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Comparing the carbon footprints of urban and conventional agriculture

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Urban agriculture (UA) is a widely proposed strategy to make cities and urban food systems more sustainable. Until now, we have lacked a comprehensive assessment of the environmental performance of UA relative to conventional agriculture, and results from earlier studies have been mixed. This is the first large-scale study to resolve this uncertainty across cities and types of UA, employing citizen science at 73 UA sites in Europe and the United States to compare UA products to food from conventional farms. Results reveal that the carbon footprint of food from UA is six times greater than conventional agriculture (420 gCO_2e versus 70 gCO_2e per serving). However, some UA crops (for example, tomatoes) and sites (for example, 25% of individually managed gardens) outperform conventional agriculture. These exceptions suggest that UA practitioners can reduce their climate impacts by cultivating crops that are typically greenhouse-grown or air-freighted, maintaining UA sites for many years, and leveraging circularity (waste as inputs).

Urban agriculture (UA) (that is, growing food in and around cities) is intended to make cities more sustainable, healthy and just. Despite strong evidence of social and nutritional benefits from UA, environmental claims are not well supported, particularly how the environmental footprint of UA compares to the conventional agriculture it could supplant[1]. As interest in UA increases[2], policymakers, citizens and scientists must ensure that UA is beneficial for people and the planet.

How UA compares with conventional agriculture depends on the crops grown, growing systems and local climate[3]. It is unclear what forms of UA are environmentally friendly, because case studies of individual cities typically only assess one form of UA[4–6]. Environmental footprints of UA remain scarce, and most that have been published so far have prioritized high-tech, energy-intensive forms of UA[1] (for example, vertical farms and rooftop greenhouses) in lieu of open-air, soil-based forms (referred to here as 'low-tech UA'), which comprise the bulk of food-growing spaces in cities[7,8]. A recent systematic review found that only a third of environmental assessments have assessed low-tech UA[1].

Furthermore, although existing research suggests that low-tech UA may produce total greenhouse-gas emissions (GHGs) per serving of vegetables similar to conventional agriculture[1,3], these findings are undermined by numerous shortcomings. Sample sizes are often small[1]. Studies with large sample sizes only consider the amounts and types of resource used and not environmental impacts (for example, GHGs)[9–11]. When impacts are considered, studies report them per

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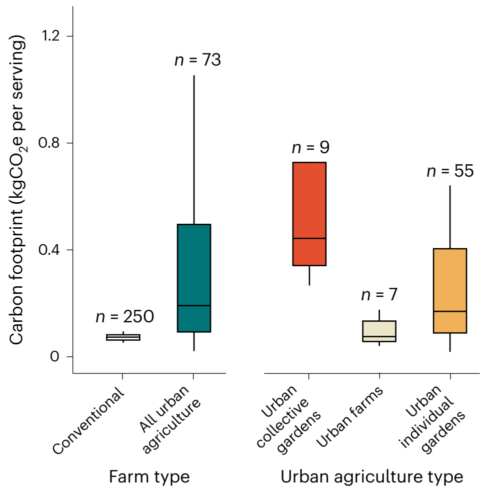

Fig. 1 | The carbon footprint of conventional versus urban agriculture. Results are shown per serving of produce as defined by the United States Department of Agriculture. Boxplots reflect the median (center bar) and interquartile range (IQR, box minima and maxima) of GHG impact, and UA sites above 1.0 $kgCO_2e$ per serving are removed to improve legibility (whiskers reflect standard approximations of range based on 1.5 × IQR; full results are provided in Supplementary Fig. 1). Two UA sites could not be classified as collective, individual or farm, so only 71 sites are included in the right panel.

kilogram of total harvest and not per crop or food group[1]. Lastly, low data representativeness is common. For example, some studies incorrectly assume that the only difference between UA and conventional agriculture is transport distance[12,13]. Taken as a whole, there remain serious knowledge gaps with respect to the environmental performance of low-tech UA.

This Article addresses these gaps through carbon footprint analysis of low-tech UA, covering 73 sites in France, Germany, Poland, the United Kingdom and the United States, using data collected through citizen science[14,15]. We assessed the carbon footprint across the lifecycle of producing food at three types of low-tech UA: urban farms (professionally managed, focused on food production), individual gardens (small plots managed by single gardeners) and collective gardens (communal spaces managed by groups of gardeners). We estimated embodied GHGs and synthetic nutrient footprints of food from UA and compared these to conventional agricultural products sold in each of our five countries.

By assessing actual inputs and outputs on UA sites, we were able to assign climate change impacts to each serving of produce (that is, recommended grams of a crop a person should consume daily to align with dietary guidelines). This revealed that UA has higher GHGs per serving of fruit or vegetable than conventional agriculture, irrespective of country. To promote UA that is more broadly sustainable—climate-friendly, resource efficient and socially beneficial—we analyze key trends across our sample of UA sites and argue that policymakers and UA practitioners should maximize the lifespan of farm infrastructure, promote urban waste streams as inputs, and use farms as sites for education, leisure and community building.

Results and discussion

Low-tech UA carbon footprint six times that of conventional agriculture

Food produced at our UA study sites is more carbon-intensive than food produced on conventional farms (Fig. 1). To reach this conclusion, we compared food produced on UA sites to conventional crops, produced both domestically and abroad, considering on-farm impacts, processing and transportation to the city (see Methods for details). On average, UA emits 0.42 kilograms carbon dioxide equivalents ($kgCO_2e$, standard error (s.e.) = 0.07 $kgCO_2e$) per serving (equivalent to a mean (μ) of 3.12 $kgCO_2e$ per kg produce, s.e. = 0.53 $kgCO_2e\ kg^{-1}$), six times higher than the 0.07 $kgCO_2e$ per serving (s.e. = 0.005 $kgCO_2e$ per serving; μ = 0.47 $kgCO_2e\ kg^{-1}$, s.e. = 0.032 $kgCO_2e\ kg^{-1}$) of conventional produce ($P \ll 0.001$).

