPA has jurisdiction over mobile sources, states pursue attainment principally by imposing emission limits on large industrial processes and utilities (point sources) and smaller stationary processes (area sources). Due to the often dominant (>50%; NO<sub>x</sub>) or large (25–50%; VOC) contribution of point sources to total stationary O<sub>3</sub> precursor emissions, the imposition of limits on permitted point source precursor emissions is a key SIP component in these nonattainment areas. Point sources comply with their NO<sub>x</sub> emission limits by installing combustion controls (fuel switching, low-NO<sub>x</sub> burners, fuel reburning, flue gas recirculation), end-of-pipe controls (selective catalytic or noncatalytic reduction), or by purchasing emission credits on the precursor-specific cap-and-trade markets established for many nonattainment areas. The 1990 Clean Air Act Amendments (20) create a further incentive for attaining NAAQS by imposing fines for VOC and NO<sub>x</sub> emissions from major sources in areas that fail to meet attainment deadlines.

In the United States, the O<sub>3</sub> problem may worsen due to continuing land use and climate change, especially rising temperatures (21–24). A possible tightening of the O<sub>3</sub> NAAQS due to health concerns (25) may cause further reductions in precursor emission limits in many areas. The picture is similar in many other regions of the world (2). Because marginal control costs are increasing (26), achieving additional abatement will become increasingly costly. Thus, there is a pressing need to find new, cost-effective approaches to addressing the O<sub>3</sub> problem.

One as yet largely unexplored possibility for O<sub>3</sub> abatement is reforestation. Forests have been shown to reduce ambient concentrations of many anthropogenic air pollutants, in both urban and immediately adjacent peri-urban areas located between rural areas and the outer boundary of urban settlements (27–30). Trees absorb and diffuse ambient NO<sub>2</sub> and O<sub>3</sub> via dry deposition and foliar gas exchange, lowering the concentrations of these gases in the air mass moving through the forest canopy (27). Trees also release VOCs in response to many biophysical factors, increasing ambient VOC concentrations (28). The net effect of a reforestation project on O<sub>3</sub> concentrations depends on the magnitude of these two processes and on whether the project is located in an area where O<sub>3</sub> formation is limited by available NO<sub>x</sub> or VOC, respectively. Using atmospheric chemistry and transport and meteorological models, Alonso et al. (31) found that peri-urban forests near Madrid, Spain, were O<sub>3</sub> sinks. Using forest structure data and a coupled dry deposition and mesoscale weather prediction model, Baumgardner et al. (32) found that a peri-urban forest near Mexico City improved regional air quality by removing O<sub>3</sub> and respirable particulate matter.

These studies raise the question of whether reforestation—and forest management and conservation more broadly—might constitute a novel and cost-effective approach to O<sub>3</sub> abatement by removing its precursor gases from the atmosphere at lower cost per unit precursor removed than engineering alternatives. If so, regulated emitters with a portfolio of abatement choices—such as many point sources, which currently can choose to achieve compliance with their emission limits via installation of various control technologies, purchase of precursor emission credits on nonattainment-area-specific cap-and-trade markets, or both—in principle might deploy reforestation projects to generate part of the required precursor abatement. Previous analyses have found that urban trees can be a cost-effective public strategy for improving air quality (33, 34). However, the private cost effectiveness and financial feasibility for regulated point sources of using peri-urban reforestation projects for O<sub>3</sub> precursor control remains unexamined.

We develop an integrated methodology that provides guidance for evaluating the long-term performance of reforestation in

peri-urban areas for O<sub>3</sub> control and analyzing its cost-effectiveness as a compliance approach. We select the Houston–Galveston–Brazoria (HGB) O<sub>3</sub> nonattainment area in Texas as a case study because it exceeds O<sub>3</sub> standards and exhibits large-scale reforestation potential.

We first identify key siting and design parameters that affect the cost of reforestation projects per ton (t) of O<sub>3</sub> precursors removed. Second, we develop a simplified tree growth-mortality model that predicts key forest canopy parameters that affect air pollutant removal. We use these canopy parameters along with meteorological and ambient pollutant concentration data and the Urban Forest Effects (UFORE) dry-deposition air-pollutant removal and biogenic emissions model (30) to generate estimates of pollutant deposition and biogenic VOC emissions for a hypothetical 405 hectare (ha) reforestation project in the HGB area. Next, we combine removal estimates and reforestation and land costs to estimate the cost of the project per ton of O<sub>3</sub> precursor abated, with and without the carbon (C) credits such a project could generate under the California Air Resources Board (CARB; ref. 35) forest project offset protocol, the highest-price carbon market US reforestation projects can currently access. We compare these costs with those of conventional point-source NO<sub>x</sub> controls in the HGB area. We also quantify the social economic value of the C sequestered by the project. Finally, we identify where guidance is needed from regulatory authorities in the selection of key estimation parameters to reduce uncertainties and narrow ranges in pollutant removal and cost-effectiveness estimates.

It is important to note that peri-urban and urban forests provide a wide range of ecosystem services in addition to air-quality improvement (36, 37). Reforestation thus may yield a series of cobenefits not provided by conventional engineering-based controls.

## Results and Discussion

Forests remove both O<sub>3</sub> and NO<sub>2</sub>. However, because O<sub>3</sub> is not emitted directly, a SIP regulates point-source emissions not of O<sub>3</sub> but of its precursors, NO<sub>x</sub> and VOC. Thus, the objective of an O<sub>3</sub> SIP reforestation project is the abatement of O<sub>3</sub> precursor equivalents. In the case of NO<sub>x</sub>, these equivalents (NO<sub>x</sub>e; *Case Study and Methods*) are the sum of the NO<sub>2</sub> directly removed by the forest, and the NO<sub>x</sub> indirectly removed in the form of O<sub>3</sub> formed from NO<sub>x</sub> and VOC. Because trees do not remove, but rather emit, VOCs (28), a reforestation project can only achieve removal of VOC equivalents (VOCe) if it removes more O<sub>3</sub> than forms from its VOC emissions. Whether the O<sub>3</sub> removed by the forest is equivalent to NO<sub>x</sub> or VOC abatement depends on whether O<sub>3</sub> formation in the area is predominantly NO<sub>x</sub> or VOC limited, respectively. Thus, depending on its location, a reforestation project may generate either only NO<sub>x</sub>e or both NO<sub>x</sub>e and VOCE abatement.

Model scenarios with different planting densities and stock sizes (*SI Appendix, Table S1*) identified seedlings planted at 1,500/ha as the planting design with the lowest cost per ton of NO<sub>x</sub>e removed. Unless indicated otherwise, all results presented below refer to this design, which achieves maximum forest crown area in year 23 after planting (*SI Appendix, Fig. S2*).

### Precursor and Ozone Removal, Carbon Storage, and VOC Emissions.

Based on our modeled forest structure and UFORE-estimated specific pollutant removal rates (Table 1), the reforestation project is estimated to remove a total of 309.7 t O<sub>3</sub> and 58.1 t NO<sub>2</sub> over our 30-y analysis period and store 24,574 t above-ground C at year 30. Annual air pollution removal was greatest in year 23 of the project at maximum canopy cover (O<sub>3</sub>: 14,156 kg; NO<sub>2</sub>: 2,659 kg), and C sequestration was highest in year 16 (1,153 t). For O<sub>3</sub> and NO<sub>2</sub>, the predicted decline in annual removal rates

**Table 1. Modeled forest structure parameters and specific pollutant removal for the 405-ha reforestation project**

	Phase 1 (DBH < 12.7 cm)	Phase 2 (DBH ≥ 12.8 cm)
Forest structure parameter		
Number of trees	502,698	94,440
Average DBH (cm)	7.6	25.4
Average tree height (m)	7.8	15.9
Average crown height (m)	3.1	6.4
Average crown width (m)	3.0	7.1
UFORE modeled forest structure		
Average LAI	3.17	3.42
Total leaf area (m <sup>2</sup> )	1,861,620	2,298,500
UFORE modeled air quality effects—Pollution removal		
NO <sub>2</sub> (g/m <sup>2</sup> crown area/y)	0.579	0.600
O <sub>3</sub> (g/m <sup>2</sup> crown area/y)	3.116	3.194

Modeling phases: 1—tree establishment (years 1–10) and 2—maturity (years 11–30).

after year 23 is due to the omission of natural forest regeneration from our model, making our removal estimates conservative.

