RSC Sustainability

ROYAL SOCIETY OF CHEMISTRY

PAPER

View Article Online
View Journal | View Issue

Cite this: *RSC Sustainability*, 2023, **1**, 960

Will the circle be unbroken? The climate mitigation and sustainable development given by a circular economy of carbon, nitrogen, phosphorus and water†

Patrick McKenna, *[a] Fiona Zakaria, [b] Jeremy Guest, [cd] Barbara Evans [b] and Steven Banwart [a]

Received 5th December 2022
Accepted 21st April 2023

DOI: 10.1039/d2su00121g

rsc.li/rscsus

Closing the loop in the flow of C, nutrients and water between agriculture, the human diet and sanitation services offers benefits for humanity across multiple platforms of public health, food security and climate mitigation. This study assesses these benefits by describing the hypothetical scenario of a global, 'fully functional' circular economy, in which 100% of C, N and P were recovered from human excreta and returned to agricultural soil. Crop nutrient demand is calculated and compared with that which could be recovered, and greenhouse (GHG) emissions from fertilizer production, fertilizer application and sanitation services are presented, as are freshwater availability and crop irrigation requirements. These are considered to analyse the broader effects of this circular economy that is driven by dietary nutrition demand on climate change, the provision of sanitation services and crop irrigation, in 2022 and with projections to 2030 and 2050. We find the capacity of the circular economy to deliver crop nutrients and mitigate GHG emissions varies by region. Some regions benefit from supplementing conventional mineral fertilizers with excreta-derived fertilizers, others from reducing GHG emissions from sanitation services through improved resource recovery rates. A hypothetical, fully functional circular economy that recovers all excreta nutrient C, N and P would reduce global GHG emissions from N and P mineral fertilizer production and application by 140 Tg CO_2 equivalents (CO_2 e) per year in 2022 (~12% of total emissions from mineral fertilizer production and application) and provide a maximum of 104 Tg C per year for sequestration in global cropland (~12% of estimated annual soil C sequestration potential). A portion of this sequestered C will return to the atmosphere *via* soil respiration, however, with co-benefits to other soil functions such as crop nutrient fertility. The maximum potential reduction in GHG emissions from sanitation services through these measures would bring reductions of 445 Tg CO_2 e per year in 2022, rising to 562 Tg CO_2 e in 2050. Our results provide evidence to guide specific regional policy on reducing GHG emissions, offsetting mineral fertilizer use and optimizing municipal water use using the circular economy.

Sustainability spotlight

Sanitation technologies are fundamental for public health, but here we argue they can also sustain global agriculture and mitigate climate change, with variation in this capacity among the SDG regions. Our research focuses on how nutrient recovery from human excreta can mitigate climate change by reducing fertilizer requirements when returned to cropland, but also by limiting CH_4 and N_2O emissions from sanitation services (*e.g.* sewers or latrines), which occur in significant volumes in some parts of the world. We believe our data and analysis can inform policy for sanitation development, particularly in the Global South, where almost all human excreta nutrients go unrecovered, and many cannot access functional sanitation services.

[a]*Global Food and Environment Institute and the School of Earth and Environment, University of Leeds, Leeds, LS2 9JT, UK. E-mail: patchmck@gmail.com*

[b]*BioResource Systems Research Group, School of Civil Engineering, University of Leeds, Leeds LS2 9JT, UK*

[c]*Department of Civil and Environmental Engineering, University of Illinois at Urbana-Champaign, Urbana, IL, USA*

[d]*DOE Center for Advanced Bioenergy and Bioproducts Innovation, University of Illinois at Urbana-Champaign, Urbana, IL, USA*

† Electronic supplementary information (ESI) available. See DOI: https://doi.org/10.1039/d2su00121g

Introduction

Sustainable Development Goal 6 (SDG6) aims to achieve access to adequate and equitable sanitation and hygiene for all. The motivation is primarily to improve public health and welfare, particularly in urban areas, but researchers are increasingly viewing sanitation technologies as a means of addressing global challenges beyond these goals. Sanitation access combined with resource recovery may provide a platform for sustainable development that contributes to additional SDGs. Examples being considered include the valorisation of human waste into nutrients for agriculture[1] or feedstock for energy,[2] the maintenance of ecosystem services and environmental conservation,[3] and the establishment of viable industries around waste valorisation.[4] Climate mitigation efforts associated with sanitation include nutrient recovery to reduce agricultural reliance on mineral fertilizer and the high fossil fuel demand for its production, development of sanitation services that limit GHG emissions, and recovery of C for utilization or sequestration that prevents release to the atmosphere as GHGs.[5] Studies documenting GHG emissions from pit latrines,[6] or the climate mitigation potential of composting human waste[7] address this, but the potential impact of a fully functional circular economy of excreta-derived nutrients on global GHG emissions remains undescribed.

A circular economy of nutrients derived from human excreta (subsequently termed excreta nutrients) can mitigate climate change in multiple ways. Circularity is defined by the flow of nutrients between soil, crops, diet, waste and back to soil. The human diet is supported by agricultural inputs of key nutrients (N and P) beyond those available from cultivation of legumes (N) and plant uptake from soil minerals (P). Additional fertilizer inputs are produced or mined for mineral fertilizer using fossil fuel energy, emitting significant amounts of GHGs in their production (CO_2) and application (N_2O for N fertilizers). Returning more excreta nutrient N and P to cropland may reduce the need for mineral fertilizer production, thus lowering production emissions, whilst climate-smart processing techniques for excreta-derived fertilizer can limit N_2O emissions.[8]

Some excreta-derived fertilizers may also be rich in organic C, which can improve soil functionality when added[9] and offers potential for C sequestration in croplands.[10] Organic C is not technically a plant nutrient and is often omitted from studies on resource recovery from wastewater, but its addition to soil can sustain yields as fertilizer does, particularly through addition of organic matter with associated nutrient elements in arable systems without ley periods or other organic amendments. The amount of potentially recoverable C in global wastewater is likely far higher than N or P, as the amount ingested is significantly higher and about 12% of this is excreted (the rest is respired as CO_2 in metabolism).[11] Recovery of this resource offers a promising means to sustain soil fertility, but this is currently unrealized, as over 80% of human waste entering sanitation services ultimately goes untreated and unrecovered;[12] this proportion and volume is projected to increase in the coming decades as populations increase in the regions lacking substantial sanitation infrastructure.

Recovery of C may also reduce GHG emissions *via* the recovery process itself, as sanitation services (sewers, septic tanks, latrines *etc.*) can emit significant masses of GHGs.[13,14] Unrecovered and untreated inputs of C and N contained in human waste can transform to CO_2 (carbon dioxide), CH_4 (methane) and N_2O (nitrous oxide) under certain environmental and management conditions. Around one third of the C undergoing treatment in wastewater treatment plants (WWTPs) is emitted as CO_2, whilst the remainder is transformed into microbial biomass resulting in biosolids.[15] A greater proportion of the C entering decentralized systems such as pit latrines without advanced treatment processes is predicted to be degraded under anaerobic conditions resulting in the production of CH_4, a more potent greenhouse gas per mass of C than CO_2. Projections for urbanisation in low and middle income countries (LMICs) between 2022 and 2050 make this recovery more salient, as urban populations in these countries are more reliant on decentralized sanitation services, which can produce significant volumes of GHGs when waste C and N goes unrecovered.

Agriculture and sanitation require water to function and conventional sewerage carries both human excreta and household sullage water where the dilution of excreta nutrients in the resultant wastewater can make their recovery more challenging. Onsite sanitation systems including pits and tanks also often collect large volumes of water; they are very common in many LMICs and comprise much of the current and planned investment in sanitation. Investment in sanitation is essential for SDG 6, but this may create problems in other policy sectors if the improvements demand large volumes of water. Water resource divestment into sanitation may pressurize agriculture in dry parts of the world, but water recovery from wastewater may equally provide a valuable resource for irrigation or fertigation in cropland. A functional circular economy must therefore provide sufficient water for agriculture whilst helping maintain water resources required for drinking water and other uses.