On average, all forms of UA studied here are more carbon-intensive than conventional agriculture, although this difference is only statistically significant for collective gardens ($P = 0.02$) and individual gardens ($P < 0.001$). Collective gardens are the most carbon-intensive form of UA (μ = 0.81 $kgCO_2e$ per serving, 7.50 $kgCO_2e\ kg^{-1}$). Individual gardens and urban farms are similar on average (both produce 0.34 $kgCO_2e$ per serving), but variation among urban farms leaves them statistically indistinguishable from conventional farms ($P = 0.33$). In fact, most urban farms are carbon-competitive with conventional farms (median = 0.08 $kgCO_2e$ per serving when one particularly carbon-intensive urban farm is excluded from the analysis). These findings mirror literature trends, which identify non-commercial UA as more carbon-intensive than commercial UA, except when the latter uses energy-intensive indoor farming[16].

The carbon intensity of UA differs by country due to variations in the forms of UA practiced. For example, UA carbon impacts are lowest in Poland ($N = 35$), where our sample of gardens was dominated by individual gardens, and highest in the UK ($N = 6$), where case studies are mostly collective gardens. Nonetheless, the average vegetable at the local grocer outperforms the average vegetable on UA sites in all five countries (Supplementary Fig. 2).

Conventional agriculture and UA have similar GHG impacts for some crops

We allocated food impacts between crops using nutritional content, calorie content, economic value and mass (Methods). Method of allocation did not affect the directionality of results (Supplementary Table 1), and results presented in-text are averaged across allocation schemes. The carbon intensity per serving of fruit ($N = 73$) is higher in low-tech UA (μ = 0.47 $kgCO_2e$, 4.07 $kgCO_2e\ kg^{-1}$) than conventional agriculture (μ = 0.07 $kgCO_2e$, 0.49 $kgCO_2e\ kg^{-1}$). The same is true of vegetables ($N = 73$; μ = 0.46 versus 0.08 $kgCO_2e$ per serving, 3.48 versus 0.52 $kgCO_2e\ kg^{-1}$). Similarly, the most popular crops consumed in our five countries are more carbon-intensive when grown using low-tech UA (Fig. 2).

However, select crops are carbon-competitive with conventional agriculture. Competitiveness depends on growing practices, both in urban and conventional settings. For example, the median urban tomato (0.17 $kgCO_2e$ per serving) outperforms conventional tomatoes (μ = 0.27 $kgCO_2e$ per serving). Although, on average, urban tomatoes are more carbon-intensive than conventional tomatoes ($P = 0.02$), this low median demonstrates that UA sites often outperform conventional tomato growing. This is largely due to the carbon-intensive greenhouses that supply most tomatoes to our case cities, as well as sub-optimal distribution patterns of the crop from farm to city[17–19]. Similarly, when we test the sensitivity of our findings to air-freight importation (common with a small subset of highly perishable vegetables such as asparagus[20]), we find that the statistical difference between individual gardens and conventional agriculture vanishes (Supplementary Table 2).

This suggests that urban food growers could maximize carbon benefits (or minimize carbon impacts) by selecting crops conventionally grown or distributed using carbon-intensive methods. Research shows that growers' motivations for crop selection vary substantially, from balancing diets to cultural preferences[21]. In our sample, environmental sustainability was the most common motivation for growing food. Research elaborating on the types of vegetable that offer carbon benefits accompanied by education on these climate-friendly crop choices could help urban food producers better achieve these goals.

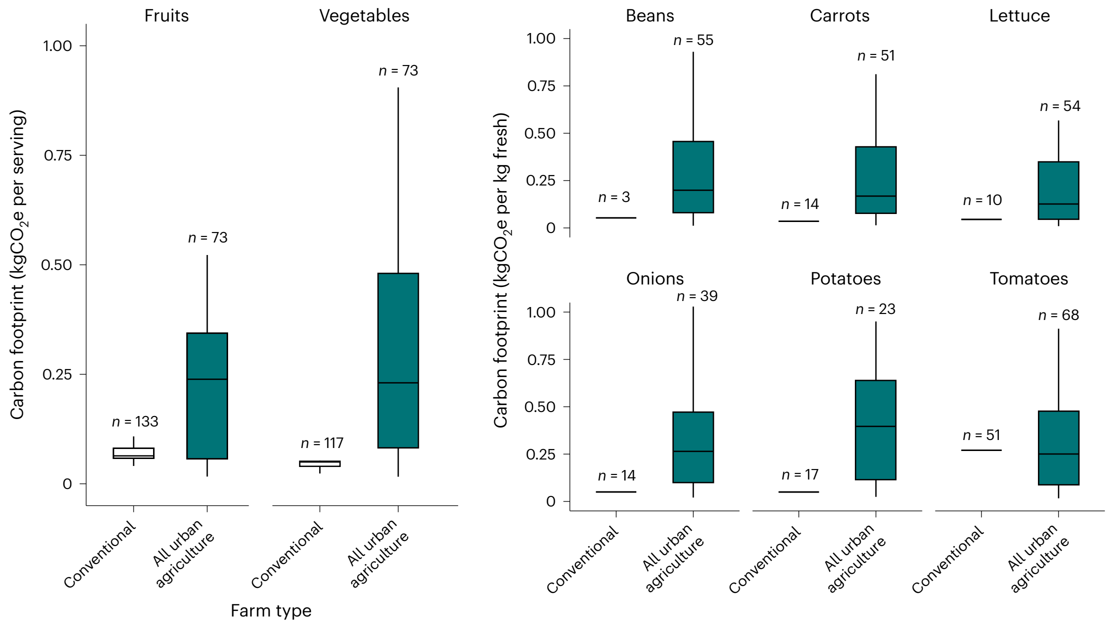

Fig. 2 | GHG emissions by farm type and product. Impacts in the left panel are shown per serving of food. Impacts in the right panel are shown per kilogram of crop. Boxplots employ the same descriptive statistics as Fig. 1 (median and IQR of GHG impact), and UA sites above 1.0 kgCO_2e per serving are removed to improve legibility.

Towards climate-friendly UA

UA is expected to continue proliferating globally[2,14]. Our findings suggest that steps must be taken to ensure that UA supports, and does not undermine, urban decarbonization efforts.