At our estimated removal rates per hectare, reforesting half or all of the estimated potentially reforestable 189,400 ha of bottomland habitat in the HGB area (*SI Appendix, Fig. S3*) would abate an estimated average 2,426–4,852 t O<sub>3</sub> and 455–911 t NO<sub>2</sub> per year over 30 y. Using the reported HGB area, O<sub>3</sub> production efficiency envelope of NO<sub>x</sub> of 3–8 (38, 39), this results in an estimated 995–1,641 to 1,990–3,282 t NO<sub>x,e</sub> removal per year, or ~1.7–5.5% of the average of the estimated 2006 and 2018 annual HGB-area NO<sub>x</sub> point-source emissions of 59,700 t (40). While these abatement levels by themselves likely would not be sufficient to achieve attainment, they do amount to several percent of the additional abatement that may be needed (*SI Appendix, section S3*).

Estimated 30-y total VOC emissions of the cost-effective (1,500 seedlings/ha) planting scenario are 115 t of isoprene, 41 t of monoterpenes, and 197 t of other VOC.

#### Cost Effectiveness of Reforestation for Ozone Precursor Abatement.

We compared reforestation and conventional controls in terms of 30-y present value (PV) cost per ton NO<sub>x,e</sub> removed. All cost-effectiveness estimates assume full provisional up-front credit of pollutant removal as per EPA guidance (41) and that our projections represent actual forest growth during the 30-y analysis period.

**NO<sub>x</sub>—land cost scenario 1: No land costs.** The reforestation project would remove an estimated total of 209 or 127 t NO<sub>x,e</sub> over the 30-y analysis period in the “high” and “low” removal scenarios, respectively (incremental O<sub>3</sub> production efficiencies of NO<sub>x</sub> = 3 and 8, respectively). On lands where reforestation does not incur land costs (*Private Land Opportunity Costs*), this translates into approximately \$1,680–\$3,210/t NO<sub>x,e</sub> (high removal) and \$2,770–\$5,300/t NO<sub>x,e</sub> (low removal), with the low and high values in each range resulting from low and high planting cost estimates, respectively (*SI Appendix, Table S4*). If CARB carbon offset revenues are included at \$12.25/t carbon dioxide equivalent (CO<sub>2,e</sub>), the cost-effectiveness of NO<sub>x,e</sub> removal improves substantially, to \$300–\$1,840 and \$500–\$3,030/t in the high and low removal scenarios, respectively. These latter ranges reflect expected revenue (\$372,400) and transaction costs (\$103,400) of carbon offsets (both PV at 7% discount rate) over the 30-y time horizon. Ozone accounts for 39% and 63% of the total removed NO<sub>x,e</sub> in the low and high removal scenarios, respectively.

In all but the low removal scenario—where a project receives less NO<sub>x</sub> credit for the O<sub>3</sub> removed due to high O<sub>3</sub> production

efficiency of NO<sub>x</sub>—reforestation in the HGB area has a lower mean cost per ton of NO<sub>x,e</sub> removed than conventional NO<sub>x</sub> controls (\$2,500–\$5,000/t) and NO<sub>x</sub> allowances (\$3,300/t; *SI Appendix, section S5*); that is, permits granting a perpetual right to emit 1 t NO<sub>x</sub> per year (Fig. 1). Because reforestation projects would be expected to be sited in areas where they generate the most precursor abatement per dollar, we expect our high-removal scenario estimates—\$1,680–\$3,210/t without, \$300–\$1,840/t with C offsets—to be more representative of actual projects. Importantly, contract bids would provide ex ante certainty for actual projects about planting, and thus project cost and cost per ton of NO<sub>x</sub> removed.

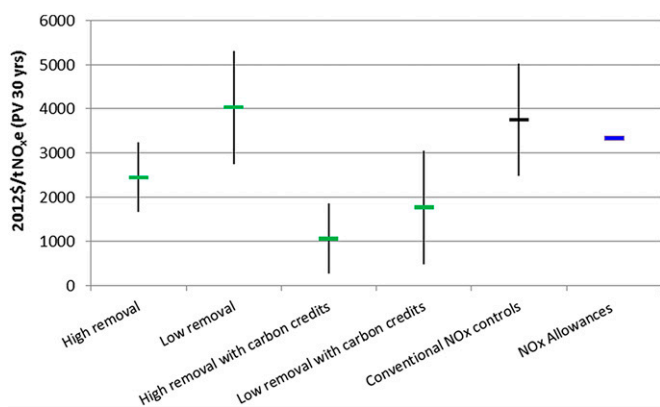
**NO<sub>x</sub>—land cost scenario 2: \$4,940/ha.** At a representative price of suitable lands in the study area of \$4,940/ha (*Case Study and Methods*), total planting-related project cost is more than 5 times the average cost (\$493,000; range \$333,000–\$654,000) in land cost scenario 1. As a result, reforestation is no longer cost competitive with engineering-based precursor abatement, even with C offsets (*SI Appendix, Fig. S4*).

**VOC.** A reforestation project could achieve net VOCe removal if it removed more O<sub>3</sub> than is formed from its VOC emissions. Because our case study project is not located in an area where O<sub>3</sub> formation is VOC limited, its O<sub>3</sub> removal is equivalent to abatement of NO<sub>x,e</sub>, not VOCe. Nevertheless, to assess whether the project might abate VOCe if it were located in VOC-limited portions of the Houston area (Galveston Bay or southern Harris County; ref. 42) we estimate its net O<sub>3</sub> balance in a VOC-limited area.

Using the maximum incremental reactivity (MIR; ref. 43) scales (gram O<sub>3</sub>/gram VOC) to estimate the quantity of O<sub>3</sub> formed by its VOC emissions, the project would produce an estimated 1,650 t O<sub>3</sub> over the 30-y analysis period, or over 5 times the estimated 310 t O<sub>3</sub> removed via dry deposition (*SI Appendix, sections S6 and S7*). The MIR scale represents relatively high NO<sub>x</sub> conditions (areas with high NO<sub>x</sub>:VOC ratios like Galveston Bay or southern Harris County in the HGB area; ref. 42), where O<sub>3</sub> is most sensitive to changes in VOC emissions, and is most often used or proposed for use in regulatory applications (43). Other reactivity scales (43) yield lower O<sub>3</sub> production estimates that however still exceed estimated removal via dry deposition. Although these findings are based on simplified models and assumptions and should be considered preliminary given the complex nature of O<sub>3</sub> formation (44), they suggest that reforestation projects may not yield net VOC abatement.

**Private and Social Value of Carbon Sequestration.** The project generates C offsets with an expected net PV of \$269,000 and avoids social costs of carbon (SCC)—the sum of future damages from increased atmospheric CO<sub>2</sub> concentrations—with an estimated PV of \$1.96 million (3% pure rate of time preference [PRTP]) to \$3.25 million (2.5% PRTP). On public or private lands with a qualifying conservation easement, lower mandatory contributions to the CARB forest offset project buffer account (12% of calculated sequestration vs. 19% in our estimates) would increase net present offset value by 12%. We consider both private and social C value estimates conservative because of the exclusion of natural regeneration in our forest model, which reduces estimated C sequestration after year 20.