This study analyses the links between excreta nutrient recovery and climate change as a contribution to SDG 13 (Climate Action) and discusses assesses potential synergies with SDG 6 (Clean Water and Sanitation) and SDG 2 (Zero Hunger). We hypothesize a fully-functional circular economy of N, P and C from human excreta (*i.e.* all N, P and C recovered and returned to agriculture) and calculate the commensurate reduction in GHG emissions from mineral fertilizer production and application, as well as from sanitation services themselves. We do this first by calculating how much N, P and C is recoverable from human excreta globally and comparing this to mineral N and P fertilizer application and requirement. We then compare recoverable C to potential sequestration in cropland. GHG emissions from mineral fertilizer use and application is calculated and the potential reduction offered by supplementation with excreta-derived N and P is also calculated. The potential reduction in GHG emissions by eliminating currently unrecovered nutrients in sanitation utilities are also calculated in both urban and rural areas globally and the impact on available water resources are also considered. We focus our analysis on the 8

SDG regions[16] and comparison of results between them, as evidence of the need for policy development. We extend the analysis to assess the potential for climate change mitigation by reducing the dependence of agriculture on mineral fertilizer use and application, and discuss the implications for pressures on soil and water resources to maintain food security at time of writing and through to 2030 and 2050.

Materials and methods

A detailed explanation of methods used is given in the ESI.† A brief description is given here.

Mass amounts of C, N and P excreted in SDG regions were calculated using the FAOSTAT Food Balance Sheets.[17] This UN database provides national dietary intake, broken down into consumption of all food groups and nutritional components (protein intake *etc.*). Food consumption data was converted to C, N and P consumption data using published conversion factors[2,11,18] and corrected for waste.[19] Consumption data was assumed to equal excretion for N and P, as these elements are homeostatically controlled. Literature conversion factors were applied to food intake to calculate C consumption, and it was assumed that 88% of this was respired and 12% entered the excreta.[18] The potential for annual C sequestration in regional cropland soil was taken from Zomer *et al.*[20] Mineral fertilizer application rates were calculated using FAOSTAT data and crop nutrient requirement was calculated using advisory application rates from IFA[21] (International Fertilizer Association). GHG emissions from fertilizer production were calculated using the methodology of Blonk Consultants,[22] using data from IFA (International Fertilizer Association) and EFMA (European Fertilizer Manufacturing Association). This method applies emission factors to the production of different types of fertilizer in different regions (regional fertilizer production data was also taken from FAOSTAT).

The reduction in emissions from mineral fertilizer production given by the circular economy was calculated by reducing the mineral fertilizer production emissions by the amount required to produce that equivalent to all of the excreta N and P from sanitation services. The reduction in emissions from mineral fertilizer application given by the circular economy was calculated by reducing the mineral fertilizer application by the amount which could be recovered, then calculating the associated reduction in emissions from less mineral fertilizer use. The estimated application emissions from prospective wastewater-derived N fertilizers were added to this value using an emission factor for biosolids from the literature.[23] The combined values and the total percentage reductions offered by the circular economy were calculated by subtracting the calculated 'circular economy' emissions from the 'no recovery' emissions and expressing as a percentage. This means that if global emissions from N and P fertilizer production and N application were 1179 Tg CO_2 e in 2022, and this were reduced to 1039 Tg CO_2 e by recovering all excreta N and P and thus eliminating some of the need for fertilizer production (CO_2), as well as some mineral fertilizer application emissions (NO_2 expressed as CO_2), we then claim the circular economy can reduce fertilizer production/application GHG emissions by 11.9%, calculation show below.

$$\left[\frac{(1178.9 - 1038.5)}{1178.9}\right] \times 100 = 11.9\ \%$$

GHG emissions from sanitation services were calculated using the IPCC estimation methodology for wastewater emissions[24] with some modifications to improve accuracy (see ESI†). This procedure estimates the methane and nitrous oxide emissions from sanitation utilities (sewers, latrines, septic tanks *etc.*) using assessments of their C and N input and expected regional emission factors. Total renewable water resources (TRWR), internal renewable water resources (IRWR), total freshwater withdrawal and agricultural freshwater withdrawal were calculated using data from AQUASTAT.[25] This UN database gives national data for these water resources. All national data were aggregated by the studied SDG regions in the same fashion as the other data previously described. This was done to frame discussion of the results as evidence for policy to support attaining SDGs within the regions future projections to 2030 and 2050 for all data were calculated by dividing each figure by the regional human population in 2022 and multiplying by the projected population in 2030 and 2050. This estimate assumes nothing changes apart from the population and that change is commensurate with population growth. All of our calculations on based on source data from FAOSTAT, AQUASTAT, IPCC and WHO, which uses either Tier 1 and Tier 2, both of which are no applied with uncertainty analysis.

The details of all calculations reported in the results and discussion are given in the SM, global calculations are also given in the SM as this paper focuses on regional considerations within the SDG framework.

Results

Most SDG regions showed significant disparity between rates of N and P excretion and agricultural application of N and P in 2022, as well in the 2030 and 2050 projections (Fig. 1a and b). Recovery of excreted N and P would not effectively replace or even supplement mineral fertilizer use in industrialized regions like North America & Europe, or Eastern & South-Eastern Asia, but Sub-Saharan Africa and Oceania could almost entirely replace current and projected rates of fertilizer application with excreta-derived N and P. Western Asia & Northern Africa could also significantly supplement their mineral fertilizer application rates with excreta-derived N and P. Rates of N and P excretion were more closer to crop fertilizer requirements in the Asian regions, but here crop requirements represent recommended applications rates and not precise crop physiological requirements. Fertilizer requirement was shown to significantly exceed that which could potentially be recovered in Sub-Saharan Africa, and this disparity was shown to increase in the 2030 and 2050 projections. Recovery of C was also shown to provide a greater percentage of the potential sequestering capacity of cropland in some places, for example the Asian regions (Fig. 2).