We can glean insights into climate-friendly UA from the share of our sites that grow low-carbon food. Although, in the aggregate, UA is more carbon-intensive than conventional agriculture, 17 of our 73 study sites outperform conventional agriculture (referred to hereafter as 'climate-friendly'—see Methods for a sensitivity analysis). Urban farms are most likely to be climate-friendly (43% of urban farm sites), followed by individual gardens (25% of sites).

Interestingly, neither environmental actions (for example, presence of solar panels) nor expressed values are predictive of carbon emissions (Supplementary Table 3). What, then, makes some sites more climate-friendly? We identify three best practices crucial to making low-tech UA carbon-competitive with conventional agriculture: (1) extend infrastructure lifetimes, (2) use urban waste as inputs and (3) generate high levels of social benefits.

UA sites should preserve infrastructure as long as possible. Infrastructure is the largest driver of carbon emissions at low-tech UA sites (63% of impacts), although this drops to roughly one-third for urban farms (Fig. 3a). This includes raised beds, compost infrastructure and structures (for example, sheds; Supplementary Table 4). UA must operate for sustained periods to amortize emissions invested in infrastructure (Fig. 3b). For example, a raised bed built and used for five years will have approximately four times the environmental impact per serving as a raised bed used for 20 years. Yet, gardens and farms are precarious, especially in cities with development pressure, and some projects are designed for temporary use, with infrastructure demolished in years, not decades[22–25]. Only urban farms overcome this challenge precisely because infrastructure plays a diminished role in their carbon footprint.

This finding points to an important synergy between environmental and social sustainability in UA. Activists and scholars have long pointed to insecure land tenure as a threat to UA[26,27]. This is most acute in cities experiencing economic growth. For example, New York City (NYC) in the 1990s saw land developers ally with city officials to displace community gardens[26]. Problematically, UA may fuel green gentrification in its vicinity, making farm sites vulnerable to development[28–30]. To avoid displacing farms and the associated demolition of infrastructure, policies are needed that promote stable land tenure for UA sites. For example, the establishment of community land trusts, such as NYC's Bronx Land Trust[26], can help remove land from the real-estate market[31].

UA sites should leverage urban waste streams as inputs. Urban symbiosis refers to processes by which urban systems reuse their own waste. According to our findings, UA is most climate-friendly when it serves as a hub for symbiosis of building materials, organic waste and rainwater. This is consistent with recent work highlighting the potential for enhanced circularity and innovative technology to reduce UA carbon footprints[32,33].

Climate-friendly sites in our sample cut their emissions by more than 52% by upcycling refuse from the urban environment for raised beds, structures and other infrastructure—twice as much savings as high-carbon sites. If our UA sites sourced all their materials from urban waste, all three forms of UA would be carbon-competitive with conventional agriculture (that is, there is no statistically significant difference). However, much of the reuse of building materials at our sites is opportunistic, and overall recycling rates of construction and demolition waste are abysmal (excepting crushed aggregates for road fill)[34]. Cities can work with the building sector to make these resources more widely available, giving second life to materials that are unusable for construction but potentially useful in UA. This would boost material-reuse rates and contribute to climate-friendly UA.

Perhaps the most well-known symbiotic relationship between UA and cities is composting[35]. The farms and gardens in our study applied 12 kg of compost per square meter annually, equivalent to ~30 kg of biomass (for example, food waste and yard trimmings) absorbed per square meter[36–39]. This reduces reliance on synthetic fertilizers. Sites in our sample used 95% less synthetic nutrients (0.06 g nitrogen per serving, 0.04 g phosphorus per serving, 0.05 g potassium per serving) than conventional farms (0.88 g nitrogen per serving, 1.4 g phosphorus per serving, 0.99 g potassium per serving). As noted by others, different UA types apply fertilizer at different rates[9,35]. None of the collective gardens in our sample applied synthetic fertilizers. Conversely, urban

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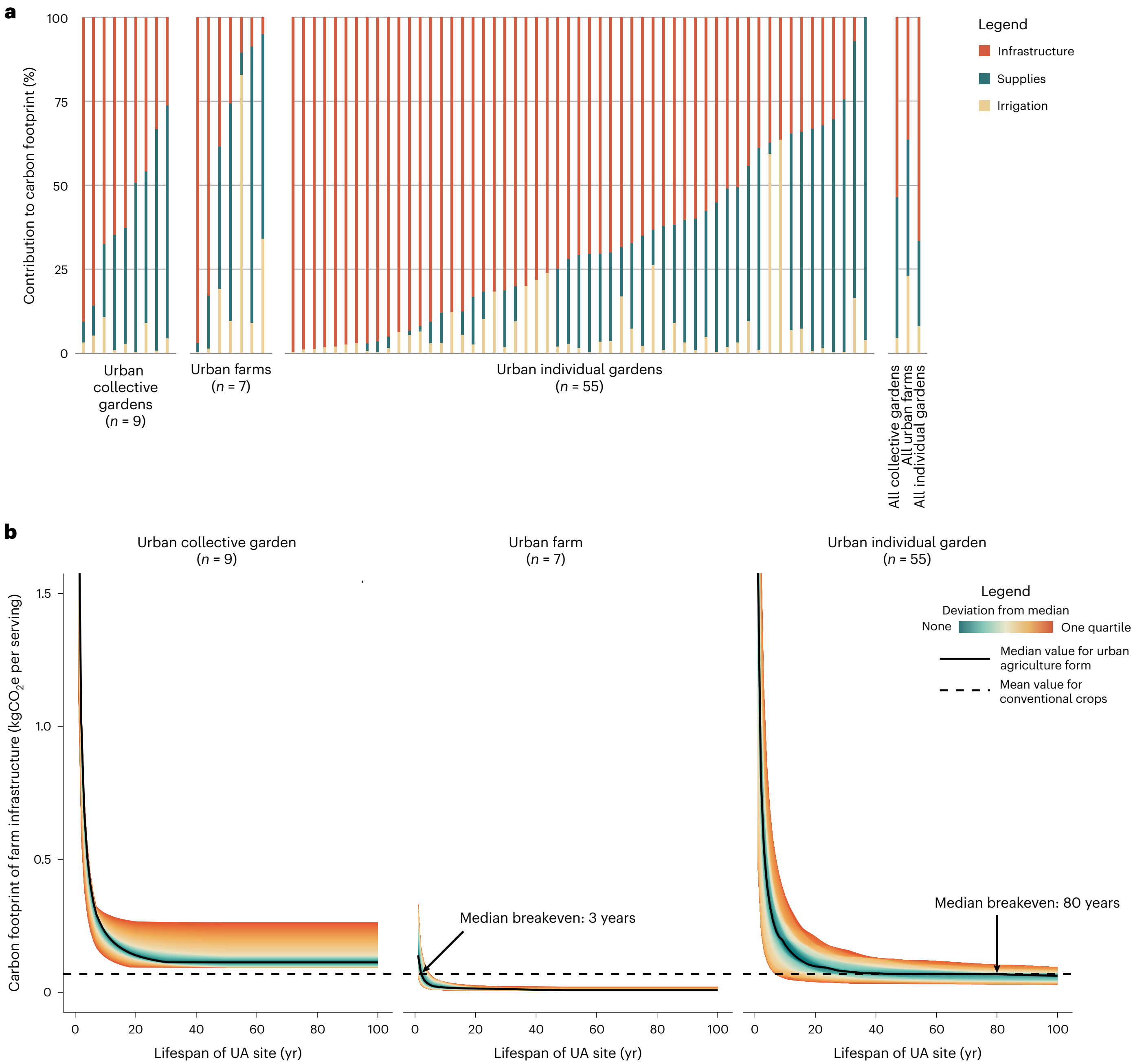