**Sensitivity Analysis.** Engineering-based NO<sub>x</sub> control options have an average economic lifetime of only 20 y (45). Full replacement of NO<sub>x</sub> control equipment in year 21, for example, would increase the abatement cost per ton of engineering-based controls by 18% (assuming annual maintenance and repair costs for chemical processes of 3% [ref. 46] of initial capital cost and a 7% discount rate; *Case Study and Methods*). Extending the time



**Fig. 1.** Average cost per ton of NO<sub>x</sub> control through reforestation at zero land cost, for high and low removal scenarios, and cost of standard point source controls and NO<sub>x</sub> stream allowances in the HGB area. Vertical lines indicate ranges caused by different cost assumptions. Costs expressed as PV over 30-y period. NO<sub>x,e</sub>—NO<sub>x</sub> equivalent.

horizon of the analysis also would increase the competitiveness of reforestation with conventional controls, as the established forest stand will keep removing O<sub>3</sub> and NO<sub>2</sub> at no additional cost beyond year 30. Thus, reforestation may be even more competitive with conventional NO<sub>x</sub> controls than our analysis suggests.

Our finding that purchasing bottomland hardwood forest habitat for reforestation would not generate cost-competitive NO<sub>x</sub> abatement is based on the assumption that emitters would fully absorb these costs. That may not necessarily be the case. Reforestation of former bottomland hardwoods can generate cobenefits, making cost-share arrangements likely for reforestation projects on some lands. Local and national stakeholders, such as the US Fish and Wildlife Service, the Texas chapter of the Nature Conservancy, and other conservation organizations in our study area have expressed strong interest in such cost-share arrangements for reforestation of ecologically valuable bottomlands. Whether cost sharing for purchased lands yields cost-competitive NO<sub>x</sub> removal through reforestation of pasture lands depends on the cost shares. For example, at \$4,940/ha land cost, cost sharing at 3:1 (project partner:point source) would achieve \$2,690/t NO<sub>x,e</sub> in the best case (high removal, low planting cost plus C offsets) scenario, which would be competitive.

The competitiveness of reforestation projects may also be enhanced by interplanting of fast-growing species such as *Populus deltoides* in our study area to support a one-time selective timber harvest (47). The impact of such harvests on the cost per ton of NO<sub>x,e</sub> reduced depends on timber prices, volume harvested, accessibility, distance to the nearest mill, and requirements of the relevant C offset protocols.

The approach we present simplifies several complex biophysical processes. First, we were unable to find growth-mortality rates and allometric equations for seedlings and saplings of our bottomland hardwood species. Therefore, we assumed that our rates and equations can be applied to trees with a diameter at breast height (DBH) < 7.6 cm. Second, by not accounting for natural regeneration (47, 48) we underestimate total leaf area and air pollution removal during the later years of our analysis period. Likewise, our assumption that 2009 pollutant levels will remain constant may bias our estimates downward (*SI Appendix, section S7*). Third, we do not model effects of stochastic disturbance events like drought, wildfire, pests and diseases, or hurricanes in our analysis. However, our mortality rates are from a Houston study (49) covering an 8-y period that included a hurricane landfall (Hurricane Ike). Thus, they reflect recent historic pest and disease induced mortality, and implicitly assume an annual hurricane landfall probability of 1/8, although historic

probability for our site (Brazoria and Fort Bend Counties) is <1/25 (50). Moreover, bottomland hardwoods are less prone to hurricane damage than mature, pine-dominated forest stands with large open-grown trees (49, 51). Southern bottomland forests also are less susceptible to drought than upland forests and have a low fire frequency (52), with fire risk for bottomland hardwoods in our area classified as very low (53). Overall, aggregate tree mortality risk from all pests and diseases in 2013–2027 is an estimated 1–5% at our study site and 1–15% for other bottomland forests in the area (54). The assumed 5.1–12% annual mortality rates for our project (*SI Appendix, Table S2.5*) thus exceed the combined risk from all these stochastic disturbance events. If disturbance risks were unknown or an additional margin of safety sought, a SIP could specify that a portion of estimated pollutant removal be deposited in a programwide risk buffer account, as is done in forest C offset protocols (35).

Another limitation is the use of pollution and weather data from fixed stations not located on our project site. Other studies on the effects of peri-urban forests on O<sub>3</sub> have used regional weather and chemistry models [e.g., Weather Research and Forecasting (WRF)-Chem; ref. 32]. The high sensitivity of the O<sub>3</sub> concentrations predicted by WRF-Chem and other advanced air quality models to meteorological conditions and the difficulty of accurately specifying those conditions for complex coastal zones make application of those models in the HGB area challenging (22, 55, 56).

Finally, the treatment of any net VOC emissions from a reforestation project will affect the overall cost effectiveness of such projects for O<sub>3</sub> precursor control, and hence their implementation. To promote reforestation for O<sub>3</sub> abatement without sacrificing air quality goals, it would seem appropriate to not debit a project with its VOC emissions as long as those emissions are unlikely to lead to O<sub>3</sub> formation. Such a treatment would be justified on the basis that it does not conflict with the defined policy (SIP) goal of reducing O<sub>3</sub> concentrations, but rather promotes adoption of novel O<sub>3</sub> control measures that would achieve additional O<sub>3</sub> abatement beyond that achieved by legally mandated control technologies. Application of such a differentiated treatment of VOC emissions and accurate conversion of the O<sub>3</sub> removed by a reforestation project to NO<sub>x,e</sub> both require reasonably reliable spatial information on the type (NO<sub>x</sub>, VOC) and degree (i.e., production efficiency) of sensitivity of O<sub>3</sub> formation to ambient precursor concentrations, and can significantly affect the siting of reforestation projects and their cost per ton of precursor removal (*SI Appendix, section S8*).

## Conclusion

Our analysis indicates that reforestation could be a viable, novel approach for abating ground-level O<sub>3</sub> pollution that complements conventional technology-based controls. Including reforestation in a comprehensive control strategy is desirable for regulators because it furthers attainment beyond what is achievable with current approaches considered technically or economically feasible. It is also desirable for regulated emitters because it may lower their compliance costs, in part due to the uniquely scalable nature of reforestation that contrasts with the lumpy costs and abatement provided by technological controls. We expect that reforestation in the Houston O<sub>3</sub> nonattainment area would be cost competitive with additional conventional point source NO<sub>x</sub> controls on lands where it has negative or negligible opportunity cost for landowners and thus does not incur land costs. It may even be cost competitive on many additional lands where it does incur opportunity costs. These lands may need to be purchased, but acquisition costs may be defrayed through suitable timber harvests, or private or public cost-share agreements motivated by not only the high conservation value of those lands (57), but additional water quality (58), and recreation and scenic benefits restored forests may provide once established and mature (59, 60).



In this paper we present, to our knowledge, a first attempt at constructing a methodology for integrating reforestation into  $O_3$  control efforts. Making such integration a reality requires additional work, above all of the development of specific regulatory guidance that addresses several of the uncertainties outlined in our analysis, in particular the ozone production efficiency of  $NO_x$ ;  $NO_x$  vs. VOC limitation of ozone production; and the portion of  $NO_x$  abatement to be deposited in a SIP “buffer” account to hedge against catastrophic disturbance events, if any. In some cases, removing or reducing these uncertainties may require additional research.

Large-scale reforestation for  $O_3$  abatement generally should be limited to  $NO_x$ -sensitive environments, as additional forest cover may increase  $O_3$  levels in situations where  $O_3$  formation is VOC limited (30, 44, 64, 65). It is important to note that with the exception of the urban cores of large metropolitan areas, many portions of  $O_3$  nonattainment and maintenance areas in the United States (19, 66) and elsewhere (61, 62) are characterized as  $NO_x$  sensitive. Use of reforestation for  $O_3$  abatement therefore may have widespread applicability. Our findings suggest that reforestation for air pollution abatement constitutes a potentially globally applicable example of “natural infrastructure” solutions to environmental challenges (e.g., 67, 68).