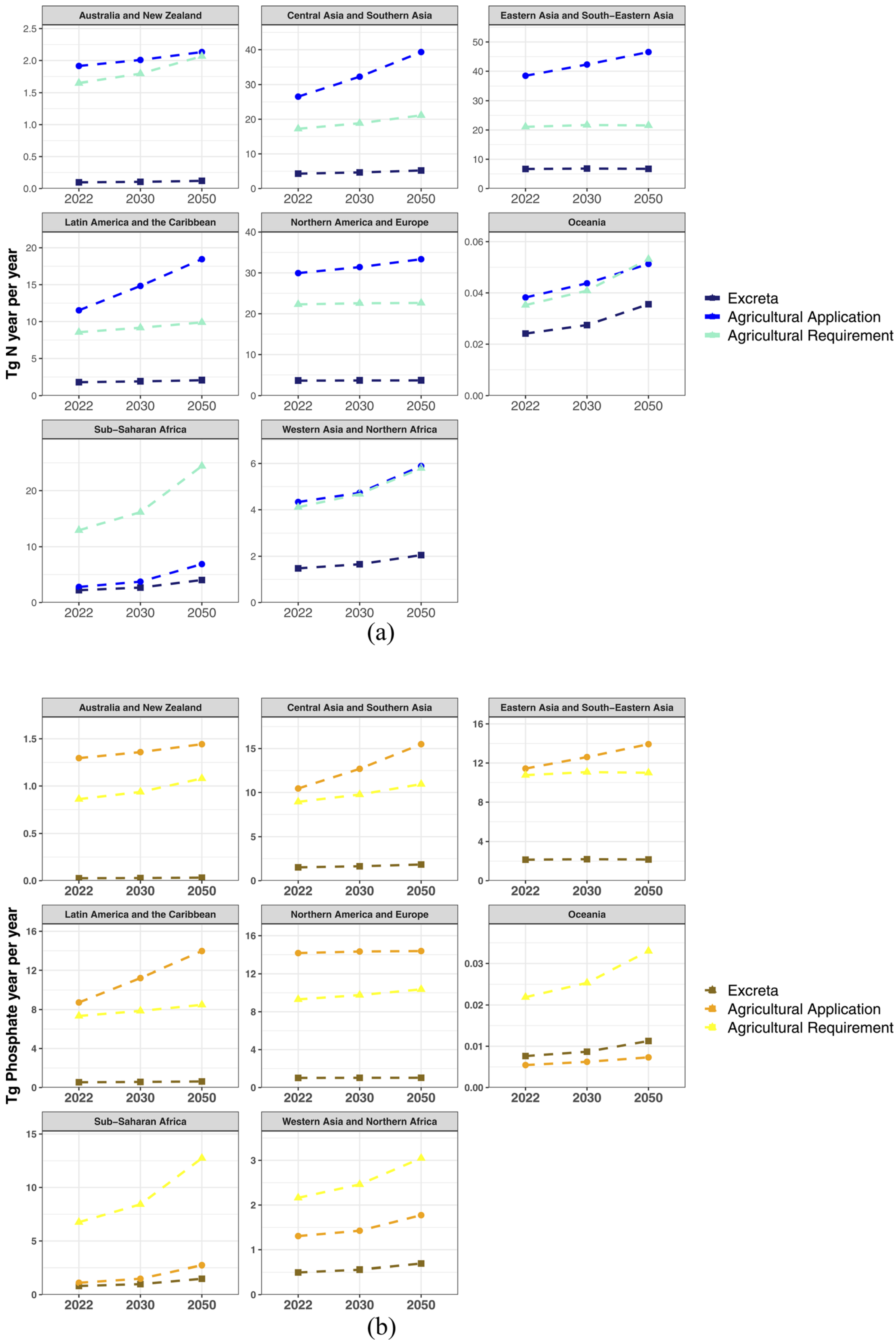


Fig. 1 (a) Nitrogen available for recovery in human excreta, total mineral nitrogen fertilizer applied to agriculture, and nitrogen fertilizer requirement of 17 key crops in Sustainable Development Goal Regions in 2022, with projections to 2030 and 2050. All units are Tg N per year. (b) Phosphate available for recovery in human excreta, total mineral nitrogen fertilizer applied to agriculture, and nitrogen fertilizer requirement of 17 key crops in Sustainable Development Goal Regions in 2022, with projections to 2030 and 2050. All units are Tg N per year.

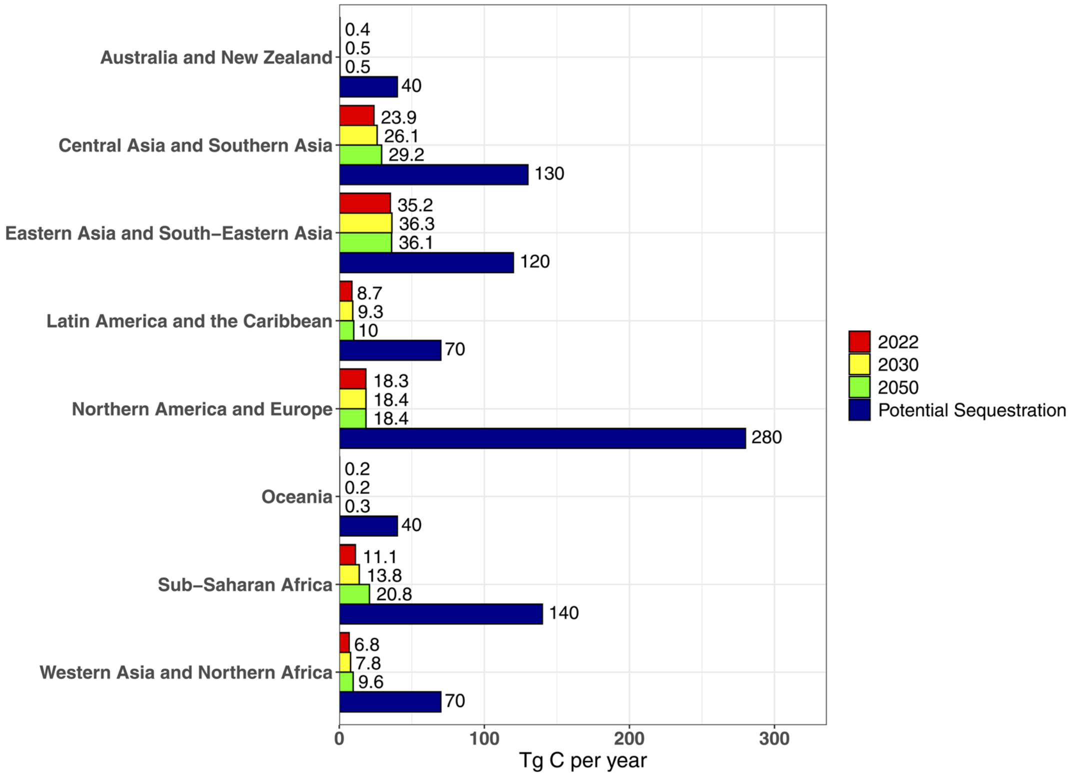


Fig. 2 Carbon available for recovery in human excreta and potential for carbon sequestration in cropland soil in Sustainable Development Goal Regions in 2022, with projections to 2030 and 2050. Potential sequestration values are taken from Zomer *et al.*[20] All units are Tg C per year.

Eastern & South-Eastern Asia and North America & Europe generate more annual GHG emissions than other regions from fertilizer production and application based on 2022 levels, and in both future scenarios for 2030. However, Central & Southern Asia was predicted to increase emissions to levels comparable to North America & Europe by 2030 (Fig. 3a). N_2O emissions, when converted to CO_2-equivalents (CO_2 e) from fertilizer application tended to be greater than CO_2 emissions from production in Europe and the Americas in all years considered, but this disparity was less significant in the Asian regions. Emissions from phosphate mining were far less than those from mineral N production in all cases. Sub-Saharan Africa could reduce emissions from fertilizer production by one third and fertilizer application by ~80% under a circular economy in 2022, although this was predicted to diminish in 2030 and 2050 (Fig. 3b). Australia & New Zealand would see a negligible reduction in fertilizer use and application emissions under a circular economy in all studied years, and Oceania would make significant reduction in fertilizer production emissions under a circular economy in 2022, but this capacity was predicted to diminish in 2030 and 2050 (Fig. 3b).

Excreted C was a small percentage (<5%) of that which could potentially be sequestered in the cropland soil of Australia & New Zealand, and North America & Europe (Fig. 2), with little change from 2022 to 2050. Excreted C in Eastern & South-Eastern Asia and Central & Southern Asia was a larger percentage of that which could be sequestered at 2022 levels (29% and 18% respectively), and this percentage was shown to increase from 2022 to 2050. A similar trend was seen in Latin America & The Caribbean, and Western Asia & North Africa, although the overall percentage was lower compared to the 2022 baseline (12.4% and 9.4% respectively). This relatively large capacity for soil C sequestration offers substantial opportunity for long-term improvement in soil fertility and related benefits that are offered by reversing the historical decline of soil C stocks in agricultural regions.[7]

GHG emissions from sanitation services were most significant in developing SDG regions. A trend of increasing emissions from 2022 to 2050 in urban areas of developing regions was observed, whilst the opposite was true in rural areas (Fig. 4a and b). Emissions from sewerage dominate in wealthier countries and in some highly urbanised regions of LMICs, whereas in rural areas and poor cities, non-sewered sanitation and uncontrolled discharge of untreated excreta dominate. Open defecation was a small source of CH_4 in urban areas, but contributed a significant share in rural areas, particularly in Central & Southern Asia and Sub-Saharan Africa. N_2O emissions from all services were generally lower than CH_4 for all regions and time periods considered. Rapidly urbanizing regions such as Sub-Saharan Africa and Central & Southern Asia were predicted to significantly increase urban emissions between 2022 and 2050; increasing from 35.4 to 87.8 Tg CO_2 e per year in Sub-Saharan Africa and from 52.3 to 87.9 Tg CO_2 e per year in Central & Southern Asia (Fig. 4a).

