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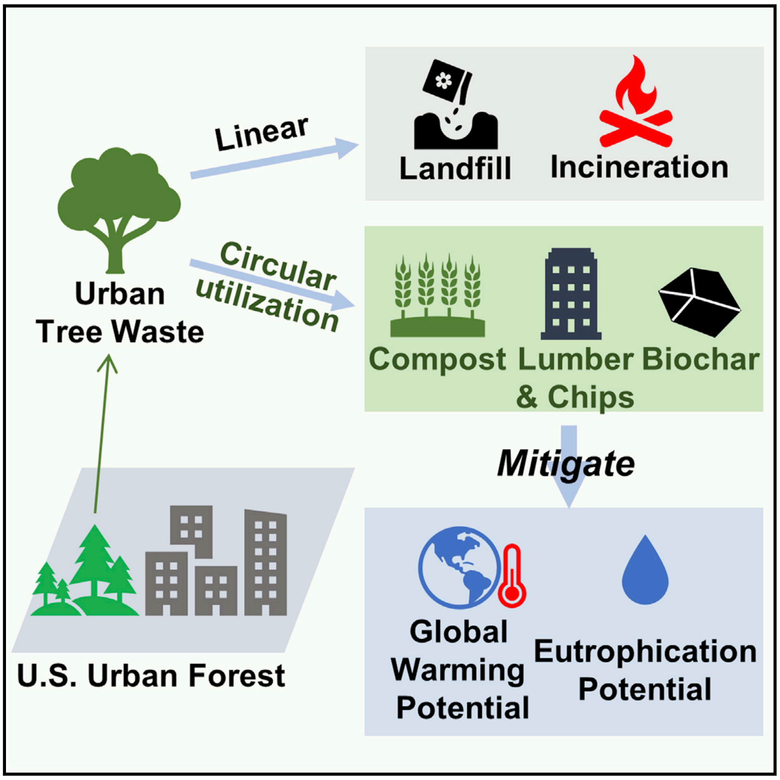
# Circular utilization of urban tree waste contributes to the mitigation of climate change and eutrophication

## Graphical abstract

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## In brief

Sustainable and circular utilization of urban tree waste is essential for urban resource management and environmental impact mitigation. We find that converting urban tree waste to compost, lumber, chips, and biochar substantially reduces the global warming potential and eutrophication potential compared with landfilling. Such environmental benefits vary with locations due to the availability and types of urban tree wastes. Our results highlight the necessity of developing a circular bioeconomy in the urban environment.

## Highlights

- A multi-scale LCA explores the pathways of utilizing urban tree waste in the US
- Utilizing urban tree waste can yield nationwide environmental benefits in the US
- Highlighting needs to move from landfilling and combusting to valuable utilization
- Environmental benefits have geospatial variations at the state and city levels

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# Circular utilization of urban tree waste contributes to the mitigation of climate change and eutrophication

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**SCIENCE FOR SOCIETY** More than 45 million dry tonnes of urban tree waste are generated every year in the US. Landfilling these wastes generates greenhouse gas emissions that contribute to climate change and nutrient-related emissions causing eutrophication that results in water crises such as harmful algal boom and fish kills. Converting urban tree waste into valuable products can help mitigate climate change and eutrophication, but it is important to understanding the extent to which reusing tree waste reduces these impacts. We find that converting urban tree waste to compost, lumber, chips, and biochar substantially reduces national environmental emissions compared with landfilling waste. Such benefits vary by location within the US, and the most environmentally beneficial combination is using merchantable logs for lumber and residues for biochar. Our results highlight the feasibility and environmental benefits of recycling/reusing urban tree wastes.

## SUMMARY

Substantial urban tree waste is generated and underutilized in the US. Circular utilization of urban tree wastes has been explored in the literature, but the life-cycle environmental implications of varied utilization pathways have not been fully understood. Here we quantify the life-cycle environmental benefits of utilizing urban tree wastes at process, state, and national levels in the US. Full utilization of urban tree wastes to produce compost, lumber, chips, and biochar substantially reduces nationwide global warming potential (127.4–251.8 Mt $CO_2$ eq./year) and eutrophication potential (93.9–192.7 kt N eq./year) compared with landfilling. Such benefits vary with state-level locations due to varied urban tree waste availability and types. Process-level comparisons identify the most environmentally beneficial combination as using merchantable logs for lumber and residues for biochar. The results highlight the climate change and eutrophication mitigation potential of different circular utilization pathways, supporting the development of circular bioeconomy in the urban environment.

## INTRODUCTION

Urban forest, as a vital component of the urban system, provides many benefits to both human and natural systems, including temperature and microclimatic modification, pollution mitigation, noise reduction, biodiversity and habitat enhancement, and recreational opportunities.[1–5] Urban forest is also a crucial source of biomass.[6] The urban forest in the US is estimated to hold over 800 million metric tons (Mt) of carbon on $\sim$ 51.5 million hectares of urban land.[7,8] Each year, tree waste is generated by deciduous trees, tree or yard maintenance, land clearing, and tree removal (e.g., due to mortality, windstorms, pests).[6] The US urban forest generates over 25 million oven dry metric tons (ODMT) of leaf waste and over 20 million ODMT of tree waste per year, equating to more than 20 Mt of carbon mass.[7] Substantial urban tree waste is underutilized or not used, such as being landfilled (potentially high methane emissions), burned (immediate release of biogenic $CO_2$), or left on site.[7,9,10] There is increasing interest in exploring economically valuable and environmentally beneficial urban tree waste utilization pathways.[3,6,11,12] For example, Nowak et al. estimated the potential economic benefits of producing compost, lumber, chips, firewood, and pallets from US urban tree waste ($89–$786 million/year, depending on the combinations of products).[7] Other studies discussed potential environmental benefits such as mitigating climate change and reducing eutrophication by converting urban tree wastes to products that can be carbon sinks and replace materials with high environmental burdens.[9,13] Different utilization pathways exist for urban tree waste, and understanding the potential environmental benefits of each pathway is critical for the large-scale sustainable management of urban forest and the implementation of industrial-urban symbiosis.[7,14,15]

Several studies have evaluated the potential environmental benefits of utilizing urban tree waste.[3,6,11,12,16] Most of them have quantified the carbon storage changes in urban forests or the carbon footprint of varied urban tree waste management (e.g., landfill and incineration).[3,6,11,12,16] However, few studies have systematically quantified and compared the life-cycle emissions of products made from urban tree waste with the counterpart products made from virgin materials. Substantial variabilities throughout the life cycle of waste-derived and virgin-material-based products affect the environmental benefits of diverse urban tree waste utilization pathways, which have not been explored previously. Furthermore, previous studies rarely considered the counterfactual end-of-life cases (e.g., landfill or field application) affecting the carbon balances and environmental emissions.

Here we address those research gaps by developing a multiscale cradle-to-grave life cycle assessment (LCA) integrating process and Monte Carlo simulation (MCS). This study aims to tackle the research question of what the life-cycle environmental implications are of managing and utilizing the US urban tree waste in various pathways. To answer this question, this study chose varied urban tree waste utilization pathways to produce compost, mulch, electricity, lumber and chips, and biochar. They were selected based on their potential economic and environmental benefits discussed in the previous literature.[7,17–21] LCA is a standardized and widely accepted tool to evaluate the environmental impacts of a product or a service throughout its life cycle.[17,22–26] To explore the potential environmental benefits, counterfactual systems for alternative end-of-life cases were considered. (e.g., landfill, incineration, and mulch represent current treatment methods of urban tree waste). Different utilization scenarios that produce various products were explored, including compost, electricity, mulch, lumber, chips, and biochar. The environmental benefits of product substitutions are included (i.e., mineral fertilizers, grid-purchased electricity, lumber and chips made from virgin wood, and charcoal for soil amendment). All the urban tree utilization pathways were developed based on commercially available and proven technologies.[17,27,28] Furthermore, this study contributes to the LCA community by providing process-based models and life-cycle inventory (LCI) data for each tree waste utilization pathway. These models are parametric, and can be used by other researchers and LCA practitioners for different tree types and operational conditions. The stakeholders and policymakers can further use the results presented in this study to tailor their strategies or policy toward sustainable management of urban tree waste.

## RESULTS

### Methods summary

In this study, the system boundary includes raw material extraction, production, transportation, and end of life. This study developed and coupled process simulation models with LCA. The process simulation models provided LCI data (e.g., mass and energy balances, and environmental emissions) of composting, wood product manufacturing (i.e., lumber and chips), and biochar production by pyrolysis. These process simulation models also allow for investigating the impacts of parameter variations (e.g., variations of biomass composition, conversion process operations, and end of life) on LCI. To run MCS, random samples of parameters with variations were generated based on their statistical characteristics (determined from literature data), which were used as inputs to the process simulation models to produce LCI data and further LCA results.

Since climate change and eutrophication are the two major environmental challenges related to urban tree waste management widely discussed in previous literature,[11,12,16,29,30] this study focuses on two indicators: global warming potential (GWP) and eutrophication potential. GWP is a standard indicator to measure "how much energy the emissions of 1 ton of a gas will absorb over a given period of time, relative to the emissions of 1 ton of carbon dioxide ($CO_2$)."[31] It has been widely used in LCAs to quantify the climate change implications of different greenhouse gas (GHG) emissions.[17,18,32] Eutrophication is a phenomenon of nutrient enrichment in aquatic ecosystems and eutrophication potential is a common impact category in different life-cycle impact assessment methods (e.g., Tool for Reduction and Assessment of Chemicals and Other Environmental Impacts [TRACI], ReCiPe).[31,33,34]

The urban tree wastes in this study are categorized into three groups based on their sizes, including leaf waste, merchantable logs, and residues. Removed trees with stem diameters larger than 22.9 cm (9.0 inches) for softwood and 27.9 cm (11.0 inches) for hardwood are classified as merchantable trees. The stem parts of merchantable trees are merchantable logs that can be used for lumber production. The rest of the removed trees with smaller diameters are considered non-merchantable trees.[7] The residues (e.g., branches, limbs, needles[17]) of merchantable trees and non-merchantable trees are classified as residues.

In this study, five scenarios were established to understand the impacts of different pathways in processing the urban tree waste (see section "scenario analysis", Table 1, and Figure 5 for details). Scenarios were developed based on the sustainability assessment guideline for urban forests developed by the US Department of Agriculture (USDA) Forest Service.[35] The guideline provides four levels of urban wood utilization. Landfilling is ranked as low utilization, while chips and mulch are considered as fair utilization. Reusing or recycling most urban wood wastes for energy, products, or other purposes beyond chips or mulch is good utilization. Comprehensive utilization of all biomass wastes is suggested as optimal utilization. Based on this guideline, landfilling was selected for scenario 1 as the baseline for low utilization. Scenarios 2–4 are designed to explore the combinations of different options mentioned by the guideline for fair and good utilization, including chips, mulch, energy recovery (by incineration), and other products. Specifically, scenario 2 incinerates removed trees for power generation, but leaf waste is still landfilled. In scenario 3, leaf waste is collected to produce compost as organic fertilizer that replaces the mineral fertilizers (i.e., nitrogen, phosphorus, and potassium fertilizers), and removed trees are incinerated. In scenario 4, leaf waste is composted, and removed trees are chipped to serve as mulch, which is a common practice.[36] More details of each scenario are described in section "methodology."

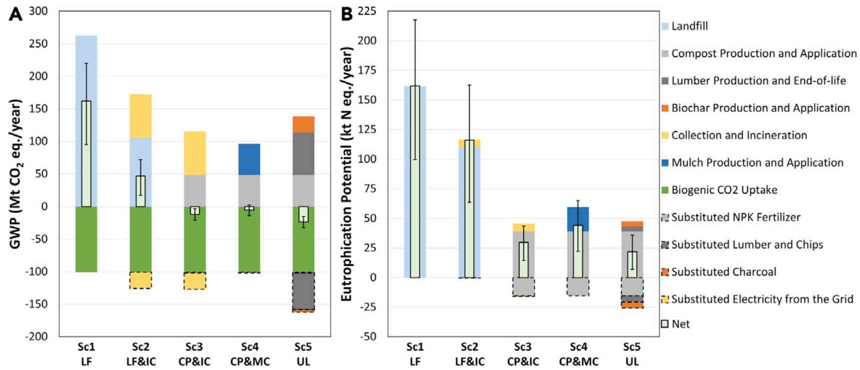


**Figure 1. National life-cycle GWP and eutrophication potential of five scenarios**

(A and B) The national life-cycle GWP (A) and eutrophication potential (B) of five scenarios. Scenario 1: landfilling (LF), all tree wastes are landfilled; scenario 2, LF and incinerating (IC), leaf waste is landfilled and removed trees are incinerated for electricity generation; scenario 3, composting (CP) and IC, leaf waste is composted and removed trees are incinerated for electricity generation; scenario 4, CP and mulching (MC), leaf waste is composted and removed trees are produced into mulch; scenario 5, UL, leaf waste is composted, merchantable logs are produced into lumber and chips, and residues of merchantable trees and non-merchantable trees are produced into biochar. Substitution benefits are shown in bars with dashed borderline (negative values). The error bars show the 5th–95th percentile range of net GWP and eutrophication potential provided by the Monte Carlo simulation.

Scenario 5 is designed for an optimal utilization case where all urban tree wastes are turned into products other than energy and mulch. Previous studies identified merchantable logs and chips as economically viable products,[7,37,38] but it is only limited to tree waste with large enough stem diameters to be salable as sawlogs or pulpwood.[39] For residues, converting them to biochar was selected given that biochar is widely considered a negative emission technology to combat climate change.[40] Biochar has been used as a soil amendment and shows high stability in soil. For example, more than 80% of carbon can be retained in biochar from medium temperature pyrolysis even after 100 years for cropland or grassland applications.[18,41] Hence, biochar can be a potential stable long-term carbon pool.[42] Urban-tree-waste-derived biochar can substitute traditional charcoal, which is included in this analysis as substitution effects. The leaf waste is not suitable for merchantable applications or biochar due to large variations in high ash and low carbon content.[7,29,43–45] Therefore, scenario 5 matches three types of urban tree wastes with different utilizations: leaf waste is composted, merchantable logs are utilized to produce lumber and chips, and residues are converted to biochar that is used as a soil amendment and substitutes for traditional charcoal.

Not all urban tree wastes are landfilled in the US;[10,35] however, the data of detailed breakdown by different utilization pathways are not available. For example, leaf waste accounts for ~46% of the total dry weight of urban tree waste in the US. They can be either landfilled or composted, and the split between the two is unknown. Therefore, scenario 2 simulates an extreme case where all leaf waste is landfilled, while scenarios 3–5 consider leaf waste to be fully composted. Similarly, removed trees (~54% of total dry urban tree waste mass) are incinerated in scenarios 2 and 3, and chipped into mulch in scenario 4. Scenario 5 fully utilizes merchantable logs (~26% of total dry urban tree waste mass) for lumber and chips, and residues (~28% total of dry urban tree waste mass) for biochar. We use the five scenarios to bound the possibility so that the environmental impacts of different utilization combinations will fall between scenario 5 (lowest environmental impact) and scenario 1 (highest environmental impact).

## National-level life-cycle results

Figure 1 displays the life-cycle GWP and eutrophication potential of processing the US urban tree waste annually in five scenarios. The product-level results of processing 1 ODMT of three types of urban tree waste are shown in Figure 2. The bars in Figures 1 and 2 were plotted based on the average values of MCS and the ranges of the net GWP and eutrophication potential are visualized by error bars (5th–95th percentile, P5–P95). The results are cradle to grave, including biogenic $CO_2$ uptake, emissions of biomass conversion, upstream burdens of producing and transporting fuels and materials, and end-of-life emissions. The substitution benefits were estimated for products that can be replaced by urban tree waste-derived products and plotted by bars with a dashed borderline. The statistical distribution of national-level GWP and eutrophication potential results from MCS are shown in Figure S1.