### Case Study and Methods

**Case Study Site.** The eight-county HGB area has an average annual temperature between 15 and 20 °C (60 and 70 °F) and annual precipitation averages 1,020–1,530 mm (40–60”) (69). It lies primarily in the Gulf Coast prairies and marshes ecoregion, and partially in the upper west gulf coastal plain ecoregion (70), and contains the eastern two-thirds of the 515,000-ha Columbia Bottomlands Conservation area extending from the Gulf Coast inland along the Brazos, Colorado, and St. Bernard Rivers (57). Out of a total of ~189,400 ha potentially available for reforestation to bottomland hardwoods (SI Appendix, Fig. S3), we selected a 404.69-ha reforestation site located just north of Brazos Bend State Park (29°24'54"N, 95°33'51"W) expected to achieve high  $O_3$  and  $NO_2$  removal (SI Appendix, section S7). Historically forested, the site is now primarily grassland with sparse tree and shrub cover. Fig. 3 shows the HGB area, the study site, and the Texas Commission on Environmental Quality (TCEQ) monitoring stations from which pollution and meteorological data were obtained.

Forests in the HGB area have been declining primarily due to anthropogenic disturbances including timber harvest, agriculture, and urbanization (49, 51, 71). Bottomland forests in the region have experienced particularly high losses (72). Although this trend is predicted to continue in a business-as-usual scenario, improved forest protection could reduce future net forest loss (73), and coupled with large-scale reforestation might even reverse the long-term trend of declining forest cover. Bottomland forests in the HGB area have a high potential for restoration (57), are less susceptible to drought and wildfire than upland forests (49, 51, 52), and have high biodiversity and recreation value (57, 72). Thus, we chose a bottomland site for our analysis.

**Selection of Suitable Reforestation Sites and Silvicultural Criteria.** Planting site selection for  $O_3$  and precursor control purposes should maximize pollutant removal per unit cost. Analysis of emissions, ambient concentrations, and wind data suggests that reforestation would remove the most  $O_3$  in southern and southwestern Harris and in Brazoria County, and  $NO_2$  removal would be highest in, and downwind of, the downtown Houston area and northwest of major  $NO_x$  point sources along the Gulf Coast (SI Appendix, section S7). Siting depends on whether  $NO_x$  or VOC abatement is prioritized (SI Appendix, section S8). We maximize  $NO_x$  abatement by locating the study site south of Houston in an area already high in biogenic VOC emissions (74) where  $O_3$  formation is expected to be mostly  $NO_x$  limited—as it is in most of the HGB area (figure 2b in ref. 42) especially during late morning to late afternoon (42, 62) when biogenic VOC emissions are highest. Thus, VOC emissions from the additional forest are unlikely to lead to additional  $O_3$  formation that would reduce the  $O_3$  net balance of the project. Our case study site is located in Brazoria County (Fig. 3), just downwind of major industrial  $NO_x$  sources along the Gulf Coast and in the path of high  $NO_x$  concentrations and  $O_3$  plumes drifting southwest over the Houston metro area (SI Appendix, section S7).

Our hypothetical project is a contiguous forest in a peri-urban area characterized by suitable soil and site properties. Such a design is expected to require less maintenance, be self-regenerating, and achieve lower mortality, allowing planting of smaller trees that in urban areas would be more prone to accidental or vandalism-related damage. This minimizes costs and emissions from maintenance, replacement, and monitoring activities (75, 76). We minimize VOC emissions by avoiding the planting of high-VOC-emitting species (SI Appendix, section S6).

**Maximizing Precursor Abatement:  $NO_x$  vs. VOCs.** Ozone and  $NO_2$  removal rates by trees generally increase with pollutant concentrations (29). Although very high pollutant concentrations can reduce photosynthetic rates

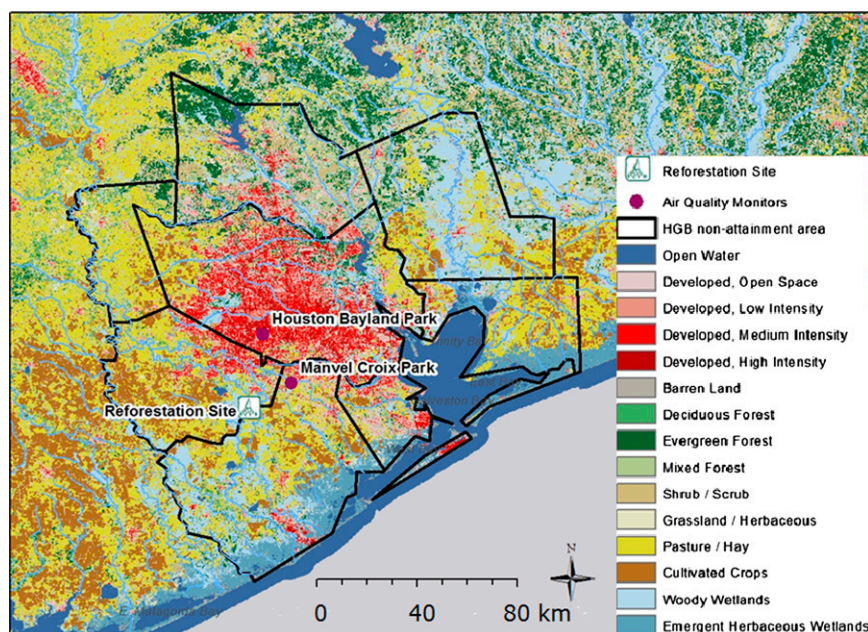


Fig. 3. Map showing the HGB nonattainment area, the reforestation site, Texas Commission on Environmental Quality air quality monitoring stations from which data were obtained, and land cover.

or makes trees more susceptible to other stressors such as insect or pathogen attack (77, 78), exposure levels to O<sub>3</sub> in the HGB area are not within the range that is expected to result in injury to shrubs and trees (79). We therefore assume that reforestation will achieve the highest O<sub>3</sub> and NO<sub>2</sub> removal rates in areas with the highest O<sub>3</sub> and NO<sub>2</sub> concentrations, respectively.

Nitrogen dioxide removal by forests—appropriately adjusted for uncertainties—can be translated directly into NO<sub>x</sub> removal a reforestation project could claim. The calculation of the precursor removal a project could claim for the O<sub>3</sub> it removes is less straightforward. Ozone removal by forests is equivalent to avoiding emission of the precursor quantities used up in the formation of the removed quantity of O<sub>3</sub>. Converting O<sub>3</sub> removed into its equivalent quantities of avoided precursors requires information on the O<sub>3</sub> production efficiency in the area, that is, the number of O<sub>3</sub> molecules formed from a precursor molecule (80).

Results from simulations (38) and measurement flights (39) indicate an O<sub>3</sub> production efficiency envelope of 3–8 in the HGB area for NO<sub>x</sub>; that is, 3–8 O<sub>3</sub> molecules are formed per NO<sub>x</sub> molecule oxidized during midday hours, making removal of an O<sub>3</sub> molecule equivalent to removal of 1/3–1/8 of a NO<sub>x</sub> molecule under NO<sub>x</sub>-limited conditions. We use this 3–8 range to develop high and low removal cases, respectively, for our reforestation project.

With O<sub>3</sub> formation in our study and much of the HGB area characterized as NO<sub>x</sub> limited, we convert O<sub>3</sub> removed by the project to NO<sub>2</sub> equivalents (NO<sub>2</sub>e) using the molecular weights of the two compounds and the 3–8 range of O<sub>3</sub> production efficiencies of NO<sub>x</sub> (i.e., 1 O<sub>3</sub> = 0.32–0.12 NO<sub>2</sub>). Because point sources are regulated on emissions of NO<sub>x</sub> (most of which are immediately converted to NO<sub>2</sub> in the atmosphere) and not NO<sub>2</sub>, we follow existing policy guidance and convert NO<sub>2</sub>e to NO<sub>x</sub>e using the national NO<sub>2</sub>:NO<sub>x</sub> default ratio of 0.75 (81), which closely matches observed ratios of 0.73–0.74 in the HGB area (82).

**Time Horizon of Analysis and Discount Rates.** Both time horizon and discount rate affect cost-effectiveness estimates. We use a 30-y time horizon that somewhat exceeds the average 20-y lifetime of conventional NO<sub>x</sub> control equipment (45) and the 25-y crediting period used for CARB forest carbon offset projects (35). Although the planted forest is expected to survive past 30 y and its cost-effectiveness increases with the time horizon, uncertainty as to future O<sub>3</sub> levels (and thus biogenic removal rates) and changes in the regulatory framework argue against much longer timeframes.