Fig. 3 | Infrastructure and carbon footprints at urban agriculture sites. **a**, Contributions of infrastructure, supplies and irrigation to GHG impacts. Supplies include fertilizer, compost, gasoline, weed block textile and so on. Irrigation is blue water used on food crops. Each column is an individual urban farm or garden. **b**, The black lines show the median infrastructure GHG impacts per serving of food produced at three types of UA space as a function of farm lifetime. The dashed lines show GHG impacts per serving using conventional agriculture. Urban farms amortize infrastructure investments after only three years. Individual gardens take decades, and collective gardens never break even.

farms used between three and five times as many synthetic nutrients as the average UA site (0.18 g nitrogen per serving, 0.14 g phosphorus per serving, 0.23 g potassium per serving), although this is still a statistically significant saving relative to conventional systems ($P = 0.014$; Supplementary Table 5 provides a breakdown by UA site type).

Compost at our farms is primarily derived from local food and yard waste. In some cases, this relationship is symbiotic, with farms receiving compost from external sources, whereas in others, internally generated food waste is composted on-site. In either form, composting saves carbon investment into potting soil (a heavy user of peat) and synthetic nutrients (energy-intensive and dwindling). However, poorly managed composting can exacerbate GHGs. The carbon footprint of compost grows tenfold when methane-generating anaerobic conditions persist in compost piles[39]. This is common during small-scale composting, and home compost is the highest-impact input on 22 of the 73 UA sites studied (Supplementary Table 4). Cities can offset this risk by centralizing compost operations for professional management or by training farmers on proper composting practices. In fact, we estimate that careful compost management could cut GHGs by 39.4% on sites that use small-scale composting.

Rainwater and graywater recycling for irrigation is a third area for symbiosis in UA[35]. In this study, more than 50 sites practiced rainwater recovery, but only four derived most of their irrigation this way. Instead, sites primarily used potable municipal water sources or groundwater

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wells, consistent with the underutilization of rainwater seen across past research[9]. Irrigation from these sources emits GHGs from pumping, water treatment and distribution, and this rose to as high as 83% of total emissions on one UA site. Cities should support low-carbon (and drought-conscious) irrigation for UA via subsidies for rainwater catchment infrastructure[40] or through established guidelines for graywater reuse[41].

UA sites should invest in social benefits. Unlike conventional agriculture, where food is typically the sole output, low-tech UA sites often blend food and social production[26,42–44]. A survey conducted with our farmers and gardeners[21] identified a variety of social benefits that align with past work[43]. UA practitioners overwhelmingly reported improved mental health, diets and social networks.

Similar to other multifunctional systems, such as organic agriculture, allocating impacts between UA's multiple benefits is challenging[45]. Because food and social benefits are co-products in UA, increasing social benefits can reduce impacts allocated to food[46]. This study takes a conservative approach by allocating all supplies and irrigation to food production, and infrastructure is allocated to food and social co-benefits based on interviews with farmers and standardized calculations (for example, 10% of a raised bed allocated to non-food if 10% of the area grows ornamentals).

Assuming farms adopt climate-friendly practices for their supplies, what percentage of infrastructure must be dedicated to non-food outputs to produce food with lower carbon intensity than conventional agriculture? Sensitivity analysis showed that most of our urban farms and individual gardens outperform conventional agriculture when more than 90% of infrastructure impacts are allocated to non-food services (Supplementary Fig. 3).

Although this threshold appears high, evidence suggests this is attainable. Cost–benefit analysis of a collective garden in the UK estimated that social benefits, such as improved well-being and reduced hospital admissions, accounted for 99.4% of total economic value generated on-site[47]. Because emissions allocation often follows economic value generation[46], growing spaces that maximize social benefits can outcompete conventional agriculture when UA benefits are considered holistically.

Future research

This study assesses the carbon impacts of low-tech UA to identify strategies for reducing these impacts. Collaboration with citizen scientists was fundamental to achieving our large sample size and will probably contribute to other large-scale carbon footprints, material flow analyzes and lifecycle assessments (LCAs) of UA. These tools, however, require reliable data on farm inputs and outputs, the collection of which was hampered by the turnover of personnel and volunteers at UA sites. For example, incomplete recordkeeping made it difficult to collect reliable data on water consumption. To avoid this, future projects should provide continuous training, compensate citizen scientists for their efforts, and automate data collection (for example, with water meters). To maintain confidence in our results, we excluded indicators compromised by errors in data collection, instead focusing on indicators where results were consistent across sites and where differences were large.