Figure 1A shows the significant climate benefits of utilizing urban tree wastes nationwide compared with traditional waste treatment methods of landfilling and incineration. The biogenic $CO_2$ uptake (green bar) associated with urban tree waste is 100.9 Mt/year by the mean value, which is either released or stored in different carbon pools, depending on the waste treatment and utilization scenarios. Landfilling urban tree waste (light blue bar in scenario 1) releases the most GHG emissions (262.8 Mt $CO_2$ eq./year), resulting in the highest GWP as 161.9 Mt $CO_2$ eq./year (P5–P95: 103.5–228.5) among all scenarios. The high GWP of landfilling is contributed by $CH_4$ emissions in landfill gas (92% of total GWP). The GWP related to collection and transportation is minor (<2% of total GWP). Scenario 2 partially avoids landfilling by incinerating removed trees for power generation, reducing the net GWP to 46.6 Mt $CO_2$ eq./year (P5–P95: 21.2–76.1). Such reduction is due to the reduced $CH_4$ emissions in landfill gas as almost all the carbon sequestered by removed trees is immediately released as $CO_2$ by combustion. However, leaf waste is still landfilled in scenario 2 and contributes to the most GWP, 105.7 Mt $CO_2$ eq./year. The second carbon emission source is collection and incineration (in total 66.8 Mt $CO_2$ eq./year), 98% of which is attributed to incineration. The electricity generated from removed trees can replace the electricity purchased from the grid, and such substitution benefit was estimated as 24.9 Mt $CO_2$ eq./year based on the GWP of average US stationary mix electricity (yellow bar with dashed borderline in Figure 1A).[46] The results of scenarios 1 and 2 highlight the need to avoid landfilling or enhance $CH_4$ recovery.[47] This study does not consider $CH_4$ recovery and utilization in the landfill baseline, following the Intergovernmental Panel on Climate Change (IPCC) practice of setting the national methane recovery

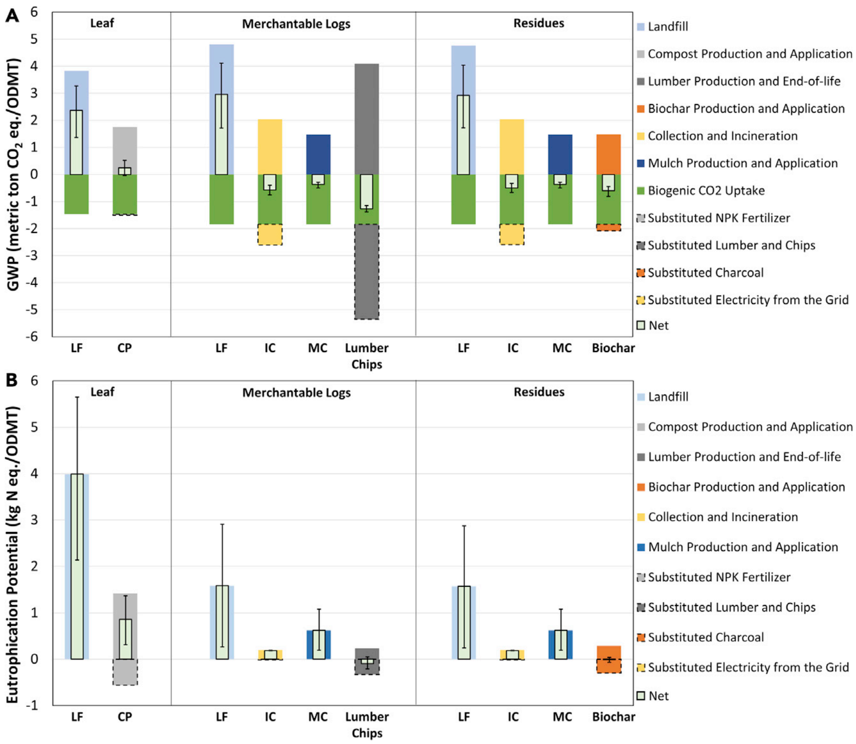


**Figure 2. Life-cycle results of processing 1 ODMT of three types of urban tree waste in varied pathways**

(A and B) The life-cycle GWP (A) and eutrophication potential (B) of processing 1 ODMT of three types of urban tree waste (i.e., leaf waste, merchantable logs, and residues) in varied pathways. These pathways include LF, CP, IC, MC, producing lumber and chips, and producing biochar. The error bars show the 5th–95th percentile range of Monte Carlo simulation results.

factor to be zero to provide the worst scenario baseline for quantifying the maximum potential of an intervention strategy.[47] If $CH_4$ recovery and utilization is included, the baseline GWP may still be higher than the incineration GWP, because (1) $CH_4$ recovery efficiency is not 100%, commonly ranging from 35% to 90%,[48–52] and (2) currently, landfill $CH_4$ recovered is either directly burned to generate electricity/heat or used as renewable natural gas that is still combusted in their end use.[53] Given the high GWP and eutrophication potential of the incineration option compared with other utilization pathways, it is unlikely that including $CH_4$ recovery and utilization will change the comparative conclusions of this study. Currently, the US Environmental Protection Agency (EPA) has a voluntary program, Landfill Methane Outreach Program, targeting reducing methane emissions from landfill sites.[53] Given the significant variations of recovery efficiency, operational status, and final products (electricity, renewable natural gas, or heat),[48–52] it is a valuable future research direction to quantify the site-specific GWP of landfills with $CH_4$ recovery and utilization.

Landfilling is completely avoided in scenario 3 where removed trees are combusted, and leaf waste is composted. Composting (light gray bar in Figure 1A) leads to 48.2 Mt $CO_2$ eq./year, much lower than landfilling the same amount of leaf waste in scenario 2. The end of life of compost (after soil applications) emits GHG and eutrophication-contributing emissions through a decay process (see Note S1). By the mean value, the end of life of compost leads to 14 Mt $CO_2$ eq./year, 29% of the total GWP of compost production and application. The compost can replace mineral fertilizers, leading to a credit of 1.0 Mt $CO_2$ eq./year (light gray bar with dashed borderline in Figure 1A but too small to see). This study does not simply assume that 1 kg of nutrient (i.e., nitrogen, phosphorus, and potassium) in compost could replace 1 kg of nutrient in mineral fertilizers. Instead, the quantity of substituted mineral fertilizers was estimated based on the plant available nutrient percentage of compost and mineral fertilizers (see section "product substitution" for details). Scenario 3 has a net GWP of −11.9 Mt $CO_2$ eq./year (P5–P95: −20.9 to −2.8), where most of the biogenic $CO_2$ uptake is released, the negative values are largely contributed by the electricity credit from incineration. Compared with scenario 3, scenario 4 collects and chips the removed trees into mulch (leaf waste is used for composting, same as scenario 3).[36] Mulch after soil application follows a similar decay process to compost and releases GHG and eutrophication-contributing emissions (see Note S5). For removed trees, GWP associated with mulch in scenario 4 is lower than the incineration option in scenario 3; however, incineration has considerable substitution benefits that mulch does not have, resulting in a higher net GWP of scenario 4 than scenario 3.

Scenario 5 fully utilizes different parts of urban tree wastes. Scenario 5 reaches the lowest net GWP −23.6 Mt $CO_2$ eq./year (P5–P95: −31.9 to −14.7). Leaf waste composting has the same GHG emissions in scenario 5 as in scenarios 3 and 4 given the same quantity of leaf waste in the three scenarios. Other GHG emissions sources in scenario 5 are associated with lumber and biochar. Lumber (gray bar in Figure 1A) contributes to most of the GWP (65.4 Mt $CO_2$ eq./year), of which 70% comes from the end of life of lumber. This study assumed that lumber is sent to landfill after it reaches the end of its lifetime (e.g., used in buildings), given that currently 69.4% of wood materials in construction and demolition wastes and 67.2% of wood wastes in municipal solid waste are sent to landfill in the US.[54,55] This dominance of lumber-related GWP can also be observed in Figure 2A, where processing 1 ODMT logs into lumber and chips has large emissions. However, the large GHG emissions associated with lumber can be offset by substituting lumber and chips made from virgin wood harvested from non-urban forests, such as plantation forests in rural areas (gray bar with dashed borderline in Figure 1A, 56.1 Mt $CO_2$ eq./year by the mean value). The quantity of substitution benefits depends on the source of wood that urban logs will replace, and this study considers wood products from both hardwood and softwood forests based on the data given by ecoinvent (see Table S8).[56] The smallest GWP contributor in scenario 5 is the pyrolysis production and soil application of biochar (25.0 Mt $CO_2$ eq./year). This substitution benefit of biochar (in replacing charcoal) is 4.0 Mt $CO_2$ eq./year, much smaller than the substitution benefits of lumber. Scenario 5 converts the tree residues to biochar, which retains most of the carbon and releases much less GHG emission (only 6% of all biochar-related emissions) than incineration in scenario 3 and mulch in scenario 4.

An additional analysis was conducted to compare the carbon stock changes of five scenarios, which provides insights into the benefits of urban tree utilization in retaining carbon in diverse carbon pools, in addition to the net GWP as presented in Figure 1A. Figure S2 shows the carbon stock changes after 100 years (the same time frame of the IPCC GWP characterization factors used in this study[57]) in five scenarios, compared with the original carbon stored in the raw urban tree waste. Due to the slow decay in landfills, scenario 1 maintains the highest carbon stock (16.9 Mt C), but scenario 1 has the highest GWP caused by significant $CH_4$ emissions, as discussed previously. Among the rest, full utilization of urban tree wastes for producing compost, lumber and chips, and biochar in scenario 5 shows the highest carbon stock (7.8 Mt C), which is 15.6%, 493.3%, and 40.7% higher than scenarios 2, 3, and 4, respectively. Among composting, incineration, mulching, producing lumber and chips, and biochar, incineration is the least favorable because of the immediate release of all carbon, in other words, the carbon stock is zero after incineration.

The eutrophication potential result in Figure 1B shows similar trends to that in Figure 1A for GWP. Landfill in scenario 1 has the highest environmental burdens, as high as 161.8 thousand metric tons (kt) of N eq./year by mean value with P5–P95 as 105.8–224.1 kt N eq./year. The high eutrophication potential is attributed to the release of $NH_3$, $NO_3^-$, and majorly $NH_4^+$ from leaf waste and removed trees in landfills (see Note S4). The net eutrophication potentials of scenario 2 decreased to 116.1 kt N eq./year, which is contributed by the reduced landfilling due to the incineration of removed trees. However, scenario 2 has higher eutrophication potential than scenarios 3–5 due to the significant environmental release of N in leaf waste that has higher nitrogen content than logs and residues (see Tables S3 and S4). Collection and incineration in scenario 2 (dark blue bar in Figure 2B) only accounts for 6.6 kt N eq./year, and the substitution benefit of electricity is minor (0.5 kt N eq./year). The eutrophication potential of leaf waste treatment can be greatly reduced by composting, as demonstrated by the results of scenario 3 (39.1 kt N eq./year in Figure 1B) along with a credit of 15.4 kt N eq./year from substituted mineral fertilizers. The net eutrophication potential of scenario 3 is 29.7 kt N eq./year (P5–P95: 15.8–45.0), 74.4% lower than the results of scenario 2. In scenario 4, the net eutrophication potential increases to 44.1 kt N eq./year (P5–P95: 23.2–66.1) mainly due to nitrogen-related emissions that contribute to eutrophication in mulch end of life (see Note S5).

Scenario 5 has the lowest net eutrophication potential (21.9 kt N eq./year, 86.4%, 81.1%, 26.3%, and 50.3% lower than scenarios 1, 2, 3, and 4, respectively). Compared with scenarios 3 and 4, the lower eutrophication results of scenario 5 are mostly attributed to biochar and lumber (as the results of leaf composting are the same across the three scenarios). Converting removed trees into wood products and biochar leads to lower eutrophication potential (compared with mulching and incineration) with additional substitution benefits. Biochar production and application lead to 4.9 kt N eq./year of eutrophication potential, most of which is caused by biochar production (e.g., emissions from the production and combustion of fuels). Such environmental burdens are almost canceled out by substituting charcoal. The production and end of life of lumber result in 3.8 kt N eq./year of eutrophication potential with a substitution benefit of 5.3 kt N eq./year based on the ecoinvent database for hardwood and softwood products (see section "product substitution" and Table S8).[56] Across three utilization scenarios (scenarios 3–5) in Figure 1B, life-cycle eutrophication impacts are always dominated by compositing (>65%), unlike the GWP results in Figure 1A, which are driven by lumber production, mulching, or incineration.

A few conclusions can be made based on the national-level results shown in Figure 1. First, fully utilizing urban tree wastes can substantially reduce climate change impacts and eutrophication burdens of landfilling or incinerating urban tree wastes in the US. In 2019, the total annual GHG emissions of the US was 6,558 Mt of $CO_2$ eq., 10% of which were agriculture emissions ($\sim$656 Mt $CO_2$ eq.).[58] By converting urban tree wastes into compost, wood products, and biochar, 186 Mt $CO_2$ eq./year can be reduced compared with traditional landfilling, accounting for $\sim$28% of total US agriculture GHG emissions. Second, different utilization pathways for diverse types of urban tree wastes have varied implications for climate change impacts and eutrophication potential. For example, lumber production from logs has significant contributions to GWP, while its contribution to eutrophication is minor. On the contrary, using leaf wastes for compost has lower contributions to GWP than their contributions to eutrophication.

## Product-level life-cycle results

To better understand the emissions and reduction potential of the individual pathway, product-level results are presented in Figure 2 by types of urban tree wastes and utilization pathways. Figure 2A shows landfilling with the highest GWP and eutrophication potential across all the processing pathways, although different types of urban tree waste in landfills lead to varied environmental burdens. Merchantable logs and residues have higher carbon contents (Tables S3 and S4), resulting in higher GWP than leaf waste in landfills. In contrast, leaf waste has higher nitrogen content, leading to higher eutrophication potential than merchantable logs and residues in landfills. Composting is a better option for leaf waste than landfilling in terms of GWP and eutrophication. For merchantable logs, producing lumber and chips shows the largest benefits in both net GWP ($-$1.3 metric ton $CO_2$ eq.) and net eutrophication potential ($-$0.1 kg N eq.), compared with incinerating ($-$0.6 metric ton $CO_2$ eq. and 0.2 kg N eq.) and mulching ($-$0.4 metric ton $CO_2$ eq. and 0.6 kg N eq.). Such benefits highly depend on the substitution effects, i.e., 3.5 metric ton $CO_2$ eq. GWP and 0.3 kg N eq. eutrophication potential. Biochar is the best option for forest residues given the lower GWP and eutrophication potential than incinerating and mulching. Additionally, the wide P5–P95 range in Figure 2B is associated with nitrogen content variability (over 50% variation from the average), which is 0.54%–1.62% for leaf waste and 0–0.8% for logs and residues (see Tables S3 and

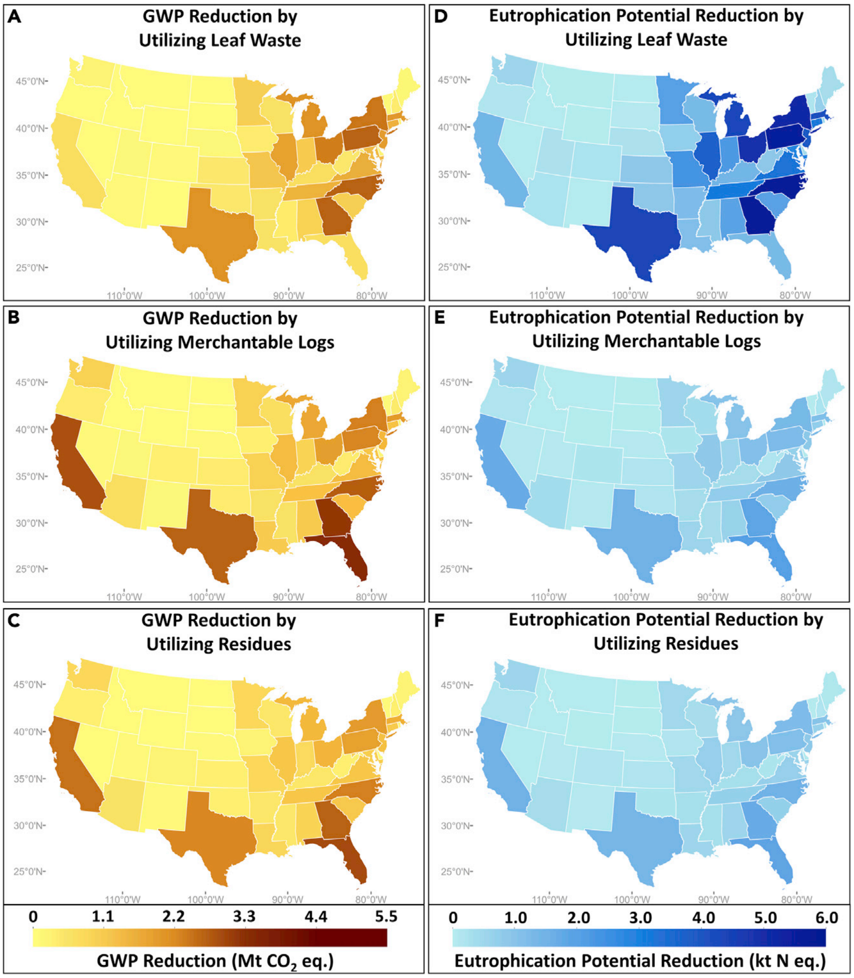


**Figure 3. Annual GWP and eutrophication reduction potential of scenario 5 of full utilization compared with scenario 1 of landfill**

(A–F) The annual life-cycle GWP and eutrophication potential reduction potential of scenario 5 compared with scenario 1 by three utilization pathways in scenario 5, namely utilizing leaf waste for CP, utilizing merchantable logs for lumber and chips, and utilizing residues for biochar. The results include (A) GWP reduction by utilizing leaf waste; (B) GWP reduction by utilizing merchantable logs; (C) GWP reduction by utilizing residues; (D) eutrophication potential reduction by utilizing leaf waste; (E) eutrophication potential reduction by utilizing merchantable logs; (F) eutrophication potential reduction by utilizing residues.