We use a 7% discount rate to calculate the PV of future project costs (SIP and C offset reporting and monitoring) and of C offset revenues, which “approximates the marginal pretax rate of return on an average investment in the private sector” (ref. 83, p. 9; *SI Appendix, Table S9*). We discount future SCC estimates using 2.5% and 3% discount rates, respectively (*Avoided Social Cost of Carbon*).

**Reforestation and Silvicultural Characteristics.** Rosen et al. (72) report that eight native tree species (*Carya aquatica*, *Celtis laevigata*, *Fraxinus pennsylvanica*, *Quercus nigra*, *Q. texana*, *Q. virginiana*, *Ulmus americana*, *U. crassifolia*) accounted for 87% of overstory basal area in a mature, protected bottomland hardwood forest in southwestern Brazoria County (*SI Appendix, Table S2.1*). Online reviews and phone inquiries with regional tree nurseries showed that six of these species (all but *C. laevigata* and *C. aquatica*) were commercially available in a variety of planting stocks and sizes. We selected these six species for planting and assume a constant forest composition (*SI Appendix, Table S2.2*) during our analysis period and no natural regeneration or introduction of any additional species.

We assessed possible climate change effects on our selected species using the *Climate Change Atlas for 134 Forest Tree Species* that predicts tree distribution ranges based on the random forests model (84, 85). Specific climate, topographic, soil, and land use parameters used in the random forest model and modeling methods are detailed in ref. 84. Our analysis suggests that habitat suitability in the bottomland forest areas in the HGB area will remain generally unchanged for the planted species during our time horizon (*SI Appendix, Table S2.3*).

Initial tree planting density and size affect overall pollutant removal and project costs and thus are key project design and modeling parameters. Recommended initial planting density for bottomland hardwood forests ranges from 400 to 3,000 plants per hectare (*SI Appendix, Table S2.4*). For this study we used 730 seedlings per hectare as a base case (47), based on recommendations for ecological restoration of bottomland hardwood forests in this region (86).

The size of tree planting stock, in terms of DBH or caliper (diameter at 61 cm above ground), is the dominant driver of reforestation costs. High-quality planting stock, and in general larger planting material, have overall higher survival rates (75) and thus will achieve a given age-specific target

density with a lower initial planting density. Tree size also is positively related to crown area (87)—a key air pollution removal variable—and thus pollutant removal per tree is greater for larger trees in any given year. However, tree stock and planting costs also increase with tree size. The most cost-effective size at planting depends on all of these factors. Much of the scientific literature examining bottomland reforestation focuses on seedlings (*SI Appendix, Table S2.4*), which may be the most cost-effective size for adding trees in open, unpopulated areas (75). We modeled different tree size classes and planting densities to identify the cost-effective planting design for O<sub>3</sub> and precursor abatement.

**Project Costs.** Our SIP reforestation project costs comprise of (i) planting-related costs for project design, planting stock, site preparation and maintenance; (ii) land opportunity costs from forgone value of displaced land uses; and (iii) transaction costs for legal or coordination activities with regulatory bodies or third parties for initial project approval and for monitoring and reporting for SIP and carbon offset compliance (verification and registration) purposes, if any.

**Planting-Related Costs.** We obtained cost estimates for planting stock, site preparation, and labor (*SI Appendix, Table S1*) from the literature and regional providers (75, 88). Recommended site preparation techniques for hand and machine planting styles comprise reduction or elimination of weeds through prescribed fire, mowing, or double disking (89). Given expected constraints on the use of prescribed fire in our study area due to potential effects of smoke on air quality, we assumed site preparation by mowing and double disking.

For tree seedlings and planting costs, our low estimate combines a stock cost estimate of \$0.24 per seedling and Texas Forest Service per-acre cost estimates for hand planting (\$75) and mowing (\$33) as recommended for this planting option (89). Our high estimate uses Texas Forest Service estimates for hardwood seedling costs (\$0.60 each) and per-acre costs for planting by wildland machine (\$85) and double disking (\$115) as recommended for this planting option (89). We assume hand and machine planting costs per acre are for commonly used planting densities like those reported in Stanturf et al. (47, 86), and scale these costs proportionally for higher planting density scenarios.

**Private Land Opportunity Costs.** Land opportunity costs are highly dependent on the value of displaced, incompatible uses by owners. Not all uses are incompatible with reforestation—including timber production (48, 86), recreation opportunities or visual amenities for people living in the viewshed of reforested lands (59, 60), although the latter two are generally lower for peri-urban reforestation sites due to their initially dense, low-height structure and their location away from populated areas. Reforestation opportunities exist on public, deforested bottomlands currently under shrub or grass cover, and on large tracts of converted former bottomland hardwoods owned by companies with several large point sources in the area on which reforestation would not displace current or anticipated future high-value uses, thus incurring negligible land opportunity costs. Private third-party land owners might also be willing to have their bottomlands reforested for free or for a charge. With most converted former bottomlands currently in agriculture, we bracket potential opportunity costs by using two estimates: (i) zero cost; and (ii) \$4,940/ha—the approximate average fee-simple cost for nonwaterfront agricultural land in bottomland habitat in the area (*SI Appendix, section S1*).

**Third-Party Land Opportunity Costs.** Reforestation may impose costs on third parties by restricting the supply of developable land. Because of its small size and location away from urban expansion, this is not a concern for our case study project. It also generally is less of a concern for peri-urban reforestation. Furthermore, any development-related opportunity costs would be at least partially offset by property value premiums and nonmarket benefits owners of properties near reforested lands would receive (59, 60).

**Transaction Costs.** To satisfy SIP-related monitoring requirements (41), we assume that a site analysis (\$2,000 each) will be carried out every 3 y to assess whether tree survival and growth (key drivers of pollutant removal) match model predictions. We also assume an initial site inventory (\$10,000). If the project registers for CARB C offsets (35), the more demanding CARB inventory would be substituted for the SIP inventory. In the case of C offset generation, we assume an initial complete forest inventory (\$20,000) followed by full verifications (\$20,000 each) in years 14, 20, and 26, and annual interim verification and data reports (35) (\$5,000 each). All offset

cost estimates are based on The Nature Conservancy data for California Climate Action Reserve integrated forest management offset projects.

**Forest Canopy Modeling.** We modeled tree canopy cover at annual time steps during the analysis period based on a simplified growth-survivability model and a tree stem-crown area allometric equation (49, 87, 90). We determined the cost-effective planting density and tree size by modeling O<sub>3</sub> and NO<sub>2</sub> removal and project costs for planting densities between 730 and 1,500 stems per hectare, for seedling and larger planting stock. We estimated the number of surviving trees (ST) during the 30-y analysis period from initial tree populations and DBH size-class-based growth and mortality rates (*SI Appendix, Table S2.5*) for Gulf Coast peri-urban forests (49, 90). These rates account for hurricane-related mortality. The annual tree crown width (CW) increments were then estimated using the following DBH-based allometric equation for Alabama *Quercus spp* (87):

$$CW = -0.8941 + 0.515 \text{ DBH} + 0.0059(\text{DBH})^2. \quad [1]$$

Crown area (CA) for each year was calculated as  $CA = \pi \times (CW/2)^2$  for each individual tree and summed over ST. Natural regeneration or stand clearing disturbance events were not separately accounted for in our 30-y analysis period.