Other challenges faced in this study led us to identify several key areas for future work in this space:

- Low-tech UA in cities with relatively cold climates, such as our case cities, is unlikely to replace fruits and vegetables for wintertime consumption. However, we did not model seasonal carbon dynamics of conventionally grown produce for lack of data, nor did we assess the environmental impacts of local, alternate supply chains that might compete with UA in the summer (for example, community-supported agriculture). This is particularly salient given our findings that excessive air freight may negate carbon advantages seen in conventional production. Modeling seasonal dynamics and assessing a wider array of rural food production systems can address these gaps[48].
- Although UA may increase the carbon intensity of fruits and vegetables, these foods account for a small share of total dietary carbon impacts, which are driven mainly by meat and dairy. Studies have shown that UA practitioners often reduce their intake of animal products[49]. Future work should quantify this trade-off between elevated carbon footprint in urban produce and shifting diets.
- Better data are needed on carbon fluxes of composting at UA sites. We found composting contributes substantially to the carbon footprint of UA (Supplementary Table 4). Despite this, little is known about differences in GHGs from various composting techniques[50,51]. Furthermore, the high application rates of compost in UA probably raises additional questions. For example, the effects of long-term composting on N_2O emissions are unclear, and strategic management of application scheduling and fertilizer combinations may be required to minimize emissions[52,53]. How the repeated use of compost affects soil carbon sequestration in raised beds is also unclear, although existing evidence suggests that compost-dependent systems may sequester substantial carbon[54,55]. Both topics warrant further study.
- Study of different case cities is needed to understand how low-tech UA performs across climates and seasons. Our UA sites are in temperate, wet cities in the global north. Impacts probably vary substantially across UA sites in more diverse climates. Furthermore, we only analyzed the 2019 growing season. Future work should include multiple years to develop a more representative snapshot of UA.
- UA produces social and food outputs. To allocate impacts between the two, we used interviews and surveys. LCA practitioners and social scientists can collaborate to develop methods to better assess UA co-products (for example, cost–benefit analysis[47]). Another way to consider this web of co-products is through a land-use lens, comparing UA to other urban land uses, such as housing, parks and industry[56]. LCA results can be sensitive to these allocation methods, which are particularly important for UA work. While we found that the most socially productive spaces studied (that is, collective gardens) are also the most carbon-intensive, variation in collective garden sites indicates that this is not a strict condition of social good provisioning. Careful allocation of impacts can help scholars and UA designers to construct socially productive spaces that have a lower carbon footprint per unit of food produced.

Conclusions

UA has numerous benefits, but this study suggests that even low-tech urban farms and gardens have high carbon footprints. Our results show that today's UA generally produces more GHGs than conventional agriculture, although this needs additional clarification in industrializing cities and in drier or warmer climates. High-production urban farms that focus on crops that are conventionally carbon-intensive (for example, greenhouse-grown or air-freighted) may offer one path to a more climate-friendly UA. Meanwhile, all UA sites must extend the useful life of infrastructure, reuse more materials, and maximize social benefits to become carbon-competitive with conventional agriculture. In other words, UA must be judiciously designed and managed to achieve climate goals. Next steps should include broader adoption of the best practices described, as well as a suite of future research that will help to expand and refine this list of best practices. Because of its critical social, nutritional and place-based environmental benefits, UA is likely to have a key role to play in future sustainable cities, but important work remains to be done to ensure that UA benefits the climate as well as the people and places it serves.

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Methods

UA carbon footprint via LCA

Goal and scope. We used LCA to estimate the carbon footprints of 73 urban farms and gardens practicing low-tech UA in industrialized cities in the global north. LCA is a widely used method to estimate the environmental impacts of a good or service across its entire value chain[46,57]. The goal of this LCA was to quantify the climate impacts of fruits and vegetables produced at an urban farm. The scope of analysis was farm to city for both UA and conventional comparisons. We considered emissions throughout the lifecycle of the materials used to support food growth, and accounted for food waste using United States Department of Agriculture (USDA) estimates[58–60]. Consumer travel was excluded here as we assumed consumers travel equivalent distances to UA sites or grocery stores.

We evaluated GHG intensity per kilogram of fresh crop to compare between specific crops (Supplementary Information). To account for heterogeneity across UA sites and to facilitate comparisons with the 'basket' of conventional produce available in each country, we also calculated carbon intensity per serving. A serving is the recommended mass of a crop, as defined by nutritionists and doctors, that an individual should consume to align with national dietary guidelines (we used USDA values to unify servings across countries). Servings convert different crops to a single, comparable unit based on their nutritional content, which is similar to converting foods to caloric content[61], while also considering macro- and micronutrients. We used the USDA Food Patterns Equivalents Database[62] (FPED) to convert harvested crops to servings, including corrections for food preparation published in the USDA Food Intakes Converted to Retail Commodities Databases (FICRCD)[58]. Servings were calculated by converting each food product to servings of fruits and vegetables using both an FPED servings count and an FICRCD conversion value, which converts fresh food to consumed food (that is, accounting for peeling and so on). For example, the total fruit servings of any given food were calculated by multiplying the yield in kilograms by the FICRCD conversion, then multiplying this new value by FPED servings (which must be multiplied by 10 to convert from servings per 100 g to servings per kilogram). All equivalencies between crops grown on-site and standardized commodities are based on the USDA Food and Nutrient Database for Dietary Studies[63]. The relevant equivalencies are provided in the Supplementary Information as part of the SI Code and Inputs–'Crops_AllocationCodebook_Current.csv'.

Case studies and typology. Our study focused on UA sites in five countries—France (Paris and Nantes), Germany (Ruhr-Rhine metropolitan region), Poland (Gorzow Wlkp), the United Kingdom (London) and the United States (New York City)—as part of the Food–Energy–Water Meter project run collaboratively by universities in each country (for details, see refs. 14,64). UA sites were selected to represent a breadth of forms of low-tech UA.

UA projects vary widely in their goals and production systems. It is difficult to classify UA projects into distinct groups, and many typologies have been presented in the literature[65–67]. The FEW-meter team developed an internal typology based on input from farmers and gardeners at our sites[9]. These sites are categorized according to their goals, their management systems and their funding structures, forming four divisions: urban farms, individual gardens, collective gardens and mixed model sites.