S4).[7,28,29,43] The detailed results of Figure 2 are available in Table S1.

## State-level life-cycle results

Figures 1 and 2 show the distinct environmental contributions of different tree waste types that have large spatial variations across the US. To quantify such geospatial variations, the state-level reductions of environmental burdens enabled by full utilization of urban tree wastes (scenario 5) compared with the worst case, landfilling (scenario 1), are presented in Figure 3. The reductions were estimated based on the average GWP and eutrophication potential of scenarios 5 and 1. The contribution of each of the three utilization pathways to total environmental reductions at the state level is shown in Figure S3.

Based on Figures 3A–3C, the states that show substantial GWP reductions (>6.0 Mt $CO_2$ eq./year) are majorly in the northern US (i.e., Illinois, Massachusetts, Michigan, New Jersey, New York, Ohio, and Pennsylvania), southeastern US (Florida, Georgia, and North Carolina), Texas, and California. This can be explained by the high urban tree waste availability in these regions (see Table S2). Among these states, northern states commonly have higher GWP reduction potential contributed by leaf waste utilization than the other two pathways. For example, in Pennsylvania, utilizing leaf waste reduces 3.8 Mt $CO_2$ eq./year, while utilizing merchantable logs and residues reduces 3.2 and 2.7 Mt $CO_2$ eq./year, respectively. Other states, such as Florida, California, and Texas, have GWP reductions mainly contributed by merchantable logs and residues rather than leaf waste, due to the lower portion of leaf waste in the urban tree waste (e.g., leaf utilization in California reduces to 1.0 Mt $CO_2$ eq./year, while utilizing merchantable logs and residues reduces 4.1 and 3.7 Mt $CO_2$ eq./year, respectively). This phenomenon can be explained by the regional differences in deciduous trees. Deciduous trees are the primary sources of leaf waste and are more common in the northern US. For example, the urban forest area covered by deciduous trees in Philadelphia (Pennsylvania) is 90.7%, compared with 52.4% in Gainesville (Florida) and 47.8% in Los Angeles (California).[7] Hence, this result sheds light on the importance of adaptively planning and managing the urban tree waste in different states based on the varied biomass availability.

Most states with large GWP reductions also have substantial eutrophication potential reduction (>5.0 kt N eq./year) except Florida and California, as shown in Figure 3D. The eutrophication potential reduction is dominated by utilizing leaf waste in those states (larger than 50%), since the other two utilizing pathways have much less effect on eutrophication potential reduction, as shown in Figure 2B.

In Figure 4, the total GWP reduction potential per year by full waste utilization in scenario 5 is shown at the level of main urban areas (see section "urban tree waste availability" for the details and definition of urban areas and Figure S4 for eutrophication potential results). The top 10 urban areas with the most considerable GWP reduction potential are highlighted in Figure 4, namely New York-Newark, Atlanta, Boston, Chicago, Philadelphia, Detroit, Miami, Los Angeles-Long Beach-Anaheim, Dallas-Fort Worth-Arlington, and Houston, in descending order. Some of these metropolitan areas already have city-wide strategies in place for utilizing urban tree waste. For example, in Chicago, the woody debris removed by the Bureau of Forestry of the

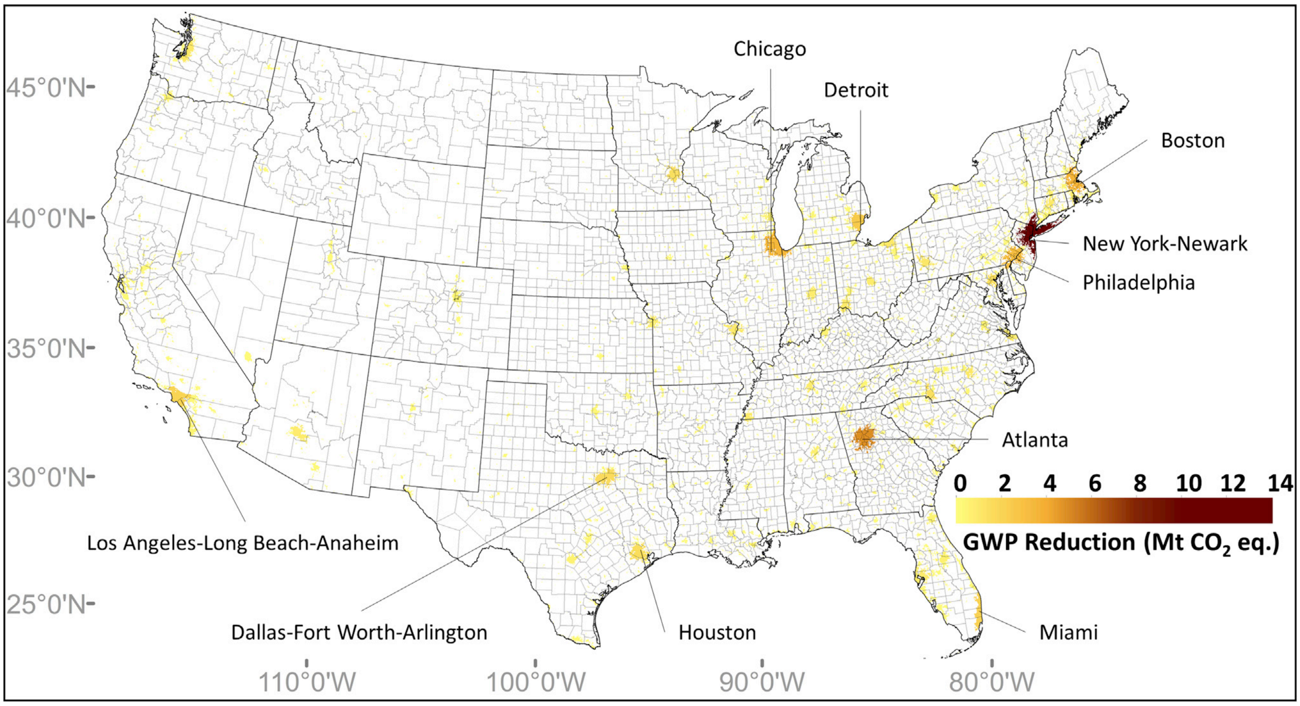


**Figure 4. Ten urban areas with the largest annual GWP reduction potential of scenario 5 for full utilization compared with scenario 1 for LF**
The annual life-cycle GWP reduction potential of scenario 5 compared with scenario 1. Top 10 urban areas with the largest GWP reduction potential are marked with names.

Department of Streets and Sanitation are 100% recycled or reused into logs and chips sold to vendors and other products (e.g., mulch).[59] In addition, Chicago launched the City of Chicago Waste Strategy, which aims to "minimize landfilling, improve recycling rates, drive new and innovative approaches for composting and materials reuse."[60] The City of Philadelphia provides free screened leaf compost, shredded wood mulch, and wood chips to residents.[61] Another example is the Baltimore Wood Project enhancing the utilization of urban wood waste to develop a diverse regional wood economy, promote sustainability, create jobs, and improve lives.[62]

## DISCUSSION

Based on the results of this study, composting, mulching, and recycling logs for lumber products that have already been promoted in some cities do bring reductions in GWP and eutrophication potential compared with landfilling or incineration. The quantity of environmental benefits depends on biomass availability and substitution products. Emerging application biochar brings additional benefits in carbon storage, soil improvement, and reducing fertilizer needs. Biochar has a growing market, especially for biochar made from low-value biomass.[63] Landfilling urban tree wastes leads to significant environmental burdens, primarily due to the $CH_4$ emissions in landfill gas and $NH_4^+$ to water.

Furthermore, this study shows the necessity of considering the end of life of products made from urban tree waste, especially for compost, lumber, and chips. Considerable carbon and nitrogen are released to the environment when the waste-derived products are applied to soil or when they reach the end of their lifetime (e.g., lumber sent to landfill). From this perspective, applications such as biochar that have a stable carbon structure and can be used for soil amendment is a win-win solution for the minimal environmental release at the end of life. In addition, this study considers variations in biomass compositions, process operations, and end-of-life emissions. These variations lead to varied results of GWP and eutrophication potential but do not change the comparative conclusions across five scenarios (Figure S1). Full utilization of urban tree wastes for different applications is highly likely to lead to the lowest GWP and eutrophication potential compared with other scenarios.

The urban tree waste utilization pathways explored in this study are aligned with circular economy principles. The five Rs principles of the circular economy include recover, recycle, repair, reuse, and reduce,[64,65] three of which are investigated in this study. Energy in tree waste is recovered by incineration, while biomass materials are recycled by converting tree wastes to valuable products, including compost, mulch, lumber and chips, and biochar. These products can be substitutes for virgin materials such as fertilizers and thus reduce associated environmental impacts.[66] This study demonstrates the benefits of applying circular economy principles to biomass wastes that are related to another emerging concept: bioeconomy. The importance of integrating circular economy and bioeconomy to avoid a linear business-as-usual approach has been discussed in the literature.[67,68] The findings in this study may inform future applications and the development of a circular bioeconomy in the urban environment.

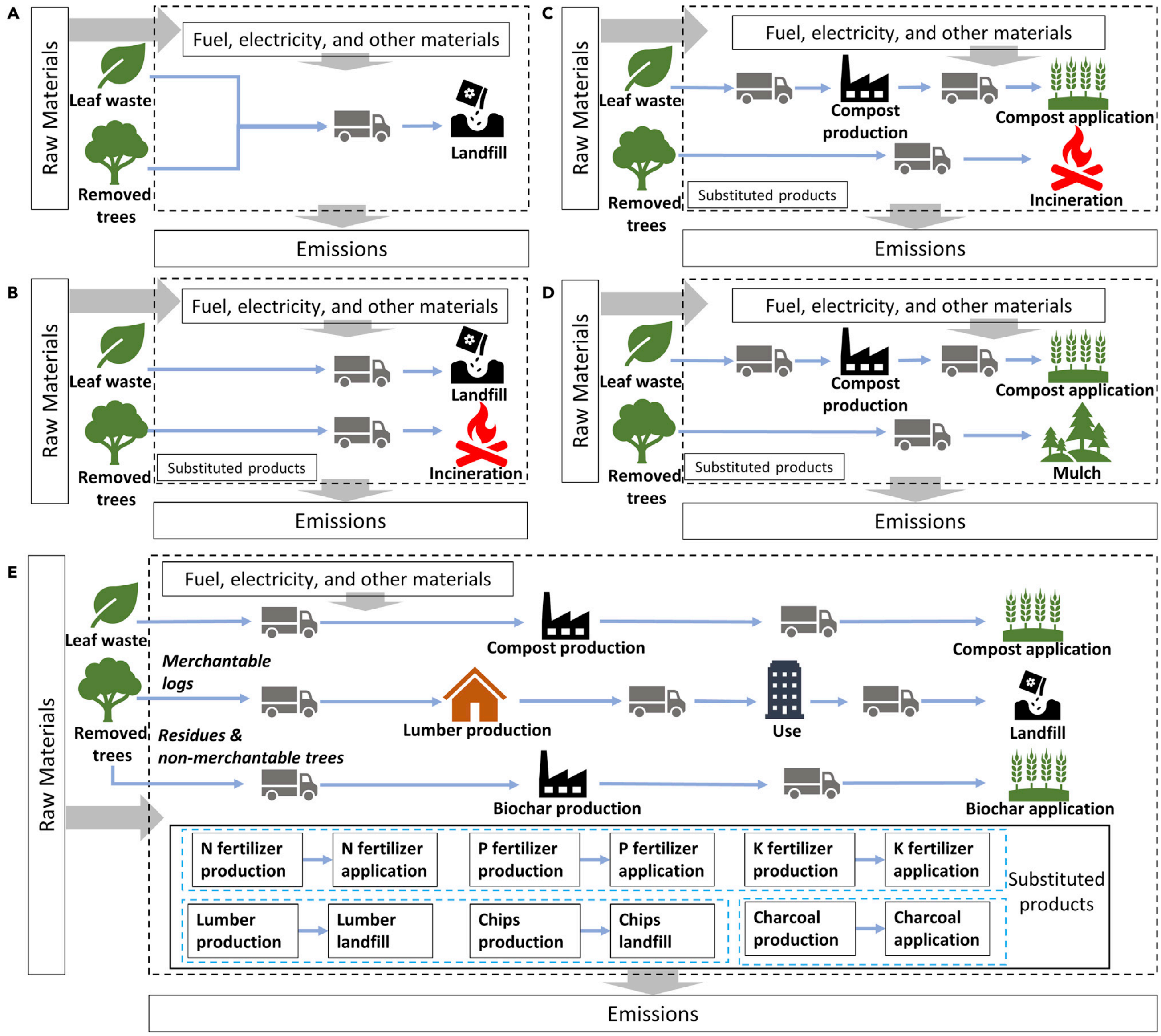


**Figure 5. System boundary of five scenarios**

(A–E) The detailed cradle-to-grave system boundary of five scenarios. (A) Scenario 1, landfill; (B) scenario 2, landfill and incineration; (C) scenario 3, compost and incineration; (D) scenario 4, compost and mulch; (E) scenario 5, full utilization.

In this study, five scenarios were developed to understand the potential environmental implications of utilizing urban tree waste (see section "scenario analysis" and Figure 5). Realizing these utilization scenarios requires the consideration of practical constraints and challenges. The first challenge is logistics, such as collecting urban tree waste from highly scattered sources. Each year, a substantial amount of urban tree waste is not used or collected in the US,[7] owing to improper dumping or disposal,[69] lack of municipal facilities collecting and dealing with urban tree waste,[69] low quality or defects of merchantable logs,[7] and other reasons. Joint efforts across different stakeholders, including individuals, communities, and cities, are needed to address this challenge. Examples include avoiding illegal dumping of yard waste, constructing community-based wood waste collection sites, developing city-wide urban tree waste management programs and guidelines, and identifying sustainable pathways for utilizing low-quality logs.[7,59,61,62,69] The second challenge is the lack of a mature market for utilizing urban tree waste.[7,69] Some efforts have been made to tackle this challenge by developing policies, market tools, and funding mechanisms for urban tree waste utilization.[37,60,70] For example, i-Tree Urban Wood Marketplace developed by the USDA Forest Service Forest Products Laboratory is a tool connecting removed tree users with sources of urban tree waste.[71] Other examples include the Urban and Community Forestry Program by USDA Forest Service,[72] Ash Utilization Options Project by Southeast Michigan Resource Conservation and Development Council,[37] and the Baltimore Wood Project.[62]

Further research on the infrastructure and market development for urban tree waste utilization is needed. Leveraging and coordinating with existing infrastructures and facilities (e.g., power plants, composting sites, lumber mills) requires a better understanding of socioeconomic and policy factors across different regions. Another future research direction is understanding the economic feasibility and social impacts of different utilization pathways. Tools such as techno-economic analysis and social LCA can be useful.[73] In the context of circular economy and bioeconomy, more future options for a circular use of urban tree biomass can be explored (e.g., converting to higher value-added chemicals), and the life-cycle environmental implications of these options will need to be assessed.