**Air Pollution Removal, VOC Emission, and Carbon Sequestration Modeling.** We used the UFORE model (UFORE-ACE version 6.5 with U4D020701.SAS and U4B020700 modules; ref. 76) and hourly pollution concentration and meteorological data from January 1 to December 31, 2009 (*SI Appendix, Table S7.1*) to estimate the annual removal of NO<sub>2</sub> and O<sub>3</sub> and VOC emissions per unit area of surviving tree cover using UFORE-estimated forest structure parameters (Table 1). Hourly pollutant concentrations and solar radiation data are from TCEQ's Manvel Croix Park C84 (29°31'–41°N, 95°23'–29°W) and Bayland Park (29°41'45"N, 95°29'57"W) monitors, respectively, and meteorological data (wind direction and speed, temperature, dew point, atmospheric pressure, precipitation, and sky cover) from the National Oceanic and Atmospheric Administration National Climatic Data Centers' Pearland station (29°31'1"N, 95°15'0"W). We assume that 2009 O<sub>3</sub> concentrations and meteorological parameters will remain constant over the 30-y analysis period (*SI Appendix, section S7*). Also, rather than model required UFORE structural parameters and annual air pollution removal and VOC emissions for every year in our 30-y analysis period, we use a simplified approach and model two representative urban forest structural phases (Table 1) and respective air quality effects to obtain necessary modeling parameters.

Our first modeling period represents the stand establishment phase at around year 3 of the project [Phase 1; leaf area index (LAI) = 3.17 and DBHs < 12.7 cm]. Our second modeling period characterizes a maturing stand at about year 20 to represent years 11–30 (Phase 2; LAI = 3.42 and DBHs ≥ 12.7 cm). To simplify our biophysical modeling, we assume that seedlings without a DBH will have similar growth and mortality rates as those trees with a DBH < 7.6 cm. Further, we assume that because of appropriate site preparation, planting criteria and monitoring, planting stock will become established within the first year of planting. Additional required modeling parameters were set at these values: Leaf dieback and missing crown, 80% and 10%, respectively, for both phases; tree condition = good, and percent crown present = 75, for both phases; and crown light exposure of 3 and 5, for phases 1 and 2, respectively.

To simplify our biogenic emission modeling, we estimate phase 1 and phase 2 VOC emissions for a growth-mortality modeled tree population of 244,456 (year 3) and 56,214 (year 20), respectively, for a reforestation project

with an initial planting density of 730/ha (*SI Appendix, Table S6.2*). Although leaf biomass is usually the key forest structural parameter used for biogenic emission modeling (44, 64), due to the lack of leaf biomass allometric equations for our species we assume a linear relationship between leaf biomass and leaf area and develop leaf area-scaled VOC emission estimates (kilograms of VOC per square meter of tree cover) for other planting densities using modeled tree cover, DBH-based growth-mortality model estimates and the crown area equation (87), and VOC emission results from phases 1 and 2 (*SI Appendix, section S6*). Detailed UFORE modeling methods and assumptions can be found in refs. 32 and 76.

Finally, due to a lack of species- and region-specific allometric equations, we estimate individual tree aboveground carbon (AGC) storage using a composite equation for mixed hardwoods (91):

$$AGC = \exp(-2.48 + 2.4835 \times \ln(\text{DBH})). \quad [2]$$

We derived annual net carbon sequestration as the year (t)-on-year difference in carbon storage during our analysis period,  $AGC_t - AGC_{t-1}$ .

**Carbon Offsets.** Reforestation projects in the United States are eligible for generating carbon offsets under California's US Forest Projects Offset Protocol if they are additional—that is, not otherwise required by law, regulation, or any legally binding mandate applicable in the offset project jurisdiction, and would not otherwise occur in a conservative business-as-usual scenario (35). Unlike the regulatory-prescribed conventional air pollution control technologies included in SIPs, emerging SIP measures like reforestation are not mandatory. Thus, we expect a reforestation project to meet the additionality requirement, except in cases where its cost effectiveness exceeds that of conventional control approaches sufficiently to make its implementation clearly profitable even without offsets.

We use the March 2012 price of \$12.25/t CO<sub>2</sub>e (\$45/t C) for December 2013 forward contracts for guaranteed California Compliance Offset Credits (92) to calculate expected offset revenue. Offset prices have increased slightly during January 2012–January 2013 (93, 94), but we assume real offset prices will remain unchanged during our analysis period—likely a conservative assumption given the large current and expected future supply shortfall (95). We estimate offset quantity as aboveground tree carbon sequestered by the project minus the maximum 19% mandatory contribution to the CARB forest offset project buffer account (using US default fire risk of 4%; ref. 35).

**Avoided SCC.** Estimates of the SCC—the total value of the sum of future damages from a 1-t increase in atmospheric CO<sub>2</sub> concentrations—vary widely (96). We use the two “middle” SCC estimates developed by the federal Interagency Working Group on Social Cost of Carbon (97), which represent the averages of the damage estimates produced by the Dynamic Integrated Climate-Economy; Policy Analysis of the Greenhouse Effect; and Framework for Uncertainty, Negotiation and Distribution models (97) for discount rates (PRT) of 3% and 2.5%. The SCC present value (2012) used to value aboveground carbon sequestered by our reforestation project declines from \$24.7 (3% PRT) and \$40.1 (2.5%) respectively per ton of C in 2012 to \$18.4 and \$31.4/t in 2042 (*SI Appendix, Table S10*).

**ACKNOWLEDGMENTS.** We thank Mark Estes, Jeff Weigel, John Herron, Bryan Duncan, Tom Smith, and Mike Lange; and three anonymous reviewers whose comments greatly improved the paper. This analysis was done as part of the collaboration between The Nature Conservancy and Dow Chemical Company and was funded by the Dow Chemical Company Foundation.

- Sillman S (1999) The relation between ozone, NO<sub>x</sub>, and hydrocarbons in urban and polluted rural environments. *Atmos Environ* 33(12):1821–1845.
- Royal Society (2008) *Ground-Level Ozone in the 21<sup>st</sup> century: Future Trends, Impacts and Policy Implications*. (Royal Society, London) Science Policy report 1508.
- World Health Organization (2006) *Air Quality Guidelines: Global Update 2005, Particulate Matter, Ozone, Nitrogen Dioxide and Sulphur Dioxide* (WHO Regional Office for Europe, Copenhagen).
- US Environmental Protection Agency (2013) *Integrated Science Assessment for Ozone and Related Photochemical Oxidants*. (US EPA, Research Triangle Park, NC) EPA 600/R-10/076F.
- Jerrett M, et al. (2009) Long-term ozone exposure and mortality. *N Engl J Med* 360(11):1085–1095.
- Brunekeef B, Holgate ST (2002) Air pollution and health. *Lancet* 360(9341): 1233–1242.
- Anderson GB, Krall JR, Peng RD, Bell ML (2012) Is the relation between ozone and mortality confounded by chemical components of particulate matter? Analysis of 7 components in 57 US communities. *Am J Epidemiol* 176(8): 726–732.
- Orru H, et al. (2013) Impact of climate change on ozone-related mortality and morbidity in Europe. *Eur Respir J* 41(2):285–294.
- Lim SS, et al. (2012) A comparative risk assessment of burden of disease and injury attributable to 67 risk factors and risk factor clusters in 21 regions, 1990–2010: A systematic analysis for the Global Burden of Disease Study 2010. *Lancet* 380(9859): 2224–2260.
- Graff Zivin JS, Neidell MJ (2012) The impact of pollution on worker productivity. *Am Econ Rev* 102(7):3652–3673.
- Berman JD, et al. (2012) Health benefits from large-scale ozone reduction in the United States. *Environ Health Perspect* 120(10):1404–1410.
- Ashmore MR (2005) Assessing the future global impacts of ozone on vegetation. *Plant Cell Environ* 28(8):949–964.
- Emberson L, Ashmore M, Murray F, eds (2003) *Air Pollution Impacts on Crops and Forests: A Global Assessment. Air Pollution Reviews* (Imperial College Press, London), Vol 4.
- Reitze AW, Jr (1999) The legislative history of U.S. air pollution control. *Houst Law Rev* 36:679–743.
- Gill GC, Bierly EW (1963) A meteorologically operated stack control system. *J Appl Meteorol* 2(4):431–439.