In this typology, urban farms are primarily commercial enterprises, managed by professional farmers to produce food (producing an average of 4,161.98 kg on site, enough vegetables to feed 40–50 people annually). On average, our urban individual gardens are relatively small, individually managed plots producing food for their owners and their friends and families (averaging 164.45 kg of produce annually). Urban collective gardens are socially productive spaces supported largely by volunteer labor or non-profit support, producing food for community benefit (an average of 1,384.70 kg annually) as a complement to broader community goals such as nature-based education, social justice and job training. Lastly, mixed model farms escape classification and are excluded from analyzes using this typology but are included in the results for UA as a whole.

Lifecycle inventory. We employed a citizen science approach, partnering with urban farmers and gardeners in case cities to document inputs and outputs at UA sites. Inputs come in many forms, which we divided into three overarching categories: infrastructure, supplies and irrigation. Infrastructure includes relatively permanent aspects of each site, such as raised beds in which food is grown or pathways between vegetable plots. Supplies consist of regular inputs to the farm or garden, including compost, fertilizer and gasoline. Irrigation includes any water applied to crops.

Infrastructure inputs were calculated by researchers in collaboration with gardeners during tours of the gardens. Researchers directly measured or estimated volumes of materials with the help of gardeners (for example, approximating the depth of a concrete path). During site tours, researchers also cataloged climate-friendly infrastructure such as solar panels. Supply and irrigation inputs were logged online or in written diaries using a system co-developed with participants[14]. Participants recorded the daily inputs and harvests from their site, keeping track of what they added and extracted throughout the growing season. In preparation for the impact assessment, unusual units (for example, one slab of concrete cladding) were converted to mass or volume using online product data to ensure compatibility with LCA databases[68].

Lifecycle impact assessment. We determined the environmental impact of UA inputs and outputs using EcoInvent 3.8 (ref. 68) and the PEF 3.0 midpoint indicators (specifically, global warming potential at 100 years). These impacts were exported from SimaPro to a csv file and then imported into R. In R, we used linear algebra to calculate the lifecycle GHG footprint of each UA site, adding up potential impacts for material extraction, production and use and end-of-life stages for all inputs. For end-of-life, we used the cutoff principle, whereby landfilling and incineration impacts were assigned to the current lifecycle, and recycling impacts were assigned to the following lifecycle. This thinking was applied to recycled inputs on our sites. We also tested crediting the systems for avoided impacts from recycling and found that it did not influence the directionality of the results nor the statistical analysis (Supplementary Table 6 shows the effects of this assumption on compost impacts). We divided total impact by total harvest to calculate the per-serving impacts at the farm level. These impacts were also assigned to individual crops through co-product allocation, as discussed in the following.

All data were processed in R[69], and both data and code are available in the Supplementary Information.

Key dimensions of LCA and sensitivity analyses. Our LCA used three major assumptions:

- Allocation between food products
- Percent of site impacts allocated to food
- Age of farm/garden at time of removal

The results in the main text are averaged across all four allocation schemes and averaged across different UA site lifetimes (1 to 100 years). We used interviews to determine the baseline percent of impacts allocated to food (versus co-products) for each UA site. Both percent impacts to food and age of farm are explored in the Results and discussion. We discuss each of these important aspects of our model in the following sections.

Allocation between food products. Although the average conventional farm employs large, mono-cropped fields to produce vegetables, low-tech UA typically hosts polycultures of vegetables, fruits and even small livestock. To quantify the carbon footprints of urban crops, we must therefore allocate the farm-level impacts between different farm products. We treat the fruits, vegetables and social output of our UA sites as co-products, allocating the farm-level impacts to crops based on their contribution to the total farm production.

Food production is measured in terms of mass, caloric, nutrient (NRF 9.3 (ref. 70)) and economic output, and impacts are allocated to individual crops based on the value of the harvest of that crop (for example, if 10 kg of tomatoes are harvested and 100 kg are produced in total, tomatoes would be allocated 10% of the food-related impacts using mass allocation). Mass allocation depended on the harvests recorded by farmers, whereas caloric and nutrient allocations used USDA food composition data[59,62,63] to convert harvests to calorie and nutrient outputs. Economic allocation was localized to each city, using prices at nearby grocery stores to estimate the economic value of UA crops.

Our model is generally robust to allocation decisions. In most cases, all four allocation schemes produce results within a factor of two. However, crops with substantial variation between caloric density, nutrient density and value per kilogram saw more variation (for example, a factor of six with potatoes). Nonetheless, no allocation decision changes the direction of the relationship between a conventional product and an urban one. When assessed across allocation scenarios, all urban crops have higher carbon footprints than their conventional counterparts.

Allocation between co-products. UA often has a variety of co-products, both material and immaterial. Allocating between these products is both challenging and extremely important to the overall findings of an UA LCA. It is necessary to clarify how we made these allocations given the sensitivity of our results to the percent of infrastructure impacts allocated to food. Our baseline scenario for impacts to food is unique to each site. Through interviews with farmers and site visits, researchers used simple rules to estimate the percent of impacts from each piece of infrastructure to be allocated to food. For example, if half of a raised bed was used to grow ornamental crops, only half of the impacts of that raised bed were allocated to food. If an on-site pavilion was used mostly for events and occasionally for sorting food, then only a small percentage of the impacts of that pavilion were allocated to food. It is worth noting that sorting infrastructure are sometimes excluded from conventional vegetable LCAs. This omission probably has little influence on the results for large farms growing monocultures. We included this infrastructure here because of its importance in small-scale production systems.

To test the impacts of our allocation methods, we conducted a sensitivity analysis of the percent of impacts allocated to food and social outputs. We tested the effects of altering the infrastructure impacts assigned to food by varying this value between 0% and 100% (intervals of 5%). Break-even analyses are discussed in the main text and are shown in the Supplementary Information.