## Conclusion

A multi-scale cradle-to-grave LCA integrating process simulation and MCS was developed in this study to analyze the product-, state-, and national-level environmental implications of utilizing urban tree waste in the US. This study established process simulation models (i.e., composting, wood product manufacturing, biochar production) to provide the LCI data under the variations of parameters that are further used as inputs to MCS. At the national level, annual full utilization of various urban tree wastes to produce compost, lumber, chips, and biochar leads to 127.4–251.8 Mt $CO_2$ eq. reduction of GWP and 93.9–192.7 kt N eq. reduction of eutrophication potential compared with landfilling, 44.5–102.0 Mt $CO_2$ eq. and 59.6–135.6 kt N eq. lower than landfilling and composting, 6.0–17.0 Mt $CO_2$ eq. and 5.9–10.3 kt N eq. lower than composting and incineration, and 14.0–21.5 Mt $CO_2$ eq. and 7.0–36.8 kt N eq. lower than composting and mulching. Such reductions have considerable geospatial variations across the US at the state and city levels, depending on the regional availability of different urban tree waste types. Based on the process-level comparisons, composting and mulching are more environmentally favorable than landfilling and incineration, but using merchantable logs for lumber production and urban tree residues for biochar are even better given the benefits of long-term carbon storage and virgin material substitutions. Lumber already has a mature market, while the biochar market is still growing, but biochar is widely regarded as a negative-emission technology for climate change mitigation. This study provides transparent LCI data and parametric process-based models for different end-of-life pathways, contributing to the LCA and broad waste treatment modeling communities. This study highlights the strong need to divert urban tree wastes from landfilling and incinerations to valuable utilization, and identifies the pathways with the largest potential in reducing GWP and eutrophication from a life-cycle perspective. Further research is needed to investigate the infrastructure enhancement and market development for urban tree waste utilization, as well as to explore the economic feasibility and social impacts of different utilization pathways.

## EXPERIMENTAL PROCEDURES

### Resource availability

#### *Lead contact*

Please contact the lead contact, Dr. Yuan Yao (y.yao@yale.edu), for information related to the data and code described in the "experimental procedures" section.

#### *Materials availability*

No materials were used in this study.

#### *Data and code availability*

All original data have been deposited and are publicly available at Zenodo: https://doi.org/10.5281/zenodo.6760986. Any additional information required to reanalyze the data reported in this paper is available in this paper's supplemental information and available from the lead contact upon request.

### Methodology

In this work, a cradle-to-grave LCA was conducted following ISO Standard 14040 series[74] to evaluate the life-cycle GWP and eutrophication of the US urban tree waste utilization (see Figure 5). The functional unit is processing 1-year US urban tree waste at the national level, and 1 ODMT of urban tree waste at the product level. As shown in Figure 5E, the system boundary includes the collection and transportation; production of compost, lumber, biochar, and end of life of the products; along with the production and end of life of substituted products. This study developed five scenarios, including scenario 5 for full utilization of urban tree wastes, and scenarios 1–4 for urban tree waste that is landfilled, incinerated, or produced into compost and mulch. Process simulations were developed to generate the LCI data. Specifically, biochar production was modeled in ASPEN Plus, and the rest of the production and end-of-life stages were modeled in Excel. The LCI data of upstream production of fuels and chemicals were collected from ecoinvent database and GREET 2020.[46,56] GWP was calculated using IPCC 2013 characterization factor and eutrophication potential was estimated using TRACI 2.1.[33,57,75] Biogenic and fossil carbon was tracked separately. The variations of key process parameters were collected from the literature. The following sections briefly discuss each life-cycle stage and major assumptions.

### Urban tree waste availability

The average annual availability of urban tree waste at the state level was evaluated based on Nowak et al.[7] The detailed availability data with standard deviation are shown in Table S2. Normal distribution was assumed and used in MCS.[7] This study defines the urban areas using the criteria established by the US Census Bureau (USCB), including urbanized areas (50,000 or more people per urban area) and urban clusters (2,500 or more people per urban area).[7,76] The Geographic Information System (GIS) data of urban areas for each state adopt the data from USCB.[76] The state-level urban tree waste data were downscaled to the urban area level based on population.[76]

### Compost production and end of life

Leaf waste is collected from urban areas, transported to composting plants, produced into compost, and applied as soil amendments.[21] The typical processes include collection, transportation, grinding, decomposition, curing, screening, and field application (see Figure S5). The LCI data of the composting plant were derived from the process model developed by the authors in Excel. The values and variations of the process parameters were collected from the literature (see Table S3).[7,29,43–45,56,77–85] Upstream burdens of producing fuels, chemicals, and other materials were included and the LCI data were collected from ecoinvent database and GREET 2020.[46,56]

The composition data of urban leaf waste were collected from literature (detailed references are provided in Table S3). Typically, high entrained ash content is expected for urban leaf waste, and the ash content can vary upon different collecting methods (e.g., sweeping, vacuum) and sources (e.g., residential areas, city centers).[44] In composting plants, wheel loaders transfer leaf waste to grinders.[86] The grinding process reduces the size of leaf waste to around 80 mm for a better decomposition result.[84] The decomposition process commonly takes 2–4 weeks under controlled moisture, temperature, and aeration conditions. After the decomposition, the curing process allows compost to further decompose within generally 6–9 weeks.[83,84] During the decomposition and curing process, the organic matters degrade and release biogenic $CO_2$, $CH_4$, $NH_3$, $N_2O$, and $NO_x$ that contribute to GWP and eutrophication potential, where $CO_2$ is the predominate emission. The variations of these gas emissions were based on the literature data collected and documented in Table S3.[77–79,82] The produced leachate is used to water the decomposition and curing streams to keep the moisture.[84] The final screening sieves out the compost under 15 mm and the refuse (e.g., large pieces that are hard to degrade) are sent to landfill.[84,86] The data of diesel and electricity were collected from the literature (see Table S3).[82–85]

The compost produced is distributed for field application. Carbon and nutrients may be released at the end of life of compost after soil application. Biogenic $CO_2$ emissions due to the compost decay on land were estimated by an exponential decay model (see Note S1).[17,87] For nutrient-related emissions in this study, several models and methods were adopted to estimate the air emissions of $NH_3$ (Agrammon model[88]), $N_2O$ (method by Nemecek et al.[89]), and $NO_x$ (method by Nemecek et al.[89]), and water emissions including nitrate and phosphorus (method by Brockmann et al.[90]) (see Note S1 for modeling details).[88–91]

### Lumber production and end of life

Merchantable logs (under bark) are transported to lumber mills that have[17] processes including sawing, kiln drying, planing, and energy generation for the dry kiln (see Figure S6).[92,93] Then lumber is distributed to market and finally landfilled after use.[94] In this study, the variations of process parameters in lumber production and end of life were collected from the literature and shown in Table S4. GHG emissions were estimated based on the upstream environmental burdens (from ecoinvent database[56] and GREET 2020[46]) and the emission factors of fuels (see Note S2).

When they arrive in the lumber mills, logs are first debarked. The mass fraction of barks is documented in Table S4.[95] Then logs are sawn into wet lumber and two byproducts: slabs/chips and wet sawdust. The wet lumber mass was estimated by the lumber yield in sawing (see Table S4).[93,96–108] For two byproducts, it was assumed that slabs/chips took 82.1 wt % and wet sawdust took 17.9 wt % of sawing residues.[108] In this study, all the slabs/chips were assumed to be further chipped and sold to pulp mills as byproducts.[109–111] Bark and wet sawdust, along with dry shavings/chips and sawdust from the planing process (see below about the final planing process), were used for energy generation.

Wet lumber was dried in a kiln operating at 90°C–120°C (dry bulb temperature) to reach the targeted moisture content.[112,113] In this study, the energy source for drying is mill residues (bark, wet sawdust, dry planing shavings/chips and sawdust). In energy generation, energy consumption for kiln drying was calculated as heat demand divided by the overall energy efficiency for energy generation and drying that was collected from the literature and is shown in Table S4.[93,98,112] When energy from mill residues is not sufficient, natural gas is used; when energy is excessive, then mill residues are used for power generation (see Note S2).[17]

Dried lumber undergoes the final planing processes to produce finished lumber.[114] Planing generates dry shavings/chips and sawdust that are sent to energy generation.[93,114] Then finished lumber is stacked and ready for transportation to the market.

Lumber can be used as structural construction materials (e.g., wall framing, floor framing), furniture (e.g., tables, cabinets), and industrial products (e.g., pallets).[115,116] After the lumber products reach the end of their lifetimes, they are commonly sent to landfill.[54,55] The lifetime of the lumber was assumed to range from 20 to 60 years.[117–122] In this study, the time horizon for GWP is 100 years.[57] Hence, the landfill decay time for GWP accounting equals 100 years minus the time length of lumber usage phase. GHG and nitrogen- or phosphorus-containing emissions of lumber landfill were estimated using the method documented in section "urban tree waste landfill."

### Biochar production and end of life

Non-merchantable trees and logging residues from merchantable trees are collected and transported to produce biochar through pyrolysis (see Figure S7 for the process diagram). In this study, a process-based simulation model of the biochar plant was established in ASPEN Plus based on the previous work of Liao et al.[28] The mass and energy balance was derived from the ASPEN Plus model for LCI data collection (see Note S3 for details). This model can quantify the impacts of variations in feedstock compositions (i.e., carbon content, C/H ratio, C/O ratio, ash content, and moisture content) on mass and energy balances. The statistical distributions of biomass composition were assessed based on the literature data and are documented in Table S5.[28,123]

The biomass feedstocks for biochar production were collected from the urban area and transported to the biochar plant within 52–87 km (see Table S5).[124] The feedstock size was reduced to around 50 mm by initial grinding.[125] Then the feedstocks were dried in the rotary drum dryer to reach the moisture content of 10% (wet basis) in a temperature range of 160°C–180°C.[126] The dried biomass was ground in the hammer mill as a secondary grinding process to reach a particle size of 2.5–3.8 mm before entering the pyrolyzer. The feedstocks were pyrolyzed in the fluidized bed reactor at 500°C and 1 atm for 60 min in the ambient of nitrogen.[28] In ASPEN Plus, pyrolysis kinetics were modeled through the multistep reaction mechanism method (see Note S3 for details).[28,127] Then biochar and the gaseous product were separated through multi-stage cyclones. In this study, the gaseous products from pyrolysis were combusted in the combustor to produce heat for the rotary drum dryer and pyrolysis reactor. If the heat from the gaseous product was not sufficient, natural gas was combusted. In the biochar plant, GHG emissions contain biogenic GHG emissions from combusting pyrolytic gaseous products and potential fossil GHG emissions from combusting natural gas. Diesel and electricity consumption in the biochar plant were considered along with their upstream burdens (see Note S3 and Table S6).

Biochar was distributed for soil application and transportation distance is documented in Table S5.[56] After the application, biochar stays in the soil and decays slowly. Previous studies show that the mean residence time of biochar in soil can vary from several centuries up to 2,000 years.[128] Mean residence time is a common indicator describing the stability of biochar in soil and equals the half-life time multiplied by ln2.[129] This study uses the exponential model that uses the decay rate (inverse of mean residence time) to simulate the GHG emissions from on land decay of biochar (see Note S3). The data range of biochar decay rate was collected from the literature and is documented in Table S5.[41,128–131]

### Urban tree waste landfill

Landfilled urban tree waste releases GHG emissions and N-containing emissions through the decay process.[47,132,133] For GHG emissions, landfill decay emits $CO_2$ and a significant amount of $CH_4$, which has a much higher GWP characterization factor than $CO_2$ (GWP-100 for $CH_4$ is 28).[57,134] The GHG emissions from landfill decay in this study were estimated based on the IPCC First Order Decay method (see Note S4 and Table S7).[17,47] For N-containing emissions, landfill decay generates gaseous ammonia and leaches ammonium,[133] which were estimated based on the experimental data from the literature (see Note S4).[132,133,135]

### Urban tree waste incineration and mulch production

Besides landfilling, removed trees can be used to generate power.[7,12] Incinerating removed trees generates power to replace the market electricity. The quantity of power generated was estimated based on the lower heating value of woody biomass multiplied by the electricity generation efficiency collected from the literature (see Note S5 and Table S7).[17,136–140] The emission factors of combusting woody biomass were collected from GREET 2020.[46]

Removed trees have also been used to produce mulch.[69,141] Mulch is typically produced by chipping the waste wood and residues and covering the top of the soil. After soil application, mulch decays on land and releases GHG emissions and nutrient-related emissions (see Note S5).

### Product substitution

The potential substitution benefits are commonly considered in LCAs for bio-based products.[11,18,79,83,126] Compost can replace mineral fertilizers that contain nitrogen, phosphorus, and potassium. The lumber made from urban tree waste can replace lumber and chips sold in the market, including both hardwood and softwood products. Biochar can replace charcoal as soil amendments. The LCI data of upstream production of substituted products in the market were collected from the ecoinvent database (see Table S8 for specific process documentation).[56] The end of life of products substituted is also included to keep a consistent system boundary of cradle to grave. The market mix of different mineral fertilizers was based on the US fertilizer consumption data from the USDA (see Table S9).[142] Because the substitution of nitrogen and phosphorus in compost and mineral fertilizers is not 1:1,[143] this study estimated the substitution ratio by calculating the average mineral fertilizer equivalent for compost and mineral fertilizers.[90,143] The modeling details are documented in Notes S6–S9.

### Scenario analysis

Table 1 summarizes the scenario analysis settings. These five scenarios were developed based on varied levels of utilizing urban woody biomass suggested by the USDA Forest Service in their sustainability assessment guideline for the urban forest.[35] The purpose of the scenario analysis is to understand the

**Table 1. Scenario analysis settings**

| Scenario | Leaf waste | Merchantable logs | Residues[a] |
|---|---|---|---|
| Scenario 1, LF | landfill | landfill | landfill |
| Scenario 2, LF and IC | landfill | incineration | incineration |
| Scenario 3, CP and IC | compost | incineration | incineration |
| Scenario 4, CP and MC | compost | mulch | mulch |
| Scenario 5, UL | compost | lumber and chips | biochar |

IC, incinerating; LF, landfilling; MC, mulching; UL, composted leaf waste.
[a]Residues in this study refer to non-merchantable trees and logging residues from merchantable trees.

environmental implications of processing the urban tree waste with maturely proven technologies (e.g., landfilling, composting, incinerating, mulching, producing lumber) and emerging technology (i.e., producing biochar). Scenario 1 represents a low-utilization case where urban tree waste is landfilled, or the worst scenario that can provide a bottom line for the results. As mentioned in section "results," scenarios 2–4 aim to explore the combinations of different options mentioned by the guideline for fair (e.g., chips and mulch) and good utilization (e.g., energy recovery, compost). Scenario 2 represents a situation where removed trees are incinerated for power generation, but leaf waste is still landfilled. Compared with scenario 2, scenario 3 composts the leaf waste to substitute the mineral fertilizers instead of landfilling leaf waste. In scenario 4, leaf waste is composted, and removed trees are mulched as a common practice.[36] In scenario 5, this study seeks to simultaneously maximize the waste utilization and carbon storage in the products, which stands for optimal utilization.[35] In scenario 5, leaf waste is composted, merchantable logs are produced into lumber and chips, and residues are converted into biochar.

To address the variations in system parameters, this study adopted MCS. The parameter values are independently generated 500 times based on the statistical distributions shown in Tables S3–S5. These independently generated parameter values were used as input to the process models to produce LCI data that were used to calculate the LCA results for each scenario.

## SUPPLEMENTAL INFORMATION

Supplemental information can be found online at https://doi.org/10.1016/j.oneear.2022.07.001.

