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

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High ambient ozone (O_3) concentrations are a widespread and persistent problem globally. Although studies have documented the role of forests in removing O_3 and one of its precursors, nitrogen dioxide (NO_2), the cost effectiveness of using peri-urban reforestation for O_3 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_3 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_3 nonattainment area in Texas. The project would remove an estimated 310 tons of (t) O_3 and 58 t NO_2 total over 30 y. Given its location in a nitrogen oxide (NO_x)-limited area, and using the range of Houston area O_3 production efficiencies to convert forest O_3 removal to its NO_x equivalent, this is equivalent to 127–209 t of the regulated NO_x. The cost of reforestation per ton of NO_x 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_3 control efforts in Houston, other US nonattainment areas, and areas with O_3 pollution problems in other countries, wherever O_3 formation is predominantly NO_x limited.

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

Ground-level (tropospheric) ozone (O_3) is a secondary air pollutant formed through the chemical interaction of nitrogen oxides (collectively referred to as NO_x and comprising NO and NO_2) and volatile organic compounds (VOC) in the presence of conducive solar radiation and temperature conditions (1). Ground-level O_3 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_3 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_3 health effects (4), O_3 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_3 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_3 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_3 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_3 concentrations remain a widespread problem in many areas of the world. In the United States, O_3 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_3 nonattainment areas because at least one monitor exceeded the 75 ppb (daily 8-h average) 2008 National Ambient Air Quality Standard (NAAQS) for O_3 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.

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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_x) or large (25–50%; VOC) contribution of point sources to total stationary O_3 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_x emission limits by installing combustion controls (fuel switching, low-NO_x 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_x emissions from major sources in areas that fail to meet attainment deadlines.

In the United States, the O_3 problem may worsen due to continuing land use and climate change, especially rising temperatures (21–24). A possible tightening of the O_3 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_3 problem.

One as yet largely unexplored possibility for O_3 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_2 and O_3 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_3 concentrations depends on the magnitude of these two processes and on whether the project is located in an area where O_3 formation is limited by available NO_x 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_3 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_3 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_3 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_3 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_3 control and analyzing its cost-effectiveness as a compliance approach. We select the Houston–Galveston–Brazoria (HGB) O_3 nonattainment area in Texas as a case study because it exceeds O_3 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_3 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_3 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_x 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_3 and NO_2. However, because O_3 is not emitted directly, a SIP regulates point-source emissions not of O_3 but of its precursors, NO_x and VOC. Thus, the objective of an O_3 SIP reforestation project is the abatement of O_3 precursor equivalents. In the case of NO_x, these equivalents (NO_xe; *Case Study and Methods*) are the sum of the NO_2 directly removed by the forest, and the NO_x indirectly removed in the form of O_3 formed from NO_x 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_3 than forms from its VOC emissions. Whether the O_3 removed by the forest is equivalent to NO_x or VOC abatement depends on whether O_3 formation in the area is predominantly NO_x or VOC limited, respectively. Thus, depending on its location, a reforestation project may generate either only NO_xe or both NO_xe 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_xe 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_3 and 58.1 t NO_2 over our 30-y analysis period and store 24,574 t aboveground C at year 30. Annual air pollution removal was greatest in year 23 of the project at maximum canopy cover (O_3: 14,156 kg; NO_2: 2,659 kg), and C sequestration was highest in year 16 (1,153 t). For O_3 and NO_2, 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^2)	1,861,620	2,298,500
UFORE modeled air quality effects—Pollution removal		
NO_2 (g/m^2 crown area/y)	0.579	0.600
O_3 (g/m^2 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_3 and 455–911 t NO_2 per year over 30 y. Using the reported HGB area, O_3 production efficiency envelope of NO_x of 3–8 (38, 39), this results in an estimated 995–1,641 to 1,990–3,282 t NO_xe removal per year, or ∼1.7–5.5% of the average of the estimated 2006 and 2018 annual HGB-area NO_x 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_xe 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_x—land cost scenario 1: No land costs. The reforestation project would remove an estimated total of 209 or 127 t NO_xe over the 30-y analysis period in the "high" and "low" removal scenarios, respectively (incremental O_3 production efficiencies of NO_x = 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_xe (high removal) and $2,770–$5,300/t NO_xe (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_2e), the cost-effectiveness of NO_xe 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_xe in the low and high removal scenarios, respectively.