Farm longevity. Because of the strong influence of infrastructure on our results, we also tested the sensitivity of UA impacts to farm (and infrastructure) longevity. For most main-text graphs, we calculate the average impact of food produced at each site if it was moved anywhere between 1 and 100 years after establishment (intervals of one year). In the Results and discussion, we display the break-even points for infrastructure on each type of UA site.

We use 100 years as the maximum land tenure considered, because that is the longest lifespan of any material used on one of our farms. Some of the oldest allotment gardens in Europe can trace their roots to the nineteenth century, and several gardens in the eastern United States began as Victory Gardens during World War II, but little of the original structures remain on these sites, and 100 years is a highly conservative upper limit for UA infrastructure.

Climate-friendly UA. Climate-friendly UA sites were defined as farms that had lower GHG emissions per serving than conventional agriculture when averaged across all sensitivity scenarios. The total number of scenarios per farm is given by

4 allocation schemes × 21 values of percent impact × 100 ages = 8,400 scenarios

As defined, climate-friendly farms have a lower GHG impact than conventional agriculture when averaged across all 8,400 scenarios.

Synthetic fertilizer inventories. In tandem with the LCA, we also collected data on synthetic fertilizer application, tracking the flows of synthetic nutrients into food products. We tracked the mass of synthetic nutrients consumed on all sites and allocated them evenly across all servings of food produced on the sites. Data and code are available in the Supplementary Information.

Conventional agriculture comparison

To compare carbon footprints of UA to conventional agriculture, we quantified the GHG footprint of the five most consumed fruits and vegetables (by mass) in each case-study country. We chose the top five fruits and vegetables because they collectively make up more than three-quarters of fruit and vegetable intake in each country of interest. Using data from the Food and Agriculture Organization of the United Nations (FAO), we identified the countries that collectively serve as sources of at least 90% of each of these fruits and vegetables. For example, 96% of onions available in German supermarkets are grown in Germany (71%), Spain (13%) and the Netherlands (12%). Taking a weighted average (weighted by percent of sales) of the carbon footprint of onions grown and shipped from each of these sources, we approximated the carbon footprint of a typical onion in a German supermarket. We can then compare these supermarket onions to onions grown on our sites.

Because crops are often imported from multiple locations, we required data on 107 unique crop–country combinations of vegetables consumed in the five countries. To quantify the carbon footprints, we sought to identify either (1) at least three studies (LCAs or carbon footprints) for each crop consumed or (2) a systematic summary of the impacts of a particular crop in each consuming country. We used this system of focusing on large reviews or multiple case studies to iron out differences between cases and identify a relatively representative mean value for the carbon footprint of each vegetable in each consuming country. In a few exceptional cases, we could not locate a summary and only identified two supporting studies. To quantify nutrient inputs, we sought at least one study of nitrogen, phosphorus and potassium inputs into conventional agriculture for each unique crop–country combination. Most crop–country combinations were available from existing summaries. LCAs useful for this summarization come in three forms:

- Farm-to-supermarket LCA of a particular product sold in one of our countries of interest (for example, Agribalyse analysis of strawberries sold in France, which already accounts for inputs across countries)
- Farm-to-supermarket LCA of a particular product that matches one of our country–country combinations (for example, an analysis of Spanish strawberries imported to England)
- Farm-to-farm-gate LCA of a particular product grown in one of our producing countries (for example, an analysis of Spanish strawberries that ends at the farm gate, to which we can manually add estimates of food waste, travel and supermarket impacts)

The Supplementary Information provides a database of conventional vegetable impacts developed to support this study, as well as the R code used to compare these values to UA crops. In the case of farm-gate studies, we employed reasonable estimates of food waste, travel and supermarket impacts. Specifically, we assumed food waste rates as reported by peer-reviewed articles for the United States, United Kingdom and European Union (EU). We used EU wastes from ref. 71

(3.8% in distribution, 1.3% in retail), UK waste rates from ref. 72 (1.6% in processing, 9.6% in retail) and US waste rates from a Commission for Environmental Cooperation white paper[73] combined with USDA estimates[60] of overall waste (3.9% in distribution, 2.5% in retail). For travel, we assumed that vegetables were transported via semi-trailer and ocean freight, as most fruits and vegetables are not perishable enough to justify air freight[20]. We tested the sensitivity of our results to this assumption and found that UA in general is still statistically significantly more carbon-intensive than conventional solutions even when conventional crops are air-freighted. The exception to this, individual gardens, is discussed in the Article. In all travel cases, we assume travel from the capital city of each country or from the largest city in a major agricultural export region (details are provided in the Supplementary Information). Using emissions estimates from SimaPro and distance estimates from online tools, we added travel impacts to farm gate studies based on each unique country combination (for example, products traveling from South Africa to Dortmund were estimated to travel 530 km by road and 11,036 km by sea). Finally, we used a generic supermarket impact value from ref. 74 to supplement the farm-gate studies with supermarket emissions.

Because our UA sites produce a wide variety of crops, we also created a 'basket' of crops for each country, comprising the top five fruits and the top five vegetables (as well as independent fruits and vegetables). Using a weighted average (weighted by the percent of consumption in that country—by mass), we calculated the impacts per serving for each of these country-level baskets. Finally, we calculated the average conventional produce impact by averaging across these baskets.

We conducted two-sided *t*-tests at the 0.05 significance level to test for statistically significant differences between urban and conventional crops and country-level baskets. We used a false discovery rate correction to adjust for multiple tests. All assessment was done in R, and all code is available in the Supplementary Information.

Farmer survey

We surveyed farmers at each UA site on their motivations for practicing UA. We used survey results to clarify the relationships between motivations and UA carbon footprints (Supplementary Table 3). Participants responded to 'People have many different motivations for gardening and farming. How important is each of the following reasons for gardening/farming to you?' on a Likert-type scale ranging from 'not important at all' to 'very important'. The list of motivations assessed was based on previous literature[21,43], and the survey was translated into the local language for each site. For more details on survey administration in each country, see an existing analysis of the survey by Kirby and others[21].

Reporting summary

Further information on research design is available in the Nature Portfolio Reporting Summary linked to this Article.