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## AUTHOR CONTRIBUTIONS

K.L. and Y.Y. designed the study, conceptualized the manuscript, and wrote the manuscript. K.L. collected the data and performed the analysis. K.L. and B.Z. processed the data and created the figures for this study. Y.Y. supervised this study. All authors edited the manuscript and were responsible for the final manuscript.

## DECLARATION OF INTERESTS

The authors declare no competing interests.

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## Supplemental information

# Circular utilization of urban tree waste contributes to the mitigation of climate change and eutrophication

**Kai Lan, Bingquan Zhang, and Yuan Yao**

**Note S1. Compost end-of-life emissions**

The end-of-life emissions of compost on land include carbon emissions (e.g., $CO_2$ and $CH_4$) and nutrient-related emissions (e.g., $NH_3$, $N_2O$, $NO_x$, nitrate, phosphorus), which contribute to GWP and eutrophication potential results. The carbon emissions from compost decay on land are calculated using Equation S1:[1]

$$E_t = [(CM_{t-1} - CM_t) \times CC_{compost} \times F_{air}] \times \frac{44}{12} \quad \text{(Equation S1)}$$

$E_t$ is total carbon emission from decay in year $t$; $CM_t$ and $CM_{t-1}$ is the oven dry mass of compost remained in the year $t$ and the previous year ($t$-1); $CC_{compost}$ is the carbon content of compost. There are two destinations for the carbon loss, either emitted to air or to soil. In this study, the fraction of residue carbon loss emitting to air as $CO_2$, $F_{air}$, was assumed to be 76%.[1] Then $CM_t$ is derived from the exponential model as shown in Equation S2:[2]

$$CM_t = CM_o \times e^{-k_c t} \quad \text{(Equation S2)}$$

where $CM_o$ is the initial oven dry mass of compost.[3] The decay rate, $k_c$, can vary due to varied climate conditions and soil conditions. In this study, the decay rate value and variations were collected from the literature and assumed to be a uniform distribution between 0.02 and 0.08.[4]

For nutrient-related emissions, this study adopted several models and methods to evaluate the results.[5–8] The ammonia emissions were evaluated based on the Agrammon model as shown in Equation S3.[5] $E_{NH3,compost}$ is $NH_3$ emission (kg $NH_3$/kg N input); $TAN$ is total ammonium nitrogen of the compost (kg N); $TN$ is total nitrogen of the compost (kg N); $EF_{NH_3}$ is the emission factor of $NH_3$-N expressed as the portion of $TAN$ (or-say how much $TAN$ can be released as $NH_3$). $c_x$ is the correction factor referring to the corrections caused by equipment use, weather conditions, and application timing. In this study, ($TAN/TN$) adopted the experiment data in the literature that range from 0.02 to 0.10 for mature compost.[9,10] $EF$ is assumed to be 0.8 based on the literature data.[6] $c_x$ assumes to be 1 as the default number due to the lack of data.[5,6]

$$E_{NH_3,compost} = \frac{TAN}{TN} \times EF_{NH_3} \times c_x \times \frac{17}{14} \quad \text{(Equation S3)}$$

The amount $NO_3^-$ leaching to the water is estimated based on the study by Brockmann et al.[7] where the emission factor $E_{NO_3^-,compost}$ is 0.49 kg $NO_3^-$/kg N input.

$N_2O$ emission to air is evaluated by the model given by Nemecek et al. as shown in Equation S4.[6] $E_{N_2O,compost}$ is $N_2O$ emission (kg $N_2O$/kg N input). Then $NO_x$ emission $E_{NO_x,compost}$ is derived from $N_2O$ emission as shown in Equation S5.[6]

$$E_{N_2O,compost} = (0.01 + 0.01 \times \frac{14}{17} \times E_{NH_3,compost} + 0.0075 \times \frac{14}{62} \times E_{NO_3^-,compost}) \times \frac{44}{28} \quad \text{(Equation S4)}$$

$$E_{NO_x,compost} = E_{N_2O,compost} \times 0.21 \quad \text{(Equation S5)}$$

Phosphorus leaching to water and running off to the river is a small amount compared to the input amount. In this study, phosphorus emission $PO_4^{3-}$ $E_{PO_4^{3-},compost}$ of compost (kg $PO_4^{3-}$/kg P input) adopts the value 0.0067 from the study by Brockmann et al.[7]

## Note S2. Lumber production

The energy demand from the drying kiln was estimated by the total heat to evaporate water out of wood, $TH_{drying}$, and the overall energy efficiency for energy generation and drying, $\eta_{total}$, as shown in Equation S6.

$$Total\ energy\ demand\ = \frac{TH_{drying}}{\eta_{total}} \qquad \text{(Equation S6)}$$

$TH_{drying}$ was calculated based on the study by Bowyer et al.,[11] where the desorption heat was presented in MJ energy needed for 1 kg water evaporated out of wood in the drying process (see the table below); $\eta_{total}$ data range was from the literature and shown in Table S4.

**Heat demand of evaporating water from wood in varied moisture content**

| Moisture Content (%, *dry basis*) | 0 | 10 | 20 | 30 | 40 | 50 | 60 | 70 | 80 | 90 | 100 | 120 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Desorption Heat (MJ/kg water) | 3.489 | 2.559 | 2.384 | 2.326 | 2.326 | 2.326 | 2.326 | 2.326 | 2.326 | 2.326 | 2.326 | 2.326 |

The total energy demand is met by the sum of the lower heating value (LHV) of the available mill residues. For LHV of mill residues, varied moisture content can lead to varied LHV as shown in Equation S7.[12,13] *HHV* (MJ/kg) is higher heating value; *MC* moisture content (*dry basis*) of mill residues; *H* hydrogen content percentage in fuel (assuming 6).[12–15] The LHV for natural gas is 47.1 MJ/kg.[16] $CO_2$ emissions from burning mill residues are all biogenic. If the mill residues are excessive for the total heat demand, then the excessive amount of mill residues are utilized for power generation.

$$LHV = HHV - 0.0245((\frac{MC}{1+MC}) \cdot 100 + 9H) \qquad \text{(Equation S7)}$$

The potentially generated power (by potential excessive mill residues) was estimated by the total LHV of woody biomass multiplied by the electricity generation efficiency. The total LHV of woody biomass used the same method shown in Equation S17. The electricity generation efficiency data range was collected from the literature and shown in Table S7. The emission factors of combusting woody biomass in hog fuel boilers were based on the GREET 2020 values.[16] The environmental burdens of electricity adopted the data of the U.S. stationary mix electricity from the GREET 2020.[16]

## Note S3. Biochar production and end-of-life emissions

The ASPEN plus model was built based on the pyrolysis modeling work of Liao et al.[17] and we added a pretreatment area that includes the initial grinding, drying, and secondary grinding. In the drying stage, the flue gas from the combustor was utilized to heat the inlet air of the dryer to reach around 200 °C. The pyrolysis reactions were simulated through the multi-step reaction mechanism (MSRM) where the biomass feedstock is decomposed into major lignocellulosic components (cellulose, hemicellulose, and lignin) and then go through a series of reactions.[18] In this study, the model compounds were selected as glucose for cellulose, xylose for hemicellulose, lignin-C, lignin-O, and lignin-H for lignin.[18] As the biomass composition data were collected from the ultimate analysis (e.g., carbon content, oxygen content), the triangular method was applied to calculate the mass fraction of each model compound.[19] After the contents of five model compounds were determined, the kinetic model was established based on MSRM with a series of reactions. The detailed pyrolysis kinetic model reactions and parameters can be found in Liao et al.[17] To model the series of pyrolysis reactions in ASPEN Plus, four reactors are set in sequence. The first reactor (RYield) decomposed the biomass into five model compounds and ash; the second reactor (RBatch) conducted the primary pyrolysis kinetic reactions; the third reactor (RCSTR) calculated the tar cracking reactions; the fourth reactor reacted the remaining metaplastic components into biochar.[17] Then the gas-phase products were separated and sent to the combustor for energy recovery. The combustor was modeled by using RStoic reactor in ASPEN Plus with assumed 80% efficiency. Key process parameter assumptions are shown in Table S6.

In the ASPEN Plus model, input feedstock amount was normalized to 1 oven dry metric ton (ODMT) of wood residues. Upon varied feedstock compositions in MCS, the following LCI data were collected: biochar output with biochar composition data, nitrogen consumption, natural gas consumption, and flue gas emissions. The electricity and diesel consumption of unit operations are displayed in Table S6. The upstream burdens of electricity and diesel adopted the data from the GREET 2020.[16]

After biochar is applied to the field, the biochar starts the slow decay process. This study adopts the exponential decay model for the biochar after soil application,[20] as shown in Equation S8. The data range of decay rate for biochar, $k_{biochar}$, was converted from the mean residence time (MRT) ($k_{biochar}$=1/MRT) for biochar collected from the literature and shown in Table S5.[20]

$$E_t = \left[CM_o \times \left(e^{-k_{biochar}(t-1)} - e^{-k_{biochar}t}\right) \times CC_{biochar}\right] \times \frac{44}{12} \qquad \text{(Equation S8)}$$

The biochar is assumed to substitute charcoal use as soil amendments.[21–23] The LCI of charcoal production were collected from the ecoinvent database.[24] In this study, the substituted charcoal is assumed to undergo the same decay process as biochar.

**Note S4. Urban tree waste landfill**

In this study, the landfill GHG emissions (mainly $CH_4$ and $CO_2$) were evaluated following the Intergovernmental Panel on Climate Change (IPCC) First Order Decay (FOD) method.[25] In year *t* after landfill, the accumulative $CH_4$ generated by landfill decay is presented by Equations S9 and S10.

$$C_{decomposed} = W \cdot DOC \cdot DOC_f \cdot (1 - e^{-kt}) \qquad \text{(Equation S9)}$$

$$CH_{4\ generated} = [(C_{decomposed} \cdot MCF \cdot F \cdot 16/12) - R] \cdot (1 - OX) \qquad \text{(Equation S10)}$$

In Equation S9, $C_{decomposed}$ is the accumulative decomposed carbon mass from year 0 to year *t*; *W* is the mass of deposited wood waste (*wet basis*); *DOC* is degradable organic carbon of wood waste which is the mass fraction of cellulose and hemicellulose; $DOC_f$ is the faction of *DOC* that can decompose; *k* is the reaction constant which can be estimated by landfill decay half-life time.[26] In Equation S10, *MCF* is the $CH_4$ correction factor which is determined by the site management (e.g. disposal depth in the soil, anaerobic conditions);[25] *F* is the volume fraction of $CH_4$ in landfill gas released; *R* is the total recovered $CH_4$ by energy recover device, which largely depends on landfill site management practice.[26] This study adopts the IPCC default value of *R* to be 0.[25] *OX* is the average oxidation factor describing the fraction of methane oxidized in the soil or covering materials.[26] In this study, the key parameters and their ranges related to $CH_4$ emissions followed the uncertainty analysis given by IPCC (see Table S7).[25–28] In this study, *DOC* is estimated by the average carbon content of the wet tree waste, or say can be derived by using wood moisture content (*dry*) and carbon content (*dry*) in this study.[29] It is worth noting that following the IPCC guidelines, if *DOC* includes lignin carbon mass, then $DOC_f$ should assume to use a value of 0.5–0.6.[26] $CO_2$ emissions by landfill are largely affected by the $CO_2$ equilibrium between atmosphere and soil along the long decay period. Hence, this study estimated the $CO_2$ emissions of wood waste landfills by using the experimental data on the volume rate of $CH_4$ to $CO_2$ [30] but can be further modified by future researchers.

For nitrogen-containing emissions, the ammonification process turns a portion of the total nitrogen of the tree wastes into $NH_3$ or $NH_4^+$.[31] The detailed mixture portion of $NH_3$ and $NH_4^+$ depends on the landfill site pH conditions. The formation of $NO_3^-$ in the landfill decay process largely depends on the anaerobic conditions and is typically low in anaerobic conditions.[31,32] In this study, the percentage and the corresponding range of total nitrogen in the tree waste that converts to $NH_3$, $NH_4^+$, and $NO_3^-$ were estimated based on the literature data (see Table S7).[31,33,34]

## Note S5. Urban tree waste incineration and mulch end-of-life

The removed trees can be incinerated to generate power. The removed trees are chipped and then transported to the incineration plant. The generated power was estimated by the total LHV of woody biomass multiplied by the electricity generation efficiency. The total LHV of woody biomass used the same method shown in Equation S17. The electricity generation efficiency data range was collected from the literature and shown in Table S7. The emission factors of combusting woody biomass in hog fuel boilers were based on the GREET 2020 values.[16] The electricity credit data adopted the U.S. stationary mix electricity from the GREET 2020.[16] The transportation distance assumes the same range as landfill.

Mulch from chipped removed trees follows a decay process after application. In this study, the decay of mulch is assumed to follow the same decay model of compost as shown in Note S1.

## Note S6. Nitrogen fertilizer substitution

The life cycle emission *i* of substituted nitrogen fertilizers, $TE_{substituted\ N,i}$, were decided by Equation S11. *TN* is the total nitrogen in compost as mentioned in Note S1; $MFE_N$ is the average mineral fertilizer equivalent (kg N) in mineral fertilizers that can substitute 1 kg N in compost; $E_{N,production,j,i}$ is the emission *i* of producing and transporting fertilizer *j* that contains 1 kg N to the field; $E_{N,EOL,j,i}$ is the end-of-life emission *i* of fertilizer *j* that contains 1 kg N; $MMP_{N,j}$ is the market mixture percentage of nitrogen fertilizer *j*. Then Equations S12-S19 explain how to determine the parameters in Equation S11.

$$TE_{substituted\ N,i} = (-1) \times TN \times MFE_N \times \sum\left(\left(E_{N,production,j,i} + E_{N,EOL,j,i}\right) \times MMP_{N,j}\right) \quad \text{(Equation S11)}$$

The market mixture percentage of nitrogen fertilizer *j*, $MMP_{N,j}$, in the U.S. was estimated based on the market consumption data given by the U.S. Department of Agriculture (USDA), as shown in Table S9.[35]

To determine how much N in mineral fertilizers can be substituted by 1 kg N in compost, Equation S12 shows the average mineral fertilizer equivalent $MFE_N$ (kg N) in mineral fertilizers.[7,36] $PAN_{compost}$ is the plant available nitrogen content in compost which is assumed to be 0.150–0.246 based on the literature;[7] $c_{N,mineral}$ is the average nitrogen loss of mineral fertilizers after application. $c_{N,mineral}$ can be derived by Equation S16 based on the nitrogen-related emissions of substituted mineral fertilizers.[7] $c_{Phosphorus}$ is the correction factor due to the nitrogen content in substituted phosphorus fertilizers (most phosphorus fertilizers contain nitrogen). $c_{Phosphorus}$ is estimated to be the total N of total nitrogen fertilizers divided by total N of total nitrogen and phosphorus fertilizers, and the value is 0.93 in this study.[35]

$$MFE_N = PAN_{compost}/(1 - c_{N,mineral}) \times c_{Phosphorus} \quad \text{(Equation S12)}$$

$E_{N,production,j,i}$ (per kg N in fertilizer) for nitrogen fertilizer listed in Table S9 used the data from ecoinvent database (process names shown in Table S7).[24]

$E_{N,EOL,j,i}$ (per kg N in fertilizer) mainly include $NH_3$, $N_2O$, $NO_x$, nitrate as emission *i*, and is determined by the models and methods shown below. $E_{N,EOL,j,NH_3}$ (kg $NH_3$/ kg N in fertilizer) adopted the model presented in the EMEP/EEA air pollutant emission inventory guidebook 2009 by European Environment Agency as shown in Equation S13.[8] $EF_{j,NH_3}$ is the emission factor for $NH_3$ for fertilizer *j* (kg $NH_3$/ kg N in fertilizer); $p_{alk}$ is the portion of land with soil pH>7 and assumed to be 0.25 based on the survey results by Holmgren et al.[37]; $c_{NH_3,j}$ is the correction factor for soil pH for fertilizer *j*. The values of $EF_{j,NH_3}$ and $c_{NH_3,j}$ for fertilizer *j* can be found in EMEP/EEA air pollutant emission inventory guidebook 2009 4.D Crop production and agricultural soils Table 3-2 by European Environment Agency.[8] Note that the average spring temperature for the U.S. was assumed to be 11°C.[38]

$$E_{N,EOL,j,NH_3} = EF_{j,NH_3} \times (1 - p_{alk} \times (1 - c_{NH_3,j})) \quad \text{(Equation S13)}$$