In all but the low removal scenario–where a project receives less NO_x credit for the O_3 removed due to high O_3 production efficiency of NO_x–reforestation in the HGB area has a lower mean cost per ton of NO_xe removed than conventional NO_x controls ($2,500–$5,000/t) and NO_x allowances ($3,300/t; *SI Appendix*, section S5); that is, permits granting a perpetual right to emit 1 t NO_x 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_x removed.

NO_x—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_3 than is formed from its VOC emissions. Because our case study project is not located in an area where O_3 formation is VOC limited, its O_3 removal is equivalent to abatement of NO_xe, 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_3 balance in a VOC-limited area.

Using the maximum incremental reactivity (MIR; ref. 43) scales (gram O_3/gram VOC) to estimate the quantity of O_3 formed by its VOC emissions, the project would produce an estimated 1,650 t O_3 over the 30-y analysis period, or over 5 times the estimated 310 t O_3 removed via dry deposition (*SI Appendix*, sections S6 and S7). The MIR scale represents relatively high NO_x conditions (areas with high NO_x:VOC ratios like Galveston Bay or southern Harris County in the HGB area; ref. 42), where O_3 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_3 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_3 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_2 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_x control options have an average economic lifetime of only 20 y (45). Full replacement of NO_x 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

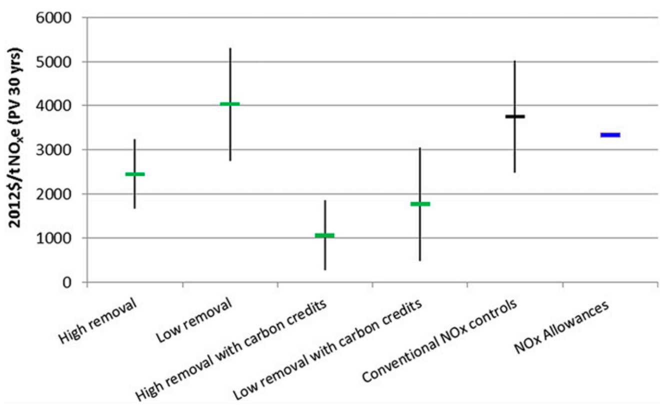

Fig. 1. Average cost per ton of NO_x control through reforestation at zero land cost, for high and low removal scenarios, and cost of standard point source controls and NO_x stream allowances in the HGB area. Vertical lines indicate ranges caused by different cost assumptions. Costs expressed as PV over 30-y period. NO_xe—NO_x equivalent.

horizon of the analysis also would increase the competitiveness of reforestation with conventional controls, as the established forest stand will keep removing O_3 and NO_2 at no additional cost beyond year 30. Thus, reforestation may be even more competitive with conventional NO_x controls than our analysis suggests.

Our finding that purchasing bottomland hardwood forest habitat for reforestation would not generate cost-competitive NO_x 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_x 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_xe 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_xe 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_3 have used regional weather and chemistry models [e.g., Weather Research and Forecasting (WRF)-Chem; ref. 32]. The high sensitivity of the O_3 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_3 precursor control, and hence their implementation. To promote reforestation for O_3 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_3 formation. Such a treatment would be justified on the basis that it does not conflict with the defined policy (SIP) goal of reducing O_3 concentrations, but rather promotes adoption of novel O_3 control measures that would achieve additional O_3 abatement beyond that achieved by legally mandated control technologies. Application of such a differentiated treatment of VOC emissions and accurate conversion of the O_3 removed by a reforestation project to NO_xe both require reasonably reliable spatial information on the type (NO_x, VOC) and degree (i.e., production efficiency) of sensitivity of O_3 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_3 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_3 nonattainment area would be cost competitive with additional conventional point source NO_x 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).