Agriculture accounts for 70% of global water withdrawal, but variation is broad among SDG regions (Table 1). Agricultural

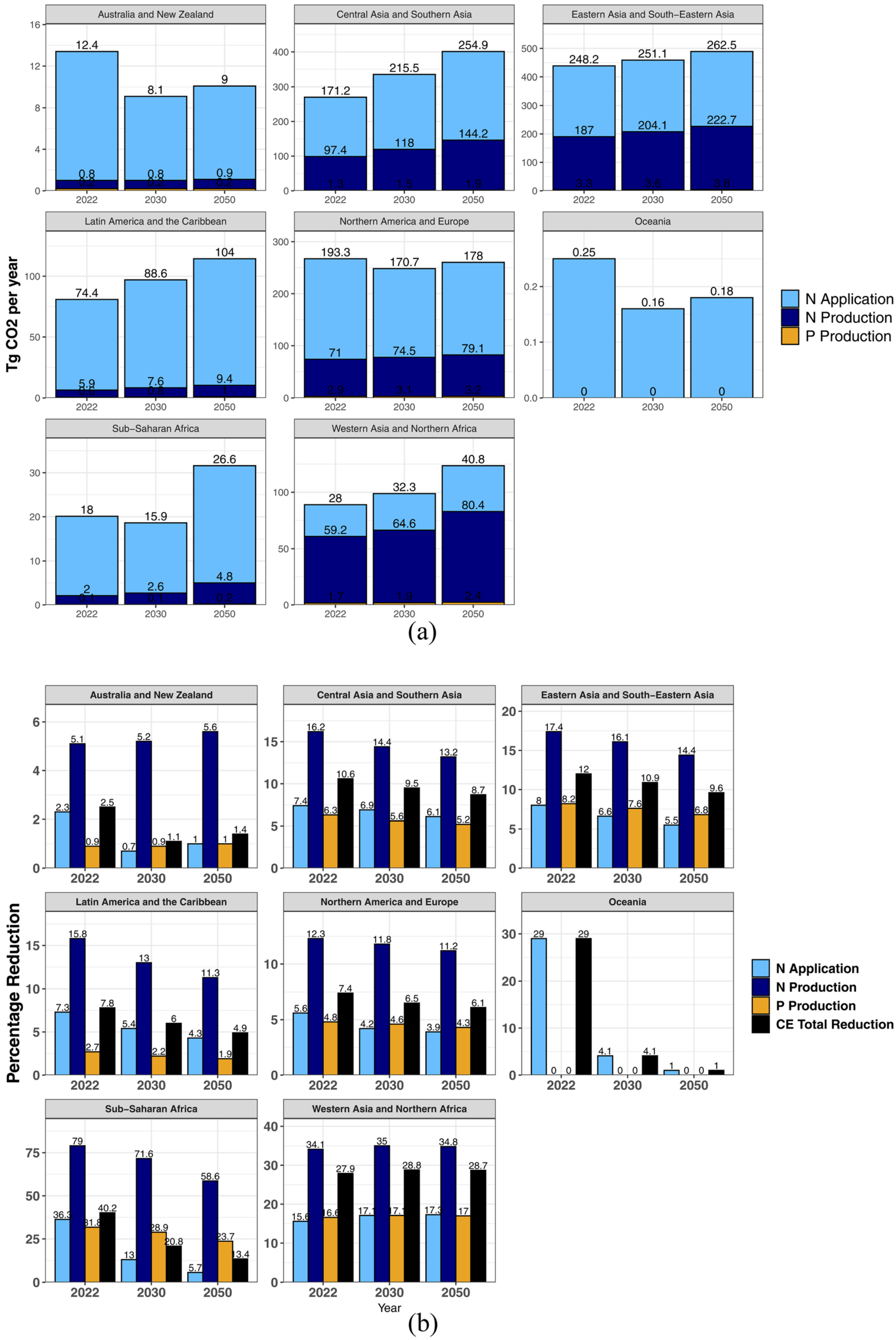
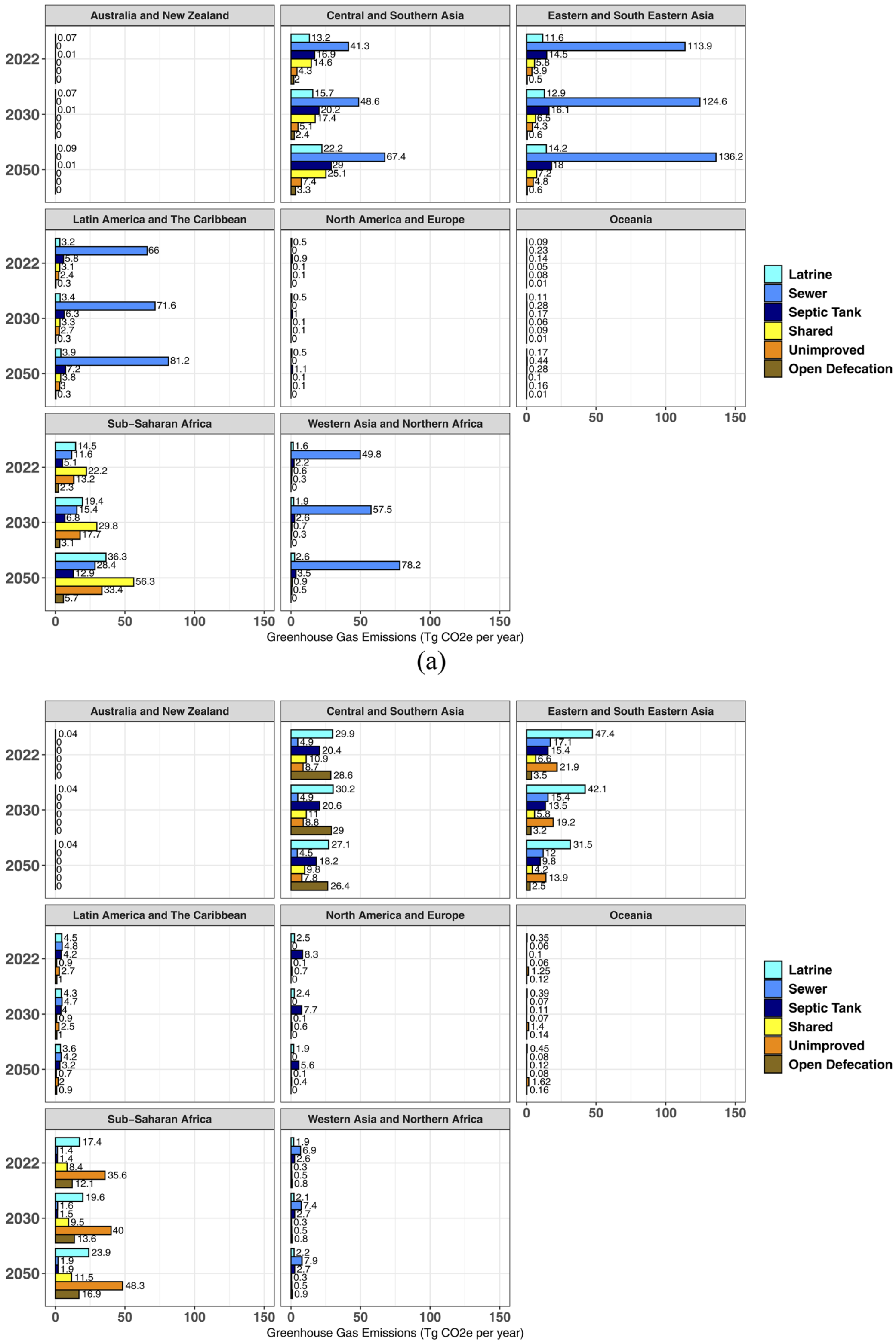


Fig. 3 (a) Greenhouse gas emissions from mineral N synthesis, phosphate production, and N fertilizer application (NO_2 expressed as CO_2 e) in Sustainable Development Goal Regions in 2022 with projects to 2030 and 2050. All units are Tg CO_2 e per year. (b) Percentage reduction of greenhouse gases from fertilizer production and application offered by a fully functional circular economy of excreta N and P in Sustainable Development Goals in 2022 with projections to 2030 and 2050.