$E_{N,EOL,j,NO_3^-}$ (kg $NO_3^-$ / kg N in fertilizer) leaching to the water was 0.49 kg $NO_3^-$/kg N input based on the study by Brockmann et al.[7] $E_{N,EOL,j,N_2O}$ and $E_{N,EOL,j,NO_x}$ used the same methods shown in Equation S4 and S5, respectively, shown in Equations S14 and S15.[6]

$$E_{N,EOL,j,N_2O} = (0.01 + 0.01 \times \frac{14}{17} \times E_{N,EOL,j,NH_3} + 0.0075 \times \frac{14}{62} \times E_{N,EOL,j,NO_3^-}) \times \frac{44}{28} \quad \text{(Equation S14)}$$

$$E_{N,EOL,j,NO_x} = E_{N,EOL,j,N_2O} \times 0.21 \quad \text{(Equation S15)}$$

After $E_{N,EOL,j,i}$ is derived for each fertilizer, $c_{N,mineral}$ can be assessed to present the average nitrogen loss after applying mineral fertilizers as shown in Equation S16.[7]

$$c_{N,mineral} = \sum(\left(E_{N,EOL,j,NH_3} \times \frac{14}{17} + E_{N,EOL,j,NO_3^-} \times \frac{14}{62} + E_{N,EOL,j,N_2O} \times \frac{28}{44} + E_{N,EOL,j,NO_x} \times 0.42\right) \times MMP_{N,j}) \quad \text{(Equation S16)}$$

**Note S7. Phosphorous fertilizer substitution**

The life cycle emission *i* of substituted phosphorus fertilizers, $TE_{substituted\ P,i}$, were decided by Equation S17. *TP* is the total phosphorus in compost; $MFE_P$ is the average mineral fertilizer equivalent (kg P) in mineral fertilizers that can substitute 1 kg P in compost; $E_{P,production,j,i}$ is the emission *i* of producing and transporting fertilizer *j* that contains 1 kg P to the field; $E_{P,EOL,j,i}$ is the end-of-life emission *i* of fertilizer *j* that contains 1 kg P; $MMP_{P,j}$ is the market mixture percentage of phosphorus fertilizer *j*. Then Equations S17 and S18 explain how to determine the parameters in Equation S17.

$$TE_{substituted\ P,i} = (-1) \times TP \times MFE_P \times \sum((E_{P,production,j,i} + E_{P,EOL,j,i}) \times MMP_{P,j}) \quad \text{(Equation S17)}$$

The market mixture percentage of phosphorus fertilizer *j*, $MMP_{P,j}$, in the U.S. was shown in Table S9.[35] $MFE_P$ (kg P) is derived from Equation S18.[7] $c_{P,mineral}$ is the average phosphorus loss of mineral fertilizers after application. $c_{P,mineral}$ can be derived by Equation S19 based on the phosphorus-related emissions of substituted mineral fertilizers.

$$MFE_P = 0.95/(1 - c_{P,mineral}) \quad \text{(Equation S18)}$$

$E_{P,production,j,i}$ (per kg P in fertilizer) for phosphorus fertilizer listed in Table S9 used the data from ecoinvent database (process names shown in Table S8).[24]

$E_{P,EOL,j,i}$ (per kg P in fertilizer) in this study only include $PO_4^{3-}$ (kg $PO_4^{3-}$/kg P input) adopts the value 0.0067 from the study by Brockmann et al.[7] After $E_{P,EOL,j,i}$ is derived for each fertilizer, $c_{P,mineral}$ can be assessed to present the average phosphorus loss after applying mineral fertilizers as shown in Equation S19.

$$c_{P,mineral} = \sum(\left(E_{P,EOL,j,PO_4{}^{3-}} \times \frac{31}{95}\right) \times MMP_{P,j}) \quad \text{(Equation S19)}$$

**Note S8. Potassium fertilizer substitution**

The life cycle emission *i* of substituted potassium fertilizers, $TE_{substituted\ K,i}$, were decided by Equation S20. *TK* is the total phosphorus in compost; $MFE_K$ is the average mineral fertilizer equivalent (kg K) in mineral fertilizers that can substitute 1 kg K in compost; $E_{K,production,j,i}$ is the emission *i* of producing and transporting fertilizer *j* that contains 1 kg K to the field; $MMP_{K,j}$ is the market mixture percentage of potassium fertilizer *j* (only KCl in this study).

$$TE_{substituted\ K,i} = (-1) \times TK \times MFE_K \times \sum(E_{K,production,j,i} \times MMP_{K,j}) \quad \text{(Equation S20)}$$

The market mixture percentage of potassium fertilizer *j*, $MMP_{K,j}$, in the U.S. was shown in Table S9.[35] $MFE_K$ (kg K) is 1.[7] $E_{K,production,j,i}$ (per kg K in fertilizer) for potassium fertilizer listed in Table S9 used the data from ecoinvent database (process names shown in Table S8).[24]

**Note S9. Lumber and Chips Substitution**

The hardwood and softwood lumber produced from urban tree waste can replace the lumber in market. The substituted lumber used the process data from ecoinvent database (process names shown in Table S8).[24]

The slabs and chips from sawing are further chipped and sold to pulp mills. The substituted hardwood chips and softwood chips used the process data from ecoinvent database (process names shown in Table S8).[24] To keep consistent with the system boundary in this study, the final end-of-life of the chips are assumed in landfill site using the same emission accounting method in Note S4.

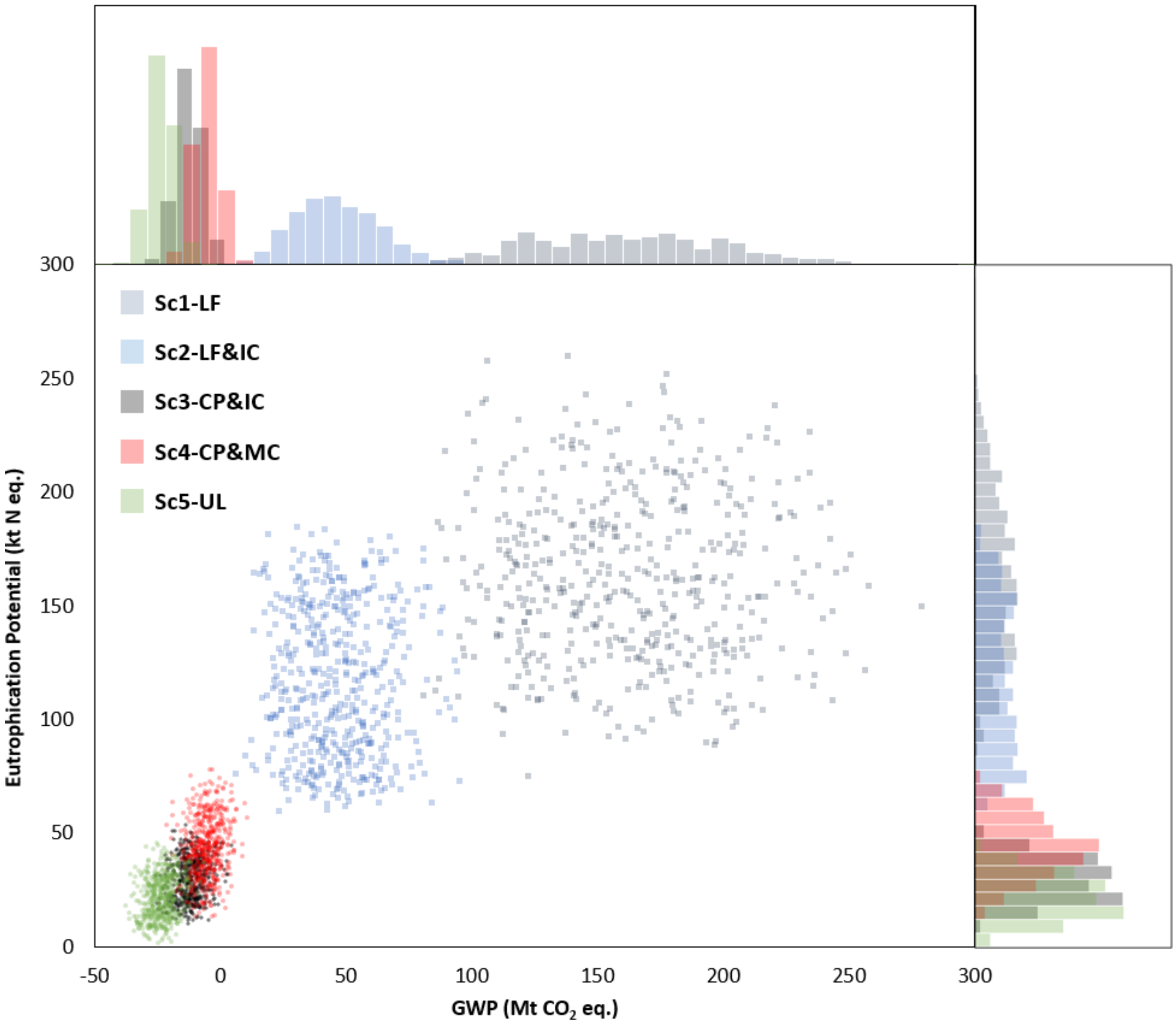


**Figure S1. Net GWP and eutrophication potential results of five scenarios based on Monte Carlo simulation.**

This figure shows the distribution of net life-cycle GWP and eutrophication potential of processing the annual U.S. urban tree waste. Scenario 1-LF: all wastes are landfilled; Scenario 2-LF&IC: leaf waste is landfilled and removed trees are incinerated for electricity generation; Scenario 3-CP&IC: leaf waste is composted and removed trees are incinerated for electricity generation; Scenario 4-CP&MC: leaf waste is composted and removed trees are produced into mulch; Scenario 5-UL: leaf waste is composted, merchantable logs are produced into lumber and chips, residues of merchantable trees and non-merchantable trees are produced into biochar.

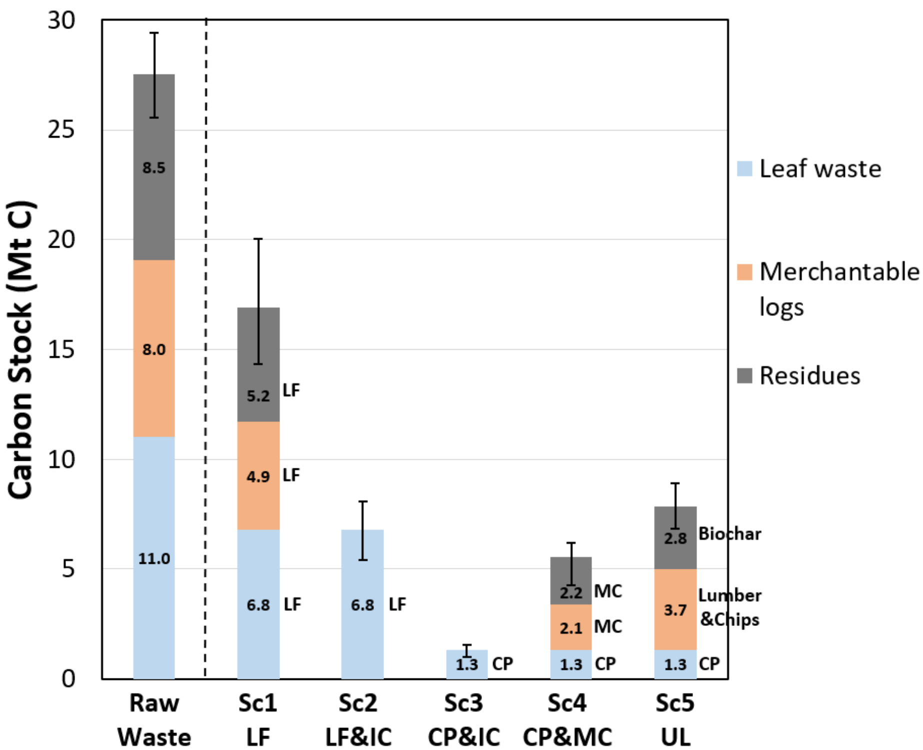


**Figure S2. Carbon stock in five scenarios over 100-year time horizon.**
This figure shows carbon stocks after 100 years of processing the urban tree waste in varied pathways, compared to the original carbon stored in urban tree waste (shown in raw waste). The letters beside bars interpret the processing pathways, including LF: landfilling, CP: composting, IC: incinerating, MC: mulching, producing lumber and chips, and producing biochar. The error bars show the 5th–95th percentile range of Monte Carlo simulation results.

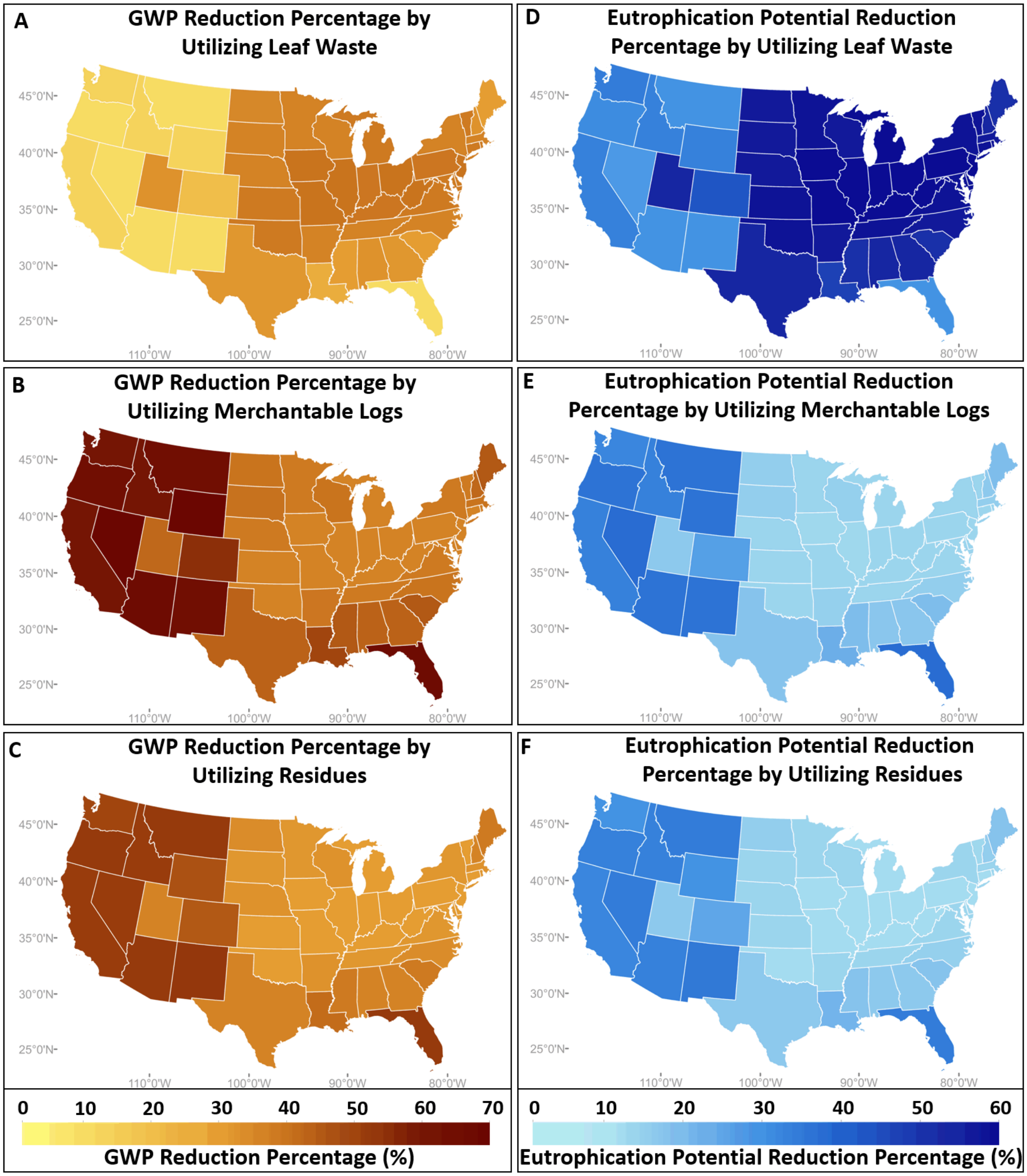


**Figure S3. State-level GWP and eutrophication potential reduction percentage of Scenario 5 compared to Scenario 1 by adopting three utilizing pathways.**

This figure shows the life-cycle GWP and eutrophication potential reduction percentage contributed by three utilization pathways in Scenario 5 compared to Scenario 1. Three utilization pathways are utilizing leaf waste for composting, utilizing merchantable logs for lumber and chips, and utilizing residues for biochar.