Data availability

All data used for this study are available in the Supplementary Information. See the attached Supplementary Information for more details and data. Public datasets, including FPED, FICRCD and FNDDS, are available for download online via their citations. EcoInvent data used for lifecycle impact assessment are proprietary and may be accessed via purchase.

Code availability

All code used for this study is available in the Supplementary Information.

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Author contributions
J.K.H. and B.P.G. provided conceptualization, methodology, formal analysis, data curation, contributed to writing the original draft, provided review and editing, and validation. J.P.N. provided conceptualization, resources, contributed to writing the original draft, provided review and editing, supervision, funding acquisition and project administration. E.D. provided conceptualization, methodology, investigation, data curation, contributed to writing the original draft, and provided review and editing. S.C. provided conceptualization, investigation, resources and review and editing. R.F.-K. provided conceptualization, investigation, resources and review and editing. B.G. provided conceptualization, investigation and review and editing. R.T.I. provided conceptualization, investigation and review and editing. A.F.-L. provided conceptualization, investigation, resources and review and editing. L.P. provided conceptualization, investigation, resources, review and editing, and project administration. V.S. provided conceptualization, investigation and review and editing. K.S. provided conceptualization, investigation, resources, review and editing. N.C. provided conceptualization, investigation, resources and review and editing.

Competing interests
The authors declare no competing interests.

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☐	☒	A description of any assumptions or corrections, such as tests of normality and adjustment for multiple comparisons
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Our web collection on statistics for biologists contains articles on many of the points above.

Software and code

Policy information about availability of computer code

Data collection	No softwares were used for data collection, though AirTable was used internally by the research team to record data across case studies.
Data analysis	SimaPro 9.3.0.3, EcoInvent 3, and R were used to conduct this analysis. R code was most recently tested in R 4.3.0.

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Human research participants

Policy information about studies involving human research participants and Sex and Gender in Research.

Reporting on sex and gender	We do not report on sex or gender as part of this research. We do not have a sufficient sample to assess the impact of farmer/gardener sex or gender on environmental impact of urban agriculture.
Population characteristics	Population: Urban farmers and gardeners in six cities. Sampling frame: Urban farmers and gardeners affiliated with any of a number of urban agriculture organizations partnered with project team members. See below for additional details.
Recruitment	In most cases, farmers and gardeners were recruited through partner organizations in each city. For example, in several cities, informational gatherings were held with members of community garden or allotment garden collaboratives. Attendees were then invited to contact the researchers if they wished to participate. In other cases, farm or garden leaders were contacted individually or leaders of garden organizations identified appropriate participants directly. Environmental impact participants were self-selected by their willingness to participate in ongoing recording practices as part of the citizen science methods. Participants in some of the case study countries were provided nominal compensation.
Ethics oversight	Ethical review was handled by the lead institution in each country according to the rules and regulations therein (e.g., US Approval via the City University of New York, IRB File #2018-0233; German Approval via ILS – Institut für Landes- und Stadtentwicklungsforschung, May 2018).

Note that full information on the approval of the study protocol must also be provided in the manuscript.

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☐ Life sciences ☒ Behavioural & social sciences ☐ Ecological, evolutionary & environmental sciences

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Behavioural & social sciences study design

All studies must disclose on these points even when the disclosure is negative.

Study description	This project sought to quantify the social and material inputs and outputs of urban farms and gardens. To trace social goods, we deployed surveys using Likert-type measures to allow participants to self-assess impacts. The survey is available in supplementary information and more detail on design is available in Kirby et al., 2021. To trace material flows, we employed "growing diaries" where participants recorded the inputs and outputs of their sites. For more details, see Caputo et al., 2020. Caputo, S., Schoen, V., Specht, K., Grard, B., Blythe, C., Cohen, N., Fox-Kämper, R., Hawes, J., Newell, J., & Poniży, L. (2020). Applying the Food-Energy-Water Nexus approach to urban agriculture: From FEW to FEWP (Food-Energy-Water-People). Urban Forestry & Urban Greening, 126934. https://doi.org/10.1016/j.ufug.2020.126934Kirby, C. K., Specht, K., Fox-Kämper, R., Hawes, J. K., Cohen, N., Caputo, S., Ilieva, R. T., Lelièvre, A., Poniży, L., Schoen, V., & Blythe, C. (2021). Differences in motivations and social impacts across urban agriculture types: Case studies in Europe and the US. Landscape and Urban Planning, 212, 104110. https://doi.org/10/gjsqbq
Research sample	72 urban farms and gardens (and associated farmers and gardeners). Relevant demographics: 53% male, 41% college-educated, 60% native born in country of survey.
Sampling strategy	Sampling was convenience. Farmers and gardeners were recruited through partner organizations in each city (see Recruitment). Environmental impact participants were self-selected by their willingness to participate in ongoing recording practices as part of the citizen science methods. Participants in some of the case study countries were provided nominal compensation.
Data collection	Material flow data (farm inputs and outputs) were collected via "diaries" provided to the farmers and gardeners which were periodically returned to researchers and recorded in an AirTable database. Surveys were conducted in-person in the local language of the site. Only the researcher and the participant were present for the survey. More information can be found in the study methods.
Timing	March 2019-November 2020
Data exclusions	Several gardens were excluded from final analysis because of incomplete data reporting or specific inconsistencies with data.

Non-participation	As overall sampling of farms and gardens was convenience, non-participation rates cannot be calculated. There was no survey non-participation among material flow participants.
Randomization	Randomization was not employed. Only descriptive analysis was reported in this manuscript.

Reporting for specific materials, systems and methods

We require information from authors about some types of materials, experimental systems and methods used in many studies. Here, indicate whether each material, system or method listed is relevant to your study. If you are not sure if a list item applies to your research, read the appropriate section before selecting a response.

Materials & experimental systems

n/a	Involved in the study
☒	☐ Antibodies
☒	☐ Eukaryotic cell lines
☒	☐ Palaeontology and archaeology
☒	☐ Animals and other organisms
☒	☐ Clinical data
☒	☐ Dual use research of concern

Methods

n/a	Involved in the study
☒	☐ ChIP-seq
☒	☐ Flow cytometry
☒	☐ MRI-based neuroimaging