Our findings have implications for other areas with O_3 pollution problems in the United States and elsewhere. Given the large extent of suitable bottomland reforestation areas—historic bottomland forest currently in shrubland, grassland, or agriculture located in NO_x-limited portions of US O_3 nonattainment areas (Fig. 2) with similar O_3 and NO_2 concentrations (*SI Appendix*, section S7) and conventional NO_x control costs (*SI Appendix*, section S5)—we expect reforestation of peri-urban lands could be a cost-competitive NO_x control approach in many other existing US O_3 nonattainment areas. With high pollutant removal rates reported for peri-urban forests in high-O_3, NO_x-limited (61, 62) cities with deforested and degraded peri-urban areas in need of restoration such as Shanghai, China (63), and Mexico City, Mexico (32), reforestation may assist in O_3 control also in other countries.

Nevertheless, total pollutant removal by forests is space constrained: our study forest removes ~0.5 t NO_2 and 2.6 t O_3 km^{-2} y^{-1} and those in Shanghai and Mexico City remove around 2.1 t NO_2 km^{-2} y^{-1} (no O_3 removal estimate due to lack of ambient concentration data; ref. 61) and 0.04 t NO_2 and 1.7 t O_3 km^{-2} y^{-1} (32), respectively. Thus, reforestation clearly could not replace all additional, let alone existing, conventional controls.

In the United States, opportunities already exist to integrate reforestation into regulatory O_3 abatement efforts. Specifically, EPA's (41) policy "Incorporating Emerging and Voluntary Measures in a State Implementation Plan" encourages states to explore novel approaches to achieve NAAQS compliance. The policy defines as "emerging" a measure that does not have the same high level of certainty for quantification purposes as traditional measures but still fulfills the usual SIP requirements of being surplus, enforceable, quantifiable, permanent, and antibacksliding. It explicitly mentions tree planting as an example of an ambient concentration-reducing measure that could be used for purposes of SIP attainment, reasonable further progress, rate of progress, or maintenance requirements. Under this policy, being an emerging SIP measure, reforestation could not replace emission controls already mandated in a SIP (antibacksliding). Rather, it would form part of a suite of additional policies, incentives, and controls implemented for attainment or maintenance purposes.

Importantly, forests may lower ambient O_3 levels in NO_x-limited areas by more than a conventional control device with the same total annual amount of NO_xe abatement. This is due to the fact that photosynthetic activity and thus O_3 and NO_2 removal by trees are clustered around the O_3 season (May–September), and abatement by conventional controls is distributed more or less evenly throughout the year. Whether or not higher O_3 reductions would result in additional health benefits depends on differences in the spatial patterns of O_3 reductions produced by the two control options and resulting differences in total human exposure.