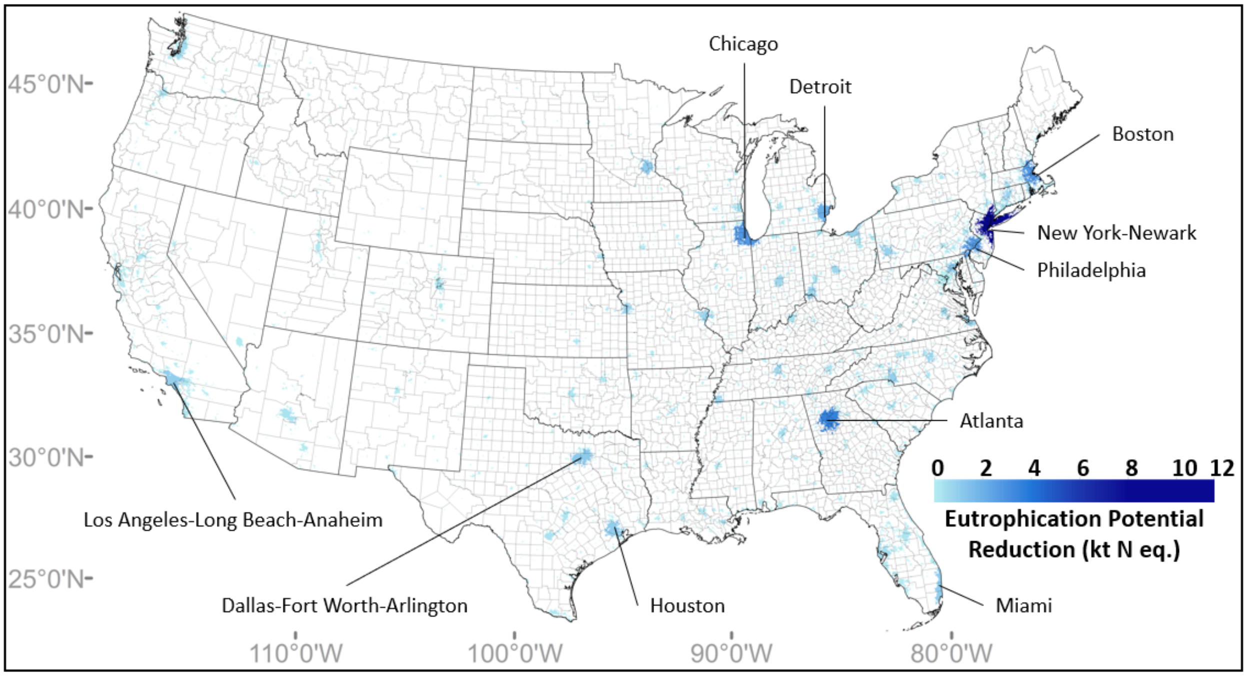


**Figure S4. Urban area eutrophication potential reduction potential per year of Scenario 5 compared to Scenario 1.**

This figure shows the life-cycle eutrophication reduction potential of Scenario 5 compared to Scenario 1 by three utilization pathways in Scenario 5 at the urban area level.

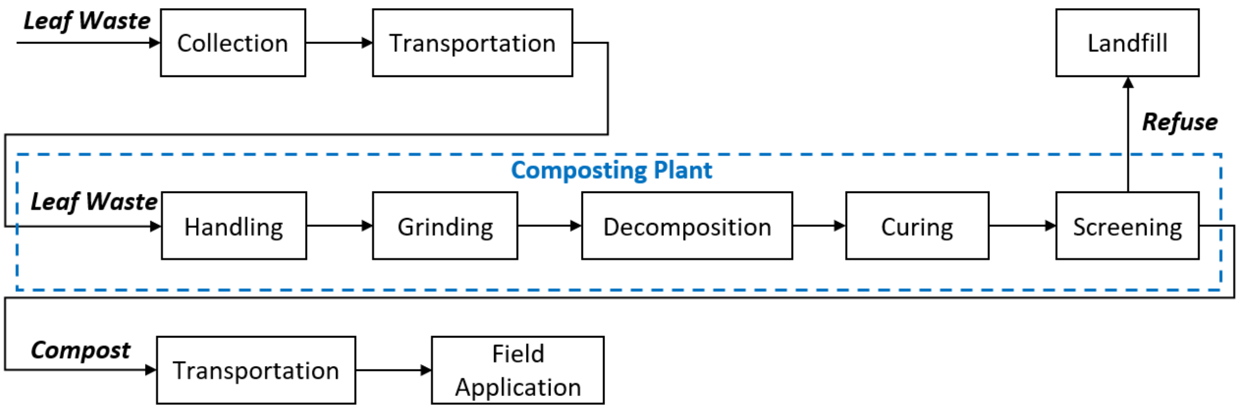


**Figure S5. Flow diagram of the composting plant.**

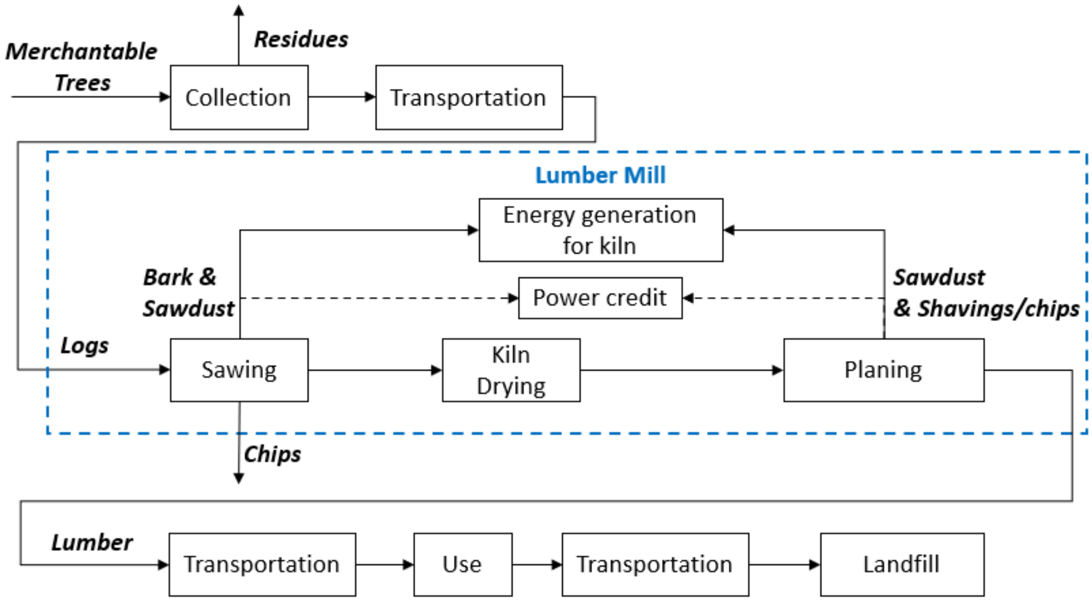


**Figure S6. Flow diagram of the lumber mill.**

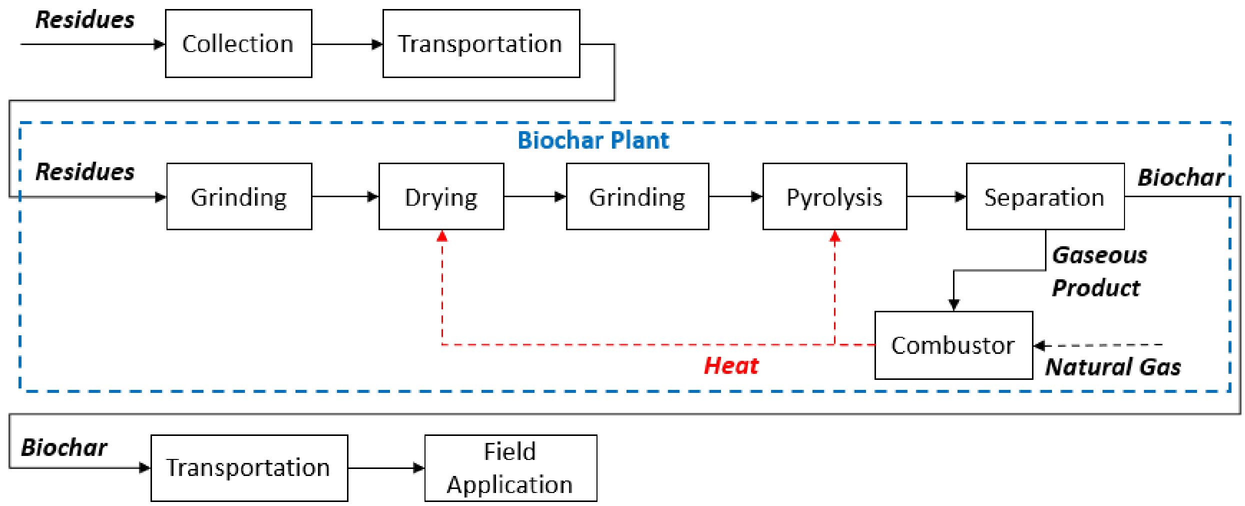


**Figure S7. Flow diagram of the biochar plant.**

**Table S1. Life-cycle results of processing 1 ODMT of three types of urban tree waste in varied pathways**

| | Leaf Waste | | Logs | | | | Residues | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| Pathway | LF | CP | LF | IC | MC | Lumber & chips | LF | IC | MC | biochar |
| | | | | | ***GWP (metric ton $CO_2$ eq.)*** | | | | | |
| Landfill | 3.8 | | 4.8 | | | | 4.8 | | | |
| Compost Production and Application | | 1.7 | | | | | | | | |
| Lumber Production and End-of-life | | | | | | 4.1 | | | | |
| Biochar Production and Application | | | | | | | | | | 1.5 |
| Collection and Incineration | | | | 2.0 | | | | 2.0 | | |
| Mulch Production and Application | | | | | 1.5 | | | | 1.5 | |
| Substituted NPK Fertilizer | | 0.0 | | | | | | | | |
| Substituted Lumber and Chips | | | | | | -3.5 | | | | |
| Substituted Charcoal | | | | | | | | | | -0.2 |
| Substituted Electricity by Incineration | | | | -0.8 | | | | -0.8 | | |
| Biogenic $CO_2$ Uptake | -1.5 | -1.5 | -1.8 | -1.8 | -1.8 | -1.8 | -1.8 | -1.8 | -1.8 | -1.8 |
| ***Net*** | 2.4 | 0.2 | 3.0 | -0.6 | -0.4 | -1.3 | 2.9 | -0.5 | -0.4 | -0.5 |
| ***Net minus P5*** | 0.9 | 0.3 | 1.2 | 0.2 | 0.1 | 0.1 | 1.1 | 0.2 | 0.1 | 0.2 |
| ***P95 minus Net*** | 1.0 | 0.3 | 1.2 | 0.2 | 0.1 | 0.1 | 1.2 | 0.2 | 0.1 | 0.2 |
| | | | | | ***Eutrophication Potential (kg N eq.)*** | | | | | |
| Landfill | 4.0 | | 1.6 | | | | 1.6 | | | |
| Compost Production and Application | | 1.4 | | | | | | | | |
| Lumber Production and End-of-life | | | | | | 0.2 | | | | |
| Biochar Production and Application | | | | | | | | | | 0.3 |
| Collection and Incineration | | | | 0.2 | | | | 0.2 | | |
| Mulch Production and Application | | | | | 0.6 | | | | 0.6 | |
| Substituted NPK Fertilizer | | -0.6 | | | | | | | | |
| Substituted Lumber and Chips | | | | | | -0.3 | | | | |
| Substituted Charcoal | | | | | | | | | | -0.3 |
| Substituted Electricity by Incineration | | | | 0.0 | | | | 0.0 | | |
| ***Net*** | 4.0 | 0.9 | 1.6 | 0.2 | 0.6 | -0.1 | 1.6 | 0.2 | 0.6 | 0.0 |
| ***Net minus P5*** | 1.7 | 0.5 | 1.3 | 0.0 | 0.5 | 0.1 | 1.3 | 0.0 | 0.5 | 0.1 |
| ***P95 minus Net*** | 1.9 | 0.5 | 1.3 | 0.0 | 0.4 | 0.1 | 1.3 | 0.0 | 0.4 | 0.1 |

**Table S2. State-level annual urban tree waste availability in the U.S. (conterminous 48 states) (in thousand oven dry metric ton)**

| | Leaf waste | | Merchantable trees | | Non-merchantable trees | |
|---|---|---|---|---|---|---|
| State | Average | Standard deviation | | Standard deviation | Average | Standard deviation |
| Alabama | 599.9 | 121.8 | 455.7 | 80.1 | 284.3 | 49.9 |
| Arizona | 82.6 | 16.8 | 277.6 | 49.3 | 172.4 | 30.7 |
| Arkansas | 402.0 | 81.6 | 246.9 | 43.2 | 153.1 | 26.8 |
| California | 455.4 | 92.5 | 1262.8 | 223.3 | 787.2 | 139.2 |
| Colorado | 82.1 | 16.7 | 130.7 | 23.1 | 81.8 | 14.4 |
| Connecticut | 1013.6 | 205.7 | 547.7 | 96.9 | 342.3 | 60.6 |
| Delaware | 126.5 | 25.7 | 69.6 | 12.4 | 42.9 | 7.6 |
| Florida | 412.7 | 83.8 | 1495.7 | 263.4 | 931.8 | 164.1 |
| Georgia | 1806.1 | 366.6 | 1367.9 | 241.8 | 852.1 | 150.7 |
| Idaho | 11.6 | 2.4 | 31.8 | 6.1 | 20.7 | 3.9 |
| Illinois | 1185.9 | 240.7 | 603.1 | 106.2 | 376.9 | 66.3 |
| Indiana | 717.1 | 145.6 | 385.8 | 67.6 | 241.7 | 42.4 |
| Iowa | 246.1 | 50.0 | 126.1 | 21.5 | 78.9 | 13.5 |
| Kansas | 295.9 | 60.1 | 164.8 | 29.3 | 102.7 | 18.2 |
| Kentucky | 457.7 | 92.9 | 244.9 | 43.1 | 152.6 | 26.9 |
| Louisiana | 371.9 | 75.5 | 427.8 | 75.4 | 267.2 | 47.1 |
| Maine | 110.2 | 22.4 | 102.3 | 18.6 | 62.7 | 11.4 |
| Maryland | 908.2 | 184.3 | 509.9 | 89.4 | 317.6 | 55.6 |
| Massachusetts | 1325.1 | 269.0 | 825.0 | 144.7 | 515.0 | 90.3 |
| Michigan | 1382.6 | 280.7 | 747.2 | 132.5 | 465.3 | 82.5 |
| Minnesota | 672.0 | 136.4 | 388.4 | 67.8 | 241.6 | 42.2 |
| Mississippi | 273.5 | 55.6 | 246.0 | 43.1 | 154.0 | 26.9 |
| Missouri | 749.4 | 152.1 | 391.2 | 69.3 | 243.8 | 43.2 |
| Montana | 8.9 | 1.8 | 31.0 | 6.2 | 19.0 | 3.8 |
| Nebraska | 86.4 | 17.6 | 52.3 | 9.2 | 32.7 | 5.8 |
| Nevada | 12.4 | 2.5 | 48.3 | 9.3 | 29.2 | 5.7 |
| New Hampshire | 238.5 | 48.4 | 176.4 | 30.7 | 111.1 | 19.3 |
| New Jersey | 1227.7 | 249.2 | 657.1 | 115.4 | 410.4 | 72.1 |
| New Mexico | 18.7 | 3.8 | 67.6 | 12.3 | 42.4 | 7.7 |
| New York | 1632.6 | 331.5 | 973.6 | 171.0 | 606.4 | 106.5 |
| North Carolina | 1777.8 | 360.9 | 1176.1 | 207.8 | 733.9 | 129.7 |
| North Dakota | 13.7 | 2.8 | 9.1 | 1.5 | 5.9 | 1.0 |
| Ohio | 1559.3 | 316.6 | 809.1 | 143.3 | 503.4 | 89.2 |
| Oklahoma | 289.4 | 58.8 | 154.8 | 27.6 | 97.7 | 17.4 |
| Oregon | 51.0 | 10.4 | 165.9 | 29.2 | 104.1 | 18.3 |
| Pennsylvania | 1766.6 | 358.7 | 935.8 | 164.7 | 584.2 | 102.8 |
| Rhode Island | 173.5 | 35.3 | 98.6 | 16.9 | 61.4 | 10.6 |
| South Carolina | 654.4 | 132.8 | 576.4 | 101.7 | 358.6 | 63.3 |
| South Dakota | 31.6 | 6.4 | 20.5 | 3.1 | 12.0 | 1.9 |
| Tennessee | 977.5 | 198.4 | 581.5 | 103.1 | 363.5 | 64.4 |
| Texas | 1384.9 | 281.1 | 1137.2 | 200.1 | 710.3 | 124.9 |
| Utah | 89.8 | 18.2 | 66.2 | 12.3 | 41.3 | 7.7 |
| Vermont | 65.2 | 13.3 | 37.5 | 6.3 | 22.5 | 3.8 |
| Virginia | 1035.2 | 210.2 | 587.5 | 103.0 | 367.5 | 64.5 |