(b)

Fig. 4 (a) Greenhouse gas emissions from sanitation services in urban areas in Sustainable Development Goal Regions in 2022 with projections to 2030 and 2050. Improved services are given in blue, unimproved services are given in yellow/tan. All units are Tg CO_2 e year per year. (b) Greenhouse gas emissions from sanitation services in urban areas in Sustainable Development Goal Regions in 2022 with projections to 2030 and 2050. Improved services are given in blue, unimproved services are given in yellow/tan. All units are Tg CO_2 e year per year.

Table 1 Total water resources (renewable and internal) with total freshwater and agricultural withdrawal in Sustainable Development Goal Regions in 2022, with projections to 2030 and 2050. All units are 10^9 m^3 (1 trillion litres)

Region	Year	Total renewable water resources	Total internal water resources	Total freshwater withdrawal	Total agricultural withdrawal
Australia & New Zealand	2022	819.0	819.0	25.8	13.7
	2030			28.1	14.9
	2050			32.4	17.2
Central & Southern Asia	2022	4155.7	2305.2	1146.7	1132.2
	2030			1253.2	1237.4
	2050			1404.9	1387.1
Eastern & South-Eastern Asia	2022	9928.4	8402.3	1255.5	927.2
	2030			1292.1	954.2
	2050			1284.0	948.2
Latin America & The Caribbean	2022	19 203.3	13 777.0	332.4	238.8
	2030			355.8	255.6
	2050			384.1	275.9
Northern America & Europe	2022	13 770.5	11 689.0	782.0	275.5
	2030			791.2	278.7
	2050			793.8	279.6
Western Asia & Northern Africa	2022	487.4	405.7	238.3	213.6
	2030			276.1	247.6
	2050			359.3	322.1
Oceania	2022	884.3	884.3	0.5	0.1
	2030			0.6	0.1
	2050			0.9	0.1
Sub-Saharan Africa	2022	5488.6	3871.4	100	74.3
	2030			113.9	84.6
	2050			141.1	104.8

withdrawal accounted for almost all freshwater withdrawal in Western Asia & Northern Africa and Central & Southern Asia, but only for a tiny fraction in Oceania. Sub-Saharan Africa and Latin America & The Caribbean used the bulk of their withdrawal for agriculture, but freshwater withdrawal was a small fraction of total renewable water resources. Renewable internal resources were much higher than that which is withdrawn in North America & Europe and Sub-Saharan Africa. The projections for 2030 and 2050 indicated the drier regions of Western Asia & Northern Africa and Central & Southern Asia will withdraw significantly increasing fractions of internal renewable water resources for agriculture.

Discussion and evidence for policy development

Excretion and agricultural application of N and P

Closing nutrient cycles between agriculture and diet is crucial for food security and environmental stewardship, as mineral fertilizers are produced using non-renewable resources, and dumping of waste N and P can cause environmental damage such as the eutrophication of waterways.[26] Cumulative heavy metal contamination and soil acidification are also problems associated with overuse of mineral N fertilizer.[9] The use of sanitation systems to recover excreta-derived N and P to supplement or replace agricultural mineral fertilizer is much discussed in the sustainability literature,[1,27–32] but if the N and P produced in excreta by humans is significantly lower than crop requirements, then sanitation systems cannot deliver the nutrient requirements within the region. Here the comparison of N and P excretion rates in SGD regions, with corresponding rates of agricultural application, places a question mark over the capacity of the circular economy of excreta-derived nutrients to supplement mineral fertilizer application rates in every region apart from Sub-Saharan Africa, and to a lesser extent Western Asia & Northern Africa. This was also found to be the case by Trimmer *et al.*[2,33] although these results confirm Oceania could also significantly supplement mineral fertilizer use with recovered nutrients from sanitation. Livestock manure N and P also plays a significant role in cropping systems (a total of 129 Tg N per year is generated but only 27.8 Tg N per year is applied to cropland). Full recovery of N from sanitation services was calculated here to provide a similar proportion to this in 2022 (30 Tg N), rising to 32 Tg N by 2050. Livestock P and C is not provided by FAOSTAT, but it is likely that it would provide a similar proportion.

Excretion and agricultural application rates of N and P show the most synchrony in Sub-Saharan Africa and Oceania because smallholder subsistence farming with localized consumption of produced food dominates these regions, and fertilizer use is restricted by economic conditions and an absence of government subsidies.[34] This also explains why application rates of N and P are below that which is required by 17 key crops in these regions (see values given by IFA[21] as described in SM2). Sub-Saharan Africa also still imports 20% of its cereals,[35] which increases the volumes of N and P in excreta beyond that which is contained in food produced in the region. Oceania remains food self-sufficient and N and P in excreta is not bolstered by food imports.[36] High prices, lack of adequate transport infrastructure and restricted market access also limit fertilizer use in these areas.[37] Fertilizer application rates are typically much higher in regions where fertilizer access is easier, such as North America & Europe and Eastern & South-Eastern Asia. North America & Europe also export significant volumes of food, as do some parts of Latin America & The Caribbean such as Brazil and Argentina. Food exports increase the demand for fertilizer beyond that required by diet domestically. Resource recovery from wastewater may also yield N and P beyond that which is excreted, as wastewater typically contains N and P from sources other than human waste, for example food processing and industrial waste,[38] and this may close the gap to an extent. But given the disparity between excretion and application rates of N and P shown in Fig. 1a and b, this addition is unlikely to make these recovered nutrients a viable alternative to mineral fertilizers in these regions.

The potential of excreta nutrients to supplement mineral fertilizers may be high in some regions and low in others, but using regional fertilizer application rates to assess the value of human-derived N and P may be unwise,[39] because applications rates often exceed or fall short of crop requirement (Fig. 1a and b) causing environmental emissions from excess use in some areas whilst limiting yields in others. It is not that soils in Malawi are rich in mineral nutrition,[40] and Chinese soils are not intrinsically devoid of essential nutrients,[41] yet average application rates of N fertilizer were 43 kg ha^{-1} per year in Malawi in 2013[42] and 503 kg ha^{-1} per year in China in 2010.[43] Key crop requirement calculations in this study indicate these application rates were insufficient and excessive, respectively. Crop requirements often have limited bearing on fertilizer application rates, as the political economy of production and subsidization often dominates farmer's decisions, as application rates in this study indicate. Excessive application of fertilizer, *i.e.* at rates beyond that which the crop is likely to take up, also has many deleterious environmental effects, and likewise insufficient application rates perpetuate poor yields. National fertilizer policy, where it exists, tends to safeguard national food security by ensuring the availability and affordability of fertilizers, but in doing this it can contribute to the environmental problems associated with excessive use.