16. Whiteman CD (1982) Breakup of temperature inversion in deep mountain valleys: Part I. Observations. *J Appl Meteorol* 21:270–289.
17. National Research Council (1975) *Air Quality and Stationary Source Emission Control* (The National Academies, Washington, DC).
18. de Nevers N (1995) *Air Pollution Control Engineering* (McGraw-Hill, New York).
19. US Environmental Protection Agency (2013) *8-Hour Ozone Nonattainment Area Summary EPA Green Book* (US EPA, Washington, DC), Available at <http://www.epa.gov/oaqps001/greenbk/gnsum.html>.
20. Clean Air Act Amendments of 1990, Pub. L. No. 101-549, 104 Stat. 2468.
21. US Environmental Protection Agency (2009) *Assessment of the Impacts of Global Change on Regional U.S. Air Quality: A Synthesis of Climate Change Impacts on Ground-Level Ozone. An Interim Report of the US EPA Global Change Research Program* (US EPA, Washington, DC).
22. Jiang X, Wiedinmyer C, Chen F, Yang Z-L, Lo J-C (2008) Predicted impacts of climate and land use change on surface ozone in the Houston, Texas, area. *J Geophys Res* 113: D20312.
23. Karl TR, Melillo JM, Peterson TC, eds (2009) *Global Climate Change Impacts in the United States* (Cambridge Univ Press, New York).
24. Wu S, Mickley LJ, Kaplan JO, Jacob DJ (2012) Impacts of changes in land use and land cover on atmospheric chemistry and air quality over the 21st century. *Atmos Chem Phys* 12:1597–1609.
25. Federal Register (2010) National ambient air quality standards for ozone. *Federal Register Proposed Rules* 75(11):2938–3052.
26. Krupnick A, McConnell V, Cannon M, Stoessel T, Batz M (2000) *Cost-Effective NOx Control in the Eastern United States* (Resources for the Future, Washington, DC).
27. Smith WH (1990) *Air Pollution and Forests* (Springer, New York).
28. Kesselmeier J, Staudt M (1999) Biogenic volatile organic compounds (VOC): An overview on emission, physiology and ecology. *J Atmos Chem* 33:23–88.
29. Nowak DJ, Crane DE, Stevens JC (2006) Air pollution removal by urban trees and shrubs in the United States. *Urban For Urban Green* 4:115–123.
30. Nowak DJ, et al. (2000) A modeling study of the impact of urban trees on ozone. *Atmos Environ* 34:1601–1613.
31. Alonso R, et al. (2011) Modelling the influence of peri-urban trees in the air quality of Madrid region (Spain). *Environ Pollut* 159(8-9):2138–2147.
32. Baumgardner D, Varela S, Escobedo FJ, Chacalo A, Ochoa C (2012) The role of a peri-urban forest on air quality improvement in the Mexico City megalopolis. *Environ Pollut* 163:174–183.
33. McPherson EG, Scott KI, Simpson JR (1998) Estimating cost effectiveness of residential yard trees for improving air quality in Sacramento, California using existing models. *Atmos Environ* 32:75–84.
34. Escobedo FJ, et al. (2008) Analyzing the cost effectiveness of Santiago, Chile's policy of using urban forests to improve air quality. *J Environ Manage* 86(1):148–157.
35. California Air Resources Board (2011) *Compliance Offset Protocol for U.S. Forest Projects* (California Environmental Protection Agency, Sacramento).
36. Escobedo FJ, Kroeger T, Wagner JE (2011) Urban forests and pollution mitigation: Analyzing ecosystem services and disservices. *Environ Pollut* 159(8-9):2078–2087.
37. Roy S, Byrne J, Pickering C (2012) A systematic quantitative review of urban tree benefits, costs, and assessment methods across cities in different climatic zones. *Urban For Urban Green* 11(4):351–363.
38. Lei W, Zhang R, Tie X, Hess P (2004) Chemical characterization of ozone formation in the Houston-Galveston area: A chemical transport model study. *J Geophys Res* 109: D12301.
39. TexAQs II Rapid Science Synthesis Team (2007) *Final Rapid Science Synthesis Report: Findings from the Second Texas Air Quality Study (TexAQs II)* (Texas Commission on Environmental Quality, Austin).
40. Texas Commission on Environmental Quality (2010) Revision to the state implementation plan for the control of ozone air pollution. *Houston-Galveston-Brazoria 1997 Eight-Hour Ozone Standard Non-Attainment Area*. (TCEQ, Austin) Project No 2009-017-SIP-NR.
41. US Environmental Protection Agency (2004) *Incorporating Emerging and Voluntary Measures in a State Implementation Plan (SIP)* (US EPA, Research Triangle Park, NC).
42. Xiao X, Cohan DS, Byun DW, Ngan F (2010) Highly nonlinear ozone formation in the Houston region and implications for emission controls. *J Geophys Res* 115:D23309.
43. Carter WPL (2010) *Development of the SAPRC-07 Chemical Mechanism and Updated Ozone Reactivity Scales* (California Air Resources Board, Sacramento).
44. Cardelino CA, Chameides WL (1990) Natural hydrocarbons, urbanization, and urban ozone. *J Geophys Res* 95(D9):13971–13979.
45. US Environmental Protection Agency (2002) *EPA Air Pollution Control Cost Manual* (US EPA, Research Triangle Park, NC), 6th Ed.
46. Peters MS, Timmerhaus K, West R (2002) *Plant Design and Economics for Chemical Engineers* (McGraw-Hill, New York), 5th Ed.
47. Stanturf JA, et al. (2009) Restoration of bottomland hardwood forests across a treatment intensity gradient. *For Ecol Manage* 257:1803–1814.
48. Lower Mississippi Valley Joint Venture (LMJV) Forest Resource Conservation Working Group (2007) *Restoration, Management, and Monitoring of Forest Resources in the Mississippi Alluvial Valley: Recommendations for Enhancing Wildlife Habitat*, eds Wilson R, Ribbeck K, King S, Twedt D (LMJV, Vicksburg).
49. Staudhammer C, et al. (2011) Rapid assessment of change and hurricane impacts to Houston's urban forest structure. *Arboric Urban For* 37(2):60–66.
50. Klotzbach P, Gray W (2013) United States Landfalling Hurricane Probability Project. Available at <http://www.e-transit.org/hurricane/welcome.html>. Accessed February 17, 2014.
51. Thompson BK, Escobedo FJ, Staudhammer CL, Matyas CJ, Qiu Y (2011) Modeling hurricane-caused urban forest debris in Houston, Texas. *Landsc Urban Plan* 101: 286–297.
52. LANDFIRE (2005) Rapid Assessment Reference Condition Model: Southern Floodplain. R550PFif. Available at [http://www.fs.fed.us/database/feis/pdfs/PNVGs/South\\_Central/R550PFif.pdf](http://www.fs.fed.us/database/feis/pdfs/PNVGs/South_Central/R550PFif.pdf). Accessed March 13, 2013.
53. Texas A&M Forest Service (2014) Wildfire risk assessment portal. Available at <http://www.texaswildfirerisk.com/>. Accessed February 17, 2014.
54. US Forest Service 2012 National Insect and Disease Risk Map Viewer. Available at <http://foresthealth.fs.usda.gov/nidrm/>. Accessed February 17, 2014.
55. Byun DW, et al. (2003) Information infrastructure for air quality modeling and analysis: Application to the Houston-Galveston ozone nonattainment area. *J Environ Inform* 2(2):38–57.
56. Zhang F, et al. (2007) Impacts of meteorological uncertainties on ozone pollution predictability estimated through meteorological and photochemical ensemble forecasts. *J Geophys Res* 112:D04304.
57. The Nature Conservancy (2004) *Strategic Conservation Plan for the Columbia Bottomlands* (TNC, San Antonio).
58. King SL, Sharitz RR, Groninger JW, Battaglia LL (2009) The ecology, restoration and management of southeastern floodplain ecosystems: A synthesis. *Wetlands* 29(2): 624–634.
59. McConnell V, Walls M (2005) *The Value of Open Space: Evidence from Studies of Nonmarket Benefits* (Resources for the Future, Washington, DC).
60. Chen WY, Jim CY (2008) Assessment and valuation of the ecosystem services provided by urban forests. *Ecology, Planning and Management of Urban Forests: International Perspectives*, eds Carreiro MM, Song YC, Wu J (Springer, New York), pp 53–83.
61. Tang W, et al. (2008) Study of ozone “weekend effect” in Shanghai. *Sci China Ser D-Earth Sci* 51(9):1354–1360.
62. Mao J, et al. (2010) Atmospheric oxidation capacity in the summer of Houston 2006: Comparison with summer measurements in other metropolitan studies. *Atmos Environ* 44(33):4107–4115.
63. Zhao M, et al. (2013) Woody vegetation composition and structure in peri-urban Chongming Island, China. *Environ Manage* 51(5):999–1011.
64. Chameides WL, Lindsay RW, Richardson J, Kiang CS (1988) The role of biogenic hydrocarbons in urban photochemical smog: Atlanta as a case study. *Science* 241(4872): 1473–1475.
65. Taha H (1996) Modeling impacts of increased urban vegetation on ozone air quality in the South Coast Air Basin. *Atmos Environ* 30(20):3423–3430.
66. Duncan BN, et al. (2010) Application of OMI observations to a space-based indicator of NO<sub>x</sub> and VOC controls on surface ozone formation. *Atmos Environ* 44:2213–2223.
67. Das S, Vincent JR (2009) Mangroves protected villages and reduced death toll during Indian super cyclone. *Proc Natl Acad Sci USA* 106(18):7357–7360.
68. Scyphers SB, Powers SP, Heck KL, Jr, Byron D (2011) Oyster reefs as natural breakwaters mitigate shoreline loss and facilitate fisheries. *PLoS ONE* 6(8):e23296.
69. Bailey RG (1995) *Description of the Ecoregions of the United States* (USDA Forest Service, Fort Collins, CO) 2nd Ed. Misc. Pub. 1391.
70. The Nature Conservancy, US Forest Service, and US Geological Survey (1995) *Description of the ecoregions of the United States based on Bailey, Robert G* (TNC, US Forest Service, USGS, Washington, DC), 2nd Ed.
71. Nowak DJ, Greenfield E (2012) Tree and impervious cover change in U.S. cities. *Urban For Urban Green* 11:21–30.
72. Rosen DJ, De Steven D, Lange ML (2008) Conservation strategies and vegetation characterization in the Columbia Bottomlands, an under-recognized southern floodplain forest formation. *Nat Areas J* 28:74–82.
73. Oguz H, Klein AG, Srinivasan R (2007) Using the Sleuth Urban Growth Model to simulate the impacts of future policy scenarios on urban land use in the Houston-Galveston-Brazoria CMSA. *Res J Soc Sci* 2:72–82.
74. Byun DW, et al. (2005) Estimation of biogenic emissions with satellite-derived land use and land cover data for air quality modeling of Houston-Galveston ozone nonattainment area. *J Environ Manage* 75(4):285–301.
75. Bond J (2006) *The Inclusion of Large-Scale Tree Planting in a State Implementation Plan: A Feasibility Study* (Davey Resource Group, Geneva, NY).
76. Nowak DJ, Crane DE, Stevens JC, Ibarra M (2002) *Brooklyn's Urban Forest*. (US Forest Service, Northeastern Research Station, Newtown Square, PA) Gen. Tech. Rep. NE-290.
77. Lovett GM, et al. (2009) Effects of air pollution on ecosystems and biological diversity in the eastern United States. *Ann N Y Acad Sci* 1162:99–135.
78. Paoletti E (2009) Ozone and urban forests in Italy. *Environ Pollut* 157(5):1506–1512.
79. US Forest Service (2007) National ozone risk map (US Forest Service, Washington DC) Available at <http://nrs.fs.fed.us/fia/topics/ozone/pubs/pdfs/National%20Ozone%20Risk%20Map.pdf>. Accessed May 17, 2012.
80. TCEQ (2009) *Houston-Galveston-Brazoria Nonattainment Area Ozone Conceptual Model*. Draft (TCEQ, Austin). Available at [http://www.tceq.texas.gov/assets/public/implementation/air/arm/modeling/hgb8h2/doc/HGB8H2\\_Conceptual\\_Model\\_20090519.pdf](http://www.tceq.texas.gov/assets/public/implementation/air/arm/modeling/hgb8h2/doc/HGB8H2_Conceptual_Model_20090519.pdf). Accessed August 29, 2014.
81. Texas Commission on Environmental Quality (TCEQ) (2010) *Guidance for Implementing 1-hour NO<sub>2</sub> NAAQS for PSD. Part 2: Applicability of Appendix W Modeling Guidance for the 1-hour NO<sub>2</sub> NAAQS* (TCEQ, Austin). Available at [http://www.tceq.texas.gov/assets/public/permitting/air/memos/webguid\\_part2\\_naaqs.pdf](http://www.tceq.texas.gov/assets/public/permitting/air/memos/webguid_part2_naaqs.pdf). Accessed November 5, 2012.
82. Rivera C, et al. (2010) Quantification of NO<sub>2</sub> and SO<sub>2</sub> emissions from the Houston Ship Channel and Texas City industrial areas during the 2006 Texas Air Quality Study. *J Geophys Res* 115:D08301.
83. Office of Management and Budget (1992) *Circular A-94: Guidelines and Discount Rates for Benefit-Cost Analysis of Federal Programs* (White House, Washington, DC).
84. Prasad AM, Iverson LR, Matthews S, Peters M (2007-ongoing) A Climate Change Atlas for 134 Forest Tree Species of the Eastern United States [database] (US Forest Service,