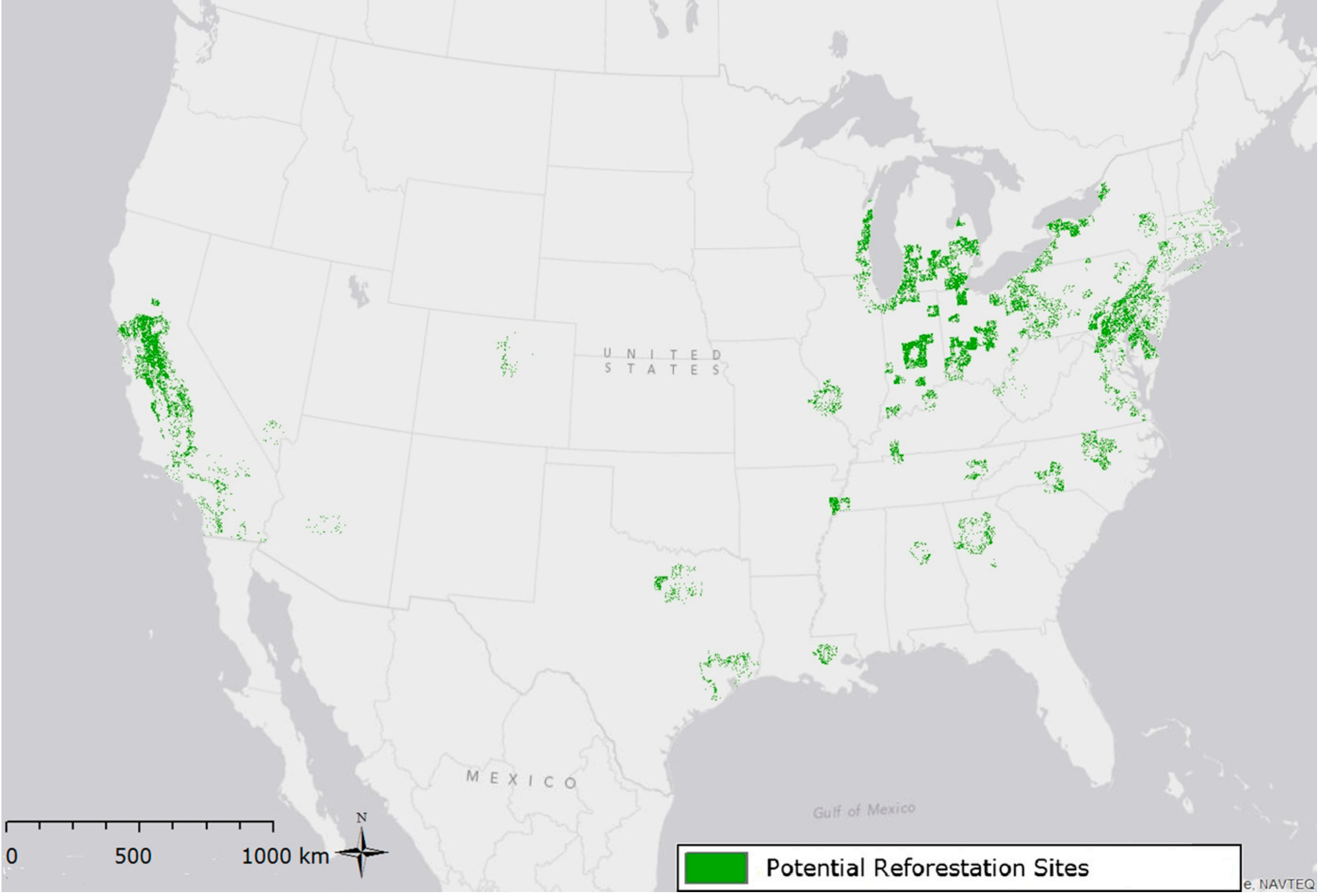

Fig. 2. Potential sites in the conterminous United States where reforestation could abate ozone. Sites were identified by intersecting O_3 nonattainment and maintenance areas (80 ppb 1997 8-h standard; ref. 19), NO_x-limited areas (formaldehyde/NO_2 ratio ≥1, from figure 6 in ref. 66), pre-European settlement forested areas [LANDFIRE Biophysical Settings Layer Refresh 2008 (lf_1.1.0), US Department of the Interior, Geological Survey, www.landfire.gov/vegetation.php] and lands presently under grass, shrub or agricultural cover (NLCD2006 Landcover, US Department of the Interior, Geological Survey, www.mrlc.gov/nlcd06_data.php).