The capacity of recovered N and P to supplement mineral fertilizers may change if national fertilizer policy were amended to reduce or increase application rates to that required by the crop. Such policies would likely increase fertilizer use significantly in places like Sub-Saharan Africa and diminish the capacity of recovered excreta nutrients to bolster mineral fertilizer application rates, in 2022 and even more so in future projections. In the developed regions of North America & Europe and Australia & New Zealand, full recovery of N and P from sanitation systems would make a negligible contribution to current and projected rates of mineral fertilizer application, but would make a significant contribution if these regions were to better align fertilizer use with crop nutritional requirement. If resource recovery is to supplement mineral fertilizer use in a meaningful way it must also be paired with improved agricultural, post-harvest and value chain practices that conserve N and P from farm to fork in all SDG regions.

The use of excreted C for soil organic carbon sequestration

C sequestration in agricultural soils is seen as a key component of the bioenergy with carbon capture and storage (BECCS) pathway for climate mitigation.[44] The idea that even a small annual increase in soil organic carbon (SOC) in agricultural soils could offset significant volumes of annual GHG emissions has garnered much interest, *e.g.* as the '4 per mil' initiative.[45] Increasing SOC in agricultural soil is also associated with improvements in soil fertility and other functionality due to residual N and P in the added organic matter, which may reduce the necessity for fertilizer applications, further contributing to climate mitigation.

The results here show excreted C can make a significant contribution to what could potentially be sequestered in agricultural soil in some SDG regions according to Zomer *et al.*[20] notably the Asian regions (Fig. 2). Human excreta-derived fertilizers in Eastern & South-Eastern Asia could provide ~29% of the C which could be sequestered in agricultural soil in 2022, but it remains unclear how much of this C is recoverable. WWTPs generally have treatment processes that generate CO_2, limiting the volumes of C available for recovery, but as the Asian regions had minimal rates of wastewater treatment in 2015 (Central & Southern Asia treated 3% of wastewater and Eastern & South-Eastern Asia treated 32% (ref. 46)), much of the untreated C remains available for recovery. The climate mitigation offered by annual additions of organic C to agricultural soil may however also be overstated, as most soils have a maximum C content which could be reached, and high rates of addition may alter the soil nitrogen cycle, causing soils to emit N_2O and cancel out the reduction in warming achieved by sequestering C.[47] A significant proportion of recovered C may also be lost from soil *via* respiration or erosion when added to agricultural land,[48] but if crop yields are increased by wastewater-derived fertilizers then this decrease may be offset by the increase in SOC associated with increased growth of root and microbiome biomass.

Excreted C in the industrialized regions (Australia & New Zealand and North America & Europe) was shown to be negligible when compared to that which could potentially be sequestered in agricultural soil. WWTPs in these areas are developing new technologies to recover C from organic materials, and although this C may contribute to soil fertility if returned to the regions agricultural soil, these results suggest the value of recovery in these regions lies more in reduction of nutrient emissions to water bodies.

GHG emissions from mineral fertilizer production and application

GHG emissions from the production of P fertilizer were significantly lower than that of N fertilizer production in all regions, for all time periods assessed (Fig. 3a), because the Haber–Bosch process requires much more energy than phosphate mining. This may change by 2050, as phosphate ore is predicted to become less pure and harder to access,[49] which will increase the energy requirements associated with mining and production. This may further incentivize P recovery from waste streams, but given how crops require much more N than P, and that P fertilizers do not have a GHG emission from their application, the recovery of N to mitigate GHG emissions from N production will likely take priority in all SDG regions, because the GHGs from N production and application are much more significant.

Regional differences in GHG emissions from N fertilizer production (CO_2) and application (N_2O expressed as CO_2 e) in 2022 and future projections stem from different practices in production and application. The Haber–Bosch process in the Asian regions produces higher masses of GHGs because more carbon-intensive fuels are used (Fig. 3b), thus increasing the GHG equivalent per kg N produced (for example, Western Europe emits 2.85 kg CO_2 kg per kg N produced, whereas China emits 6.36).[22,43] The use of natural gas as a fuel in North America & Europe reduces production emissions despite similar production amounts as the Asian regions in 2013 (34 Mt N per year in North America & Europe and 41 Mt N per year in Eastern & South Eastern Asia[17]). Regions reliant on coal for N production could reduce emissions significantly by switching to less carbon-intensive fuels or modern electrochemical methods (the 'green' ammonia project)[50] to power the reaction. Cleaner fuels may become more accessible in Eastern & South-Eastern Asia as China begins subsidizing domestic natural gas production.[51] The development of renewable energy production and improved resource recovery from waste streams could be a promising approach to sustainable fertilizer use in Eastern & South Eastern Asia. If this were to occur concomitant with fertilizer guidelines to encourage more precise fertilizer application rates, it may amount to a more sustainable fertilizer policy for the region.

Reducing GHG emissions by limiting the need for the Haber–Bosch process is a longstanding sustainable development target,[52,53] but N_2O emissions from fertilizer application are also significant, and in many SDG regions higher than the equivalent CO_2 emissions from fertilizer production (The Americas, Europe, Sub-Saharan Africa, and Oceania – Fig. 3a). Other research documenting fertilizer GHG emissions have also shown how N_2O emissions (expressed as CO_2 e) from application tend to be more significant than CO_2 emissions from production[54] when synthesis is energy efficient. Studies documenting GHG emissions from global fertilizer production specifically are also in line with the results shown for 2022 in Fig. S3† (423.1 Tg CO_2 per year), for example Liu *et al.*[55] calculated global emissions from nitrogen fertilizer production to be 420 Tg CO_2 in 2022 and Boerner[56] calculated 450 Tg CO_2 in 2019.

Global N_2O emissions from mineral fertilizer application are predicted to significantly increase by 2030 based on the data given in Fig. S3† and other studies,[57] and this raises questions about the use of excreta-derived fertilizers and their capacity to mitigate climate change. Processing of excreta-derived N and P can be minimal or extensive, generating potential fertilizers ranging from nearly raw sewage to highly processed forms; *e.g.* phosphate minerals such as struvite. Widely variable water content and thus unit mass will also influence the energy required to transport these organic fertilizers,[1] which effects their carbon footprint and their N_2O emission factor.

This study used a standard emission factor of 1.16% for excreta-derived fertilizers applied to agricultural land as biosolids (calculated in a global meta-analysis by Charles *et al.*[23]). Using this we found most SDG regions could reduce fertilizer N_2O emission by recovering N and reapplying it to agriculture, but this was found to be negligible in Australia and New Zealand (reductions were <2% in all cases). This was likely caused by low national fertilizer production emissions combined with the relatively higher emissions associated with land application of recovered N. This marks a trade-off, as the capacity of human-derived fertilizers to mitigate climate change may be compromised if their associated N_2O emission factor is higher than the mineral fertilizer they displace. Sub-Saharan Africa was shown to have the capacity to reduce mineral fertilizer production emissions by ∼80% using the circular economy at 2022 levels, although this capacity was reduced in the future projections. This was caused by the capacity for elimination of mineral fertilizer production emissions, as rates of recoverable N are higher than rates of mineral N agricultural application in this region. Novel products such as composted and pelleted faecal sludge developed in Sub-Saharan Africa[58] may further lower fertilizer emissions, as their expected N_2O emissions factor is lower.[23] Scaling up use of these more processed forms could produce viable organic fertilizers with minimal climate impact.[7] Design of fertilizer products for a low emission factor will be less critical for dry regions with lower risk of N_2O emissions from fertilizer use, such as Western Asia & North Africa.