- Northern Research Station, Delaware). Available at [http://www.nrs.fs.fed.us/atlas/tree/tree\\_atlas.html#](http://www.nrs.fs.fed.us/atlas/tree/tree_atlas.html#).
85. Iverson LR, Prasad AM, Matthews SN, Peters M (2008) Estimating potential habitat for 134 eastern US tree species under six climate scenarios. *For Ecol Manage* 254:390–406.
  86. Stanturf JA, Schoenholtz SH, Schweitzer CJ, Shepard JP (2001) Achieving restoration success: Myths in bottomland hardwood forests. *Restor Ecol* 9(2):189–200.
  87. Martin NA, Chappelka AH, Loewenstein EF, Keever GJ, Somers G (2012) Predictive open-grown crown width equations for three oak species planted in a southern urban locale. *Arboric Urban For* 38(2):58–63.
  88. Texas Forest Service (2011) Forest management sheet: Cost estimate sheet for forestry practices. Available at <http://txforests.tamu.edu/uploadedFiles/Landowners/Fact%20Sheet%20-%202011%20Forestry%20Practices%20Cost%20Estimate%20Sheet.pdf>.
  89. Allen JA, Keeland BD, Stanturf JA, Clewell AF, Kennedy HE, Jr (2001) *A Guide to Bottomland Hardwood Restoration*. (USDA Forest Service Southern Research Station, Asheville, NC) General Technical Report SRS-40.
  90. Lawrence AB, Escobedo FJ, Staudhammer CL, Zipperer W (2012) Analyzing growth and mortality in a subtropical urban forest ecosystem. *Landsc Urban Plan* 104(1):85–94.
  91. Jenkins JC, Chojnacky DC, Heath LS, Birdsey RA (2003) National-scale biomass estimators for United States tree species. *For Sci* 49(1):12–35.
  92. World Bank (2012) *State and Trends of the Carbon Market 2012* (World Bank, Washington, DC).
  93. Point Carbon (2013) California and RGGI market comment. *Carbon Market North America* 8(6):8.
  94. Point Carbon (2012) California and RGGI market comment. *Carbon Market North America* 7(4):27.
  95. Stevenson S, Morris B, Martin N, Grady M (2012) *Compliance Offset Supply Forecast for California's Cap-and-Trade Program (2013-2020)* (American Carbon Registry/Winrock International, Arlington, VA).
  96. Tol RSJ (2011) The social cost of carbon. *Annual Review of Resource Economics* 3(1):419–443.
  97. Interagency Working Group on Social Cost of Carbon (IWGSCC) (2010) *Technical Support Document: Social Cost of Carbon for Regulatory Impact Analysis Under Executive Order 12866* (US Government, Washington, DC).