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_xe 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_xe vs. VOCe. Ozone and NO_2 removal rates by trees generally increase with pollutant concentrations (29). Although very high pollutant concentrations can reduce photosynthetic rates

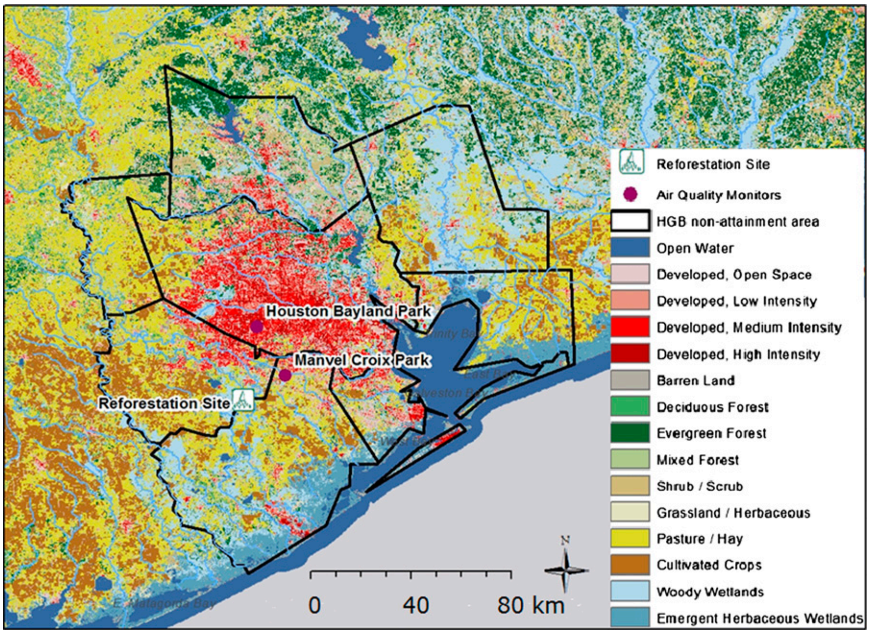

Fig. 3. Map showing the HGB nonattainment area, the reforestation site, Texas Commission on Environmental Quality air quality monitoring stations from which data were obtained, and land cover.

or makes trees more susceptible to other stressors such as insect or pathogen attack (77, 78), exposure levels to O_3 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_3 and NO_2 removal rates in areas with the highest O_3 and NO_2 concentrations, respectively.

Nitrogen dioxide removal by forests–appropriately adjusted for uncertainties–can be translated directly into NO_x removal a reforestation project could claim. The calculation of the precursor removal a project could claim for the O_3 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_3. Converting O_3 removed into its equivalent quantities of avoided precursors requires information on the O_3 production efficiency in the area, that is, the number of O_3 molecules formed from a precursor molecule (80).

Results from simulations (38) and measurement flights (39) indicate an O_3 production efficiency envelope of 3–8 in the HGB area for NO_x; that is, 3–8 O_3 molecules are formed per NO_x molecule oxidized during midday hours, making removal of an O_3 molecule equivalent to removal of 1/3–1/8 of a NO_x molecule under NO_x-limited conditions. We use this 3–8 range to develop high and low removal cases, respectively, for our reforestation project.

With O_3 formation in our study and much of the HGB area characterized as NO_x limited, we convert O_3 removed by the project to NO_2 equivalents (NO_2e) using the molecular weights of the two compounds and the 3–8 range of O_3 production efficiencies of NO_x (i.e., 1 O_3 = 0.32–0.12 NO_2). Because point sources are regulated on emissions of NO_x (most of which are immediately converted to NO_2 in the atmosphere) and not NO_2, we follow existing policy guidance and convert NO_2e to NO_xe using the national NO_2:NO_x 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_x 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_3 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_3 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_3 and NO_2 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):

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

Crown area (CA) for each year was calculated as CA = π x $(CW/2)^2$ 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_2 and O_3 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_3 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):

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

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 State 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_2e (\$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_2 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 (PRTP) 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% PRTP) 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).

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