GHG emissions from sanitation services

Improving nutrient recovery from sanitation services can mitigate climate change by providing organic C for sequestration in cropland soil and by limiting the release of methane from otherwise unrecovered organic C in sanitation services. Elements of SDG 6 (Clean Water and Sanitation) such as ending open defecation and halving the proportion of untreated wastewater may then have synergies with SDGs 2 and 13 if the effect of recovering excreta nutrients for agriculture is considered. Investment in sanitation services is generally motivated by public health concerns, and GHG emissions from sanitation services are not strongly prioritized in any part of SDG 6. Many studies document GHG emissions from WWTPs,[59–61] but these overwhelmingly describe emissions from the treatment process itself. This can be calculated at any given WWTP, but heterogeneity in treatment practices and environmental conditions make it difficult to scale up with any certainty estimates of the potential emissions.

Fig. 4a, b and S4† show a global estimate of GHG emissions (methane and nitrous oxide expressed as CO_2 e) from urban and rural sanitation services, *i.e.* emissions before reaching a WWTP, or rivers and oceans as is the case in many SDG regions. The contrast between developed regions (North America & Europe and Australia & New Zealand) and the rest of the world is caused by the dominant sewer network in the industrialized regions, which is typically sealed and does not emit significant GHGs. For developing SDG regions, the main sources of emissions are from excreta that are not contained in the system – predominantly leakage from sewers, and dumping from onsite systems, coupled with methane and nitrous emissions from within onsite systems. The extent of this is unknown, but recent work shows that a significant proportion of urban excreta end up in anaerobic conditions in poorly managed pits or open drains,[62] and in poor quality pits or open defecation in rural areas. Urban emissions from sewers in developing SDG regions are mostly CH_4 from unsealed networks,[63] and the extent of this problem is significant, but not precisely known. For example, despite significant government efforts to upgrade urban residences from latrines to a sewer network, one third of Chinese sewers are estimated to be in a state of disrepair.[64]

Latrines remain an important decentralized sanitation service for ~20% of the global urban population,[46] predominantly in Sub-Saharan Africa and Central & Southern Asia, as Fig. 4a suggests (we assume that the shared category in the JMP classification[46] refers to areas where latrines are most common). Latrines are even more widely used in rural areas and present a valuable opportunity for resource recovery, as contamination with industrial pollution and dilution by water is likely to be limited, but they may also emit significant volumes of CH_4 when anaerobic. Reid *et al.*[6] calculated CH_4 emissions from latrines in 21 countries using similar methodology to this study and found comparable regional results, although global estimations were considerably less as this study only surveyed 21 countries and did not include nitrous oxide. Global trends for latrine emissions found in this study are that rural areas will see a decline in many regions (see the Asian regions or Latin America & the Caribbean), whilst urban areas, in regions where latrines are common, will see a significant rise (Central & Southern Asia and Sub-Saharan Africa). If we assume the shared sanitation service is mostly composed of latrines in Sub-Saharan Africa, then emissions from latrines and shared services are predicted to rise from 18.4 Tg CO_2 e per year in 2022 to 46.0. Tg CO_2 e per year in 2050 (Fig. 4a) – roughly equivalent to the total national emissions of entire countries such as Syria (46.2 Tg CO_2 e per year), Denmark (46.7 Tg CO_2 e per year) or Bahrain (49.0 Tg CO_2 e per year) in 2018.[65]

These results indicate GHG emissions from Sub-Saharan sanitation services may contribute a more significant volume to total regional emissions. Field assessments in Kampala, Uganda have also suggested latrine emissions are particularly significant, and may account for up to half the city's total emissions.[66] Calculated emissions from latrines may even underestimate the true value, as they do not account for latrine leakage to an open drainage network, which may have a higher emission factor. The short-lived nature of methane in the atmosphere also implies that even a small curtailment of this rate may significantly reduce its warming effect.[67] C recovery in most regions may not provide significant volumes for C sequestration in agricultural soil (Fig. 2) but preventing this C from becoming CH_4 in latrines may have a significant climate benefit for regions with high emission rates from decentralized and poorly managed sanitation services.

GHG emissions from open defecation in rural areas were significant in some developing rural SDG regions in 2022, but this was projected to decline in 2030 and again in 2050 as these regions undergo urbanization. Achievement of SDG 6.2 (end open defecation) would eliminate these emissions, but if people switch from open defecation to latrines or septic tanks, which will likely be the case in Sub-Saharan Africa and Central & Southern Asia, the overall sanitation emissions may increase, as Shaw *et al.* have modelled,[68] due to the higher emission factors of latrines and septic tanks. This trade-off may be essential to achieve SDG 6 and end open defecation, but it can be mitigated by installing functional latrine services, *i.e.* dug in areas unlikely to flood and emptied regularly for resource recovery.[69]

Global sanitation service GHG emissions are predicted in this study to increase from 443 Tg CO_2 e per year in 2022 to 562 Tg CO_2 e per year in 2050 (Fig. S4†) – a figure comparable to the total annual emissions of some of the world's middle economies in 2018, *e. g.* Turkey (474 Tg CO_2 e per year), South Africa (520 Tg CO_2 e per year) and Australia (619 Tg CO_2 e per year).[70] This increase mostly stems from poorly constructed sewers in urban areas of the Global South, although latrines, septic tanks and basic sanitation services also make significant contributions. Investment in sanitation service improvement could therefore facilitate the achievement of SDGs 6 and 13 if development focused on resource recovery and minimal water usage, with the value of this investment best realized in Sub-Saharan Africa and Central & Southern Asia. Water usage is an important factor as household water demand for drinking, cooking and washing is substantial regardless of the sanitation system. Reducing overall water use and discharge to sanitation systems will require substantial household conservation measures with recycling and re-use of grey water.

Water availability and requirement

Recovery of nutrient C, N, and P for a circular economy also supports the achievement of SDG 2 Zero Hunger. This is through improved supply of recovered excreta nutrients to support food production in the face of a need to decrease mineral fertilizer use in order to achieve SDG 13 Climate Action. Agriculture and sanitation require water to function, and climate change is predicted to adversely impact the supply for both uses.[69,71] Drought and flooding can curtail crop production, drought can limit flow in sewers, and flooding can cause latrines and septic tanks to overflow, creating significant public health problems. Sanitation can also provide water for agriculture in certain contexts, as well as crop nutrition.[72,73] Achieving SDGs 2 (Zero Hunger) and 6 (Clean Water and Sanitation) requires increases in crop production and development of sanitation services, but this may create competition for water resources, which are limited in some regions (Table 1).

Our results indicate Western Asia & Northern Africa and Central & Southern Asia face the greatest pressure on currently available resources to deliver water for both goals, now and in projections to 2050. Sewers are accessed by 56% and 11% of the population in these regions respectively. If significantly increasing sewer access is at all infrastructurally and financially viable, the required water may make this development an impossibility, as agricultural usage amounts to such a significant percentage of the freshwater which is withdrawn. Expanding sewers further without improving the current network may also increase GHG emissions, which would be significant, as unsealed sewers in these regions already contribute a significant fraction of the total emissions from sanitation services (Fig. 4a). Trade-offs in resource allocation for sewer development may also arise in Eastern & South-Eastern Asia, particularly in China, who began a large program of urban sewer development in 2014.[74] Here agricultural withdrawal also takes the bulk of freshwater withdrawal, meaning improving these networks may demand significantly increasing the volume of water withdrawn for municipal use.

Regions such as Latin America & The Caribbean and Sub-Saharan Africa were seen to have undeveloped renewable water resources (*i.e.* renewable internal resources were far higher than water withdrawn). Other researchers have described how these regions could increase crop irrigation if water resources could be further developed,[75] and newly available water could also be useful for sewer development. A closed sewer network would significantly curtail GHG emissions from the currently malfunctional networks found in urban areas in these regions (Fig. 4a) but may render excreta nutrients more difficult to recover than from a latrine or septic tank. Individual countries and cities within each region may tailor sanitation investment programs to minimize trade-offs in water availability and climate mitigation based on specific environment and circumstance.