| | | | | | | |
|---|---|---|---|---|---|---|
| Washington | 163.5 | 33.2 | 406.8 | 72.4 | 253.2 | 45.1 |
| West Virginia | 294.9 | 59.9 | 155.1 | 27.6 | 97.4 | 17.4 |
| Wisconsin | 408.2 | 82.9 | 254.0 | 44.6 | 158.5 | 27.9 |
| Wyoming | 3.9 | 0.8 | 11.4 | 1.6 | 6.1 | 0.9 |

**Table S3. Statistical characteristics of key parameters related to compost production**

| | Unit | Mean value | Minimum | Maximum | Assumed distribution |
|---|---|---|---|---|---|
| Leaf litter carbon content[39,40] | % | 40.9 | 29.8 | 52.0 | Uniform U[29.8,52.0] |
| Leaf litter nitrogen content[39–41] | % | 1.08 | 0.54 | 1.62 | Uniform U[0.54,1.62] |
| Leaf litter phosphorus content[39] | % | 0.13 | 0 | 0.26 | Uniform U[0,0.26] |
| Leaf litter potassium content[39] | % | 0.43 | 0.07 | 0.78 | Uniform U[0.07,0.78] |
| Leaf litter ash content[41–43] | % | 16.8 | 12.0 | 21.6 | Uniform U[12.0,21.6] |
| Refuse rate[44,45] | % | 4.5 | 0 | 9.0 | Uniform U[0,9.0] |
| Total diesel consumption in the composting plant[44–47] | kg/wet metric ton feedstock | 4.6 | 1.7 | 7.5 | Uniform U[1.7,7.5] |
| Total electricity consumption in the composting plant[44–47] | kWh/wet metric ton feedstock | 61.1 | 27.3 | 95.0 | Uniform U[27.3,95.0] |
| Carbon mass emitted as $CO_2$ in composting[46,48–50] | % | 56.0 | 51.9 | 60.0 | Uniform U[51.9,60.0] |
| Carbon mass emitted as $CH_4$ in composting[48–50] | % | 0.45 | 0 | 0.90 | Uniform U[0,0.90] |
| Nitrogen mass emitted as $N_2O$ in composting[46,48,49] | % | 3.38 | 0 | 6.75 | Uniform U[0,6.75] |
| Nitrogen mass emitted as $NH_3$ in composting[46,48,50] | % | 11.6 | 5.1 | 18.0 | Uniform U[5.1,18.0] |
| Nitrogen mass emitted as $NO_x$ in composting[48,50] | % | 9.4 | 0.4 | 18.4 | Uniform U[0.4,18.4] |
| Compost moisture content[48,50] | % (*wet basis*) | 22.4 | 10 | 34.8 | Uniform U[22.4,34.8] |
| Transportation distance from collecting area to the composting plant[47,51] | km | 27.1 | 13.2 | 41.1 | Uniform U[13.2,41.1] |
| Transportation distance from the composting plant to field[24] | km | 423 | 338.4 | 549.9 | Uniform U[338.4,549.9] |
| Diesel consumption in collecting tree leaf litter[24,52] | kWh/wet metric ton feedstock | 4.0 | 0 | 8.0 | Uniform U[0,8.0] |
| Leaf litter moisture content[40] | % (*wet basis*) | 53.8 | N/A | N/A | N/A |

**Table S4. Statistical characteristics of key parameters related to lumber production**

| | Unit | Mean value | Minimum | Maximum | Assumed distribution |
|---|---|---|---|---|---|
| Hardwood log green density[53] | kg/m$^3$ | 801 | 577 | 1025 | Uniform U[577,1025] |
| Softwood log green density[53] | kg/m$^3$ | 733 | 545 | 921 | Uniform U[545,921] |
| Hardwood log moisture content[53] | % (*dry basis*) | 76.5 | 46.0 | 107.0 | Uniform U[46.0,107.0] |
| Softwood log moisture content[53] | % (*dry basis*) | 70.5 | 35.0 | 106.0 | Uniform U[35.0,106.0] |
| Hardwood bark mass fraction[54] | % | 10.0 | 8.0 | 12.0 | Uniform U[8.0,12.0] |
| Softwood bark mass fraction[54] | % | 10.5 | 9.0 | 13.0 | Uniform U[9.0,13.0] |
| Hardwood log carbon content[55] | % *daf* | 51.0 | 46.5 | 56.7 | Normal $N(51.0, 2.9^2)$ |
| Softwood log carbon content[55] | % *daf* | 51.0 | 46.2 | 61.0 | Normal $N(51.0, 2.9^2)$ |
| Hardwood log nitrogen content[55] | % *daf* | 0.4 | 0 | 0.8 | Uniform U[0,0.8] |
| Softwood log nitrogen content[55] | % *daf* | 0.4 | 0 | 0.8 | Uniform U[0,0.8] |
| Diesel consumption of hauling materials[53,56,57] | kg/m$^3$ dried lumber | 3.6 | 1.7 | 5.5 | Uniform U[1.7,5.5] |
| Gasoline consumption of hauling materials[53,56,57] | kg m$^{-3}$ dried lumber | 0.23 | 0.03 | 0.23 | Uniform U[0.03,0.43] |
| Lumber yield rate in sawing[15,58,67–70,59–66] | % | 50.0 | 36.0 | 64.0 | Triangular 36, 50, 64 |
| Electricity consumption of sawing[15,68,71,72] | kWh/m$^3$ log input | 24.4 | 16.5 | 32.3 | Uniform U[16.5,32.3] |
| Electricity consumption of kiln drying and kiln heat generation[15,66,68,72,73] | kWh/m$^3$ lumber input | 26.9 | 17.9 | 35.8 | Uniform U[17.9,35.8] |
| Lumber target moisture content[62,66,69,70,74–77] | % (*dry basis*) | 12.5 | 6.0 | 19.0 | Triangular 6.0, 12.5, 19.0 |
| Overall energy efficiency for energy generation and drying[15,68,73] | % | 23.3 | 16.7 | 29.8 | Uniform [16.7,29.8] |
| Lumber drying shrinkage[69,70,78] | % | 9.1 | 4.4 | 16.0 | Triangular 4.4, 9.1, 16.0 |
| Electricity consumption of planing[15,66,68,72] | kWh m$^3$ lumber input | 18.2 | 7.7 | 28.7 | Uniform U[7.7,28.7] |
| Planing byproduct mass percentage[15,66,68,79] | % | 17.8 | 13.8 | 21.8 | Uniform U[13.8,21.8] |
| Average transportation distance of lumber distribution[76] | km | 212 | 104 | 320 | Uniform U[104,320] |
| Average hauling distance to landfill site[80–83] | km | 256 | 32 | 480 | Uniform U[32,480] |
| Average transportation distance from field to lumber mills[15] | km | 91 | N/A | N/A | N/A |

**Table S5. Statistical characteristics of key parameters related to biochar production**

| | Unit | Mean value | Minimum | Maximum | Assumed distribution |
|---|---|---|---|---|---|
| Hardwood carbon content[53] | % *daf* | 51.0 | 46.5 | 56.7 | Normal $N(51.0, 2.9^2)$ |
| Softwood carbon content[53] | % *daf* | 51.0 | 46.2 | 61.0 | Normal $N(51.0, 2.9^2)$ |
| Hardwood C/H ratio[53] | | 8.3 | 7.2 | 10.0 | Normal $N(8.3, 0.65^2)$ |
| Softwood C/H ratio[53] | | 8.2 | 7.1 | 9.8 | Normal $N(8.2, 0.66^2)$ |
| Hardwood C/O ratio[53] | | 1.2 | 1.0 | 1.6 | Normal $N(1.2, 0.13^2)$ |
| Softwood C/O ratio[53] | | 1.2 | 1.0 | 2.0 | Normal $N(1.2, 0.14^2)$ |
| Hardwood ash content[53] | % *daf* | 2.6 | 0.1 | 10.6 | Gamma a=2.5, b=0.75 |
| Softwood ash content[53] | % *daf* | 1.3 | 0.1 | 6.3 | Gamma a=0.5, b=2.5 |
| Hardwood residue moisture content[84] | % *dry basis* | 89.2 | 53.8 | 124.7 | Uniform U[53.8,124.7] |
| Softwood residue moisture content[84] | % *dry basis* | 89.2 | 53.8 | 124.7 | Uniform U[53.8,124.7] |
| Biochar decay rate[85–89] | | 0.00095 | 0.0005 | 0.00249 | Uniform U[0.0005,0.00249] |
| Average transportation distance from collection area to the biochar plant[90] | | 69.5 | 52 | 87 | Uniform U[52,87] |
| Average transportation distance from the biochar plant to application field[24] | | 100 | 70 | 130 | Uniform U[70,130] |

**Table S6. Process parameters for biochar production**

| | Unit | Value |
|---|---|---|
| Pyrolysis time[17] | minutes | 60 |
| Pyrolysis temperature[17] | °C | 500 |
| Pyrolysis pressure[17] | atm | 1 |
| Pyrolysis nitrogen flow[17] | % of inlet feedstock flow | 16.7 |
| Pyrolysis thermal efficiency[91] | % | 90 |
| Combustor excess air portion[92] | % | 30 |
| Diesel consumption in the wheel loader[90] | kg/ODMT feedstock | 1.4 |
| Electricity consumption in the grinder[90] | kWh/ODMT fed in | 40 |
| Electricity consumption in the hammer mill[90] | kWh/ODMT fed in | 33 |
| Electricity consumption in the rotary drum dryer[90] | kWh/ODMT fed in | 45 |
| Electricity consumption in the feed hopper[90] | kWh/ODMT fed in | 1.7 |

**Table S7. Statistical characteristics of key parameters related to landfilling and incineration of urban tree waste**

| | Mean value | Minimum | Maximum | Assumed distribution |
|---|---|---|---|---|
| $DOC_f$[25–28] | 0.55 | 0.50 | 0.60 | Uniform U[0.50,0.60] |
| $MCF$[25–28] | 0.90 | 0.80 | 1.00 | Uniform U[0.80,1.00] |
| $F$[25–28] | 0.50 | 0.40 | 0.60 | Uniform U[0.40,0.60] |
| $OX$[25–28] | 0.05 | 0 | 0.10 | Uniform U[0,0.10] |
| $k$[25–28] | 0.13 | 0.05 | 0.20 | Uniform U[0.05,0.20] |
| Volume rate of $CH_4$ to $CO_2$ in landfill gas emissions[30] | 1.60 | 1.40 | 1.80 | Uniform U[1.4,1.8] |
| Percentage of total N converting to $NH_3$ that is emitted in landfill gas emissions[31,33,34] | 5.15 | 4.10 | 6.20 | Uniform U[4.10,6.20] |
| Percentage of total N converting to $NH_4^+$ that is emitted to water[24] | 34.2 | 30.7 | 37.8 | Uniform U[30.7,37.8] |
| Percentage of total N converting to $NO_3^-$ that is emitted to water[24,31,33,34] | 1.2 | 0 | 2.4 | Uniform U[0,2.4] |
| Average hauling distance to landfill site (km)[80–83] | 256 | 32 | 480 | Uniform U[32,480] |
| Power generation efficiency by wood incineration[93–98] | 33.5 | 25.0 | 42.0 | Uniform U[25.0,42.0] |

**Table S8. Ecoinvent processes of substituted products used in this study**

| Product | Process[24] |
|---|---|
| Ammonia | market for ammonia, liquid \| Cutoff, U_RoW |
| Ammonium Nitrate | market for ammonium nitrate, as N \| Cutoff, U_GLO |
| Ammonium Sulfate | market for ammonium sulfate, as N \| Cutoff, U_GLO |
| Sodium Nitrate | market for sodium nitrate, unrefined \| Cutoff, U_GLO |
| Urea | market for urea, as N \| Cutoff, U_GLO |
| Superphosphates | single superphosphate production \| phosphate fertiliser, as P2O5 \| Cutoff, U_RoW |
| Diammonium phosphate | diammonium phosphate production \| phosphate fertiliser, as P2O5 \| Cutoff, U_RoW |
| Monoammonium phosphate | monoammonium phosphate production \| phosphate fertiliser, as P2O5 \| Cutoff, U_RoW |
| Potassium chloride | potassium chloride production \| potassium chloride, as K2O \| Cutoff, U_RoW |
| Hardwood chips | hardwood forestry, birch, sustainable forest management \| wood chips, wet, measured as dry mass \| Cutoff, U_RoW |
| Softwood chips | softwood forestry, pine, sustainable forest management \| wood chips, wet, measured as dry mass \| Cutoff, U_RoW |
| Hardwood lumber | market for sawnwood, beam, hardwood, dried (u=10%), planed \| sawnwood, beam, hardwood, dried (u=10%), planed \| Cutoff, U_GLO |
| Softwood lumber | market for sawnwood, beam, softwood, dried (u=10%), planed \| sawnwood, beam, softwood, dried (u=10%), planed \| Cutoff, U_GLO |
| Charcoal | market for charcoal \| charcoal \| Cutoff, U_GLO |

**Table S9. Market mixture percentage (2015) of fertilizers in terms of nutrient contribution**

| Fertilizer | Consumption (short ton) | Market Mixture Percentage[a] |
|---|---|---|
| | *Nitrogen fertilizers* | |
| Ammonia | 4,141,218 | 28.7% |
| Ammonium Nitrate | 608,268 | 1.8% |
| Ammonium Sulfate | 1,935,361 | 3.4% |
| Nitrogen Solution[b] | 11,896,660 | 38.3% |
| Sodium Nitrate | 13,694 | 0.1% |
| Urea | 7,038,055 | 27.7% |
| | *Phosphorus fertilizers* | |
| Superphosphates | 717,929 | 12.7% |
| Diammonium phosphate | 2,466,223 | 38.6% |
| Monoammonium phosphate | 2,710,398 | 48.7% |
| | *Potassium fertilizer* | |
| Potassium chloride | 5,854,194 | 100% |

[a] Market mixture percentage is calculated based on nutrient contribution percentage that is, for 1 kg N, or P, or K fertilizer purchased in the market, the portion contributed by certain type of fertilizer

[b] Nitrogen solution was assumed to be 33% urea and 67% ammonium nitrate based on the study by Mengel.[99]

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