Conclusions

The 19th century sanitation reformer Edwin Chadwick foresaw the capacity of the circular economy to benefit humanity when he imagined a fully-functional sanitation system would create 'a post-Malthusian world where every being generated the fertilizer to sustain his own existence'.[76] Here we show this claim may not hold in SDG regions in which fertilizer application rates are very high, but different SDG regions can benefit from a functional circular economy of excreta nutrients in different ways, and there are contrasting trade-offs and co-benefits within these. The developed regions of North America, Europe, Australia and New Zealand would not replace or even meaningfully supplement current or projected rates of fertilizer application by maximum recovery of N and P from human waste, although if application rates were reduced to match actual crop requirements, then the disparity between mineral fertilizer application and potentially recoverable human-derived fertilizer may be reduced. Replacing mineral N and P fertilizer with human-derived fertilizers in these regions would not significantly reduce fertilizer emissions, it may even increase emissions in scenarios of low recovery combined with anticipated higher emission factors for human-derived fertilizers, as is the case in Australia and New Zealand. Recoverable C in these regions is only a small proportion of that which could be sequestered in cropland, but it offers an opportunity for long-term improvement in soil C stocks. GHG emissions from sanitation services here are not globally significant now or in future projections due to the dominance of the closed sewer network.

LMICs offer greater opportunities for the circular economy of human-derived nutrients to mitigate climate change. Regions such as Sub-Saharan Africa and Oceania could almost replace mineral fertilizer application rates with human derived nutrients in 2022 and in future projections, whilst Western Asia & North Africa could supplement to a significant degree, reducing GHG emissions from fertilizer production. Drier regions such as Western Asia & Northern Africa and Central & Southern Asia may face trade-offs in water allocation for both sanitation and agriculture, which could impact pathways to achieving the linked SDGs 2, 6 and 13, but this is unlikely in other regions. Although recovering sanitation waste in Latin America and the Asian regions would not achieve significant reductions in fertilizer GHG emissions, it would curb regional GHG emissions by preventing C and N from becoming CH_4 and NO_2 in each sanitation service. Recovered C may also provide further opportunities for C sequestration to cropland. The circular economy of human-derived nutrients has climate, food and water implications for every SDG region and policies focussing on different aspects of it can deliver some meaningful climate change mitigation in 2022 and in future projections to 2050.

Conflicts of interest

There are no conflicts of interest to declare.

Acknowledgements

The authors gratefully acknowledge funding for Patrick McKenna and Steve Banwart from the UKRI Newton Fund and Natural Environment Research Council, grants NE/N007514/1 and NE/S009124/1. The authors also wish to thank Franceco Tubiello at FAOSTAT for his support with using FAOSTAT.

References

1 J. T. Trimmer and J. S. Guest, Recirculation of human-derived nutrients from cities to agriculture across six continents, *Nat. Sustainability*, 2018, **1**, 427–435, DOI: 10.1038/s41893-018-0118-9.

2 J. T. Trimmer, R. D. Cusick and J. S. Guest, Amplifying Progress toward Multiple Development Goals through Resource Recovery from Sanitation, *Environ. Sci. Technol.*, 2017, **51**, 10765–10776, DOI: 10.1021/acs.est.7b02147.

3 J. T. Trimmer, D. C. Miller and J. S. Guest, Resource recovery from sanitation to enhance ecosystem services, *Nat. Sustainability*, 2019, **2**, 681–690, DOI: 10.1038/s41893-019-0313-3.

4 H. A. C. Lohman, *et al.*, Advancing Sustainable Sanitation and Agriculture through Investments in Human-Derived Nutrient Systems, *Environ. Sci. Technol.*, 2020, **54**(15), 9217–9227, DOI: 10.1021/acs.est.0c03764.

5 L. Lu, *et al.*, Wastewater treatment for carbon capture and utilization, *Nat. Sustainability*, 2018, **1**, 750–758, DOI: 10.1038/s41893-018-0187-9.
6 M. C. Reid, K. Guan, F. Wagner and D. L. Mauzerall, Global Methane Emissions from Pit Latrines, *Environ. Sci. Technol.*, 2014, **48**, 8727–8734, DOI: 10.1021/es501549h.
7 G. McNicol, J. Jeliazovski, J. J. François, S. Kramer and R. Ryals, Climate change mitigation potential in sanitation via off-site composting of human waste, *Nat. Clim. Change*, 2020, **10**, 545–549, DOI: 10.1038/s41558-020-0782-4.
8 C. Wang, B. Amon, K. Schulz and B. Mehdi, Factors That Influence Nitrous Oxide Emissions from Agricultural Soils as Well as Their Representation in Simulation Models: A Review, *Agronomy*, 2021, **11**, 770.
9 S. A. Banwart, N. P. Nikolaidis, Y.-G. Zhu, C. L. Peacock and D. L. Sparks, Soil Functions: Connecting Earth's Critical Zone, *Annu. Rev. Earth Planet. Sci.*, 2019, **47**, 333–359, DOI: 10.1146/annurev-earth-063016-020544.
10 A. E. Johnston, P. R. Poulton and J. McEwen, *Soils of Rothamsted Farm. The Carbon and Nitrogen Content of the Soils and the Effects of Changes in Crop Rotation and Manuring on Soil pH, P, K, and Mg*, Rothamsted Research, United Kingdom, 1980, vol. 5–20.
11 T. O. West, G. Marland, N. Singh, B. L. Bhaduri and A. B. Roddy, The human carbon budget: an estimate of the spatial distribution of metabolic carbon consumption and release in the United States, *Biogeochemistry*, 2009, **94**, 29–41, DOI: 10.1007/s10533-009-9306-z.
12 WWAP, *The United Nations World Water Development Report: Wastewater, the Untapped Resource*, UN, Paris, France, 2017.
13 IPCC. *Wastewater Treatment and Discharge*, Intergovernmental Panel on Climate Change, 2006.
14 M. Reid, Sanitation and climate, *Nat. Clim. Change*, 2020, **10**, 496–497, DOI: 10.1038/s41558-020-0787-z.
15 M. G. Henze, T. Mino and M. van Loosedrecht, *Biological Wastewater Treatment: Principles, Modelling and Design*, IWA Publishing, 2008.
16 Nations, U. *SDG Indicators: Regional Groupings Used in Report and Statistical Annex*, 2022, https://unstats.un.org/sdgs/indicators/regional-groups/.
17 Food and Agriculture Organization of the United Nations, *FAOSTAT Statistical Database*, FAO, Rome, 1997.
18 C. Rose, A. Parker, B. Jefferson and E. Cartmell, The Characterization of Feces and Urine: A Review of the Literature to Inform Advanced Treatment Technology, *Crit. Rev. Environ. Sci. Technol.*, 2015, **45**, 1827–1879, DOI: 10.1080/10643389.2014.1000761.
19 J. Gustavsson, C. Cederberg and U. Sonesson, *Global Food Losses and Food Waste. Extent, Causes and Prevention*, FAO, Rome, Italy, 2011.
20 R. J. Zomer, D. A. Bossio, R. Sommer and L. V. Verchot, Global Sequestration Potential of Increased Organic Carbon in Cropland Soils, *Sci. Rep.*, 2017, **7**, 15554, DOI: 10.1038/s41598-017-15794-8.
21 Association I. F. I., *World Fertilizer Use Manual*, BASF Agricultural Research Station, Limburgerhof, Germany, 1992.
22 A. Kool, M. Marinussen and H. Blonk, LCI data for the calculation tool Feedprint for greenhouse gas emissions of feed production and utilization. *GHG Emissions of N, P and K Fertilizer Production*, Blonk Consultants, The Netherlands, 2012.
23 A. Charles, *et al.*, Global nitrous oxide emission factors from agricultural soils after addition of organic amendments: A meta-analysis, *Agric. Ecosyst. Environ.*, 2017, **236**, 88–98, DOI: 10.1016/j.agee.2016.11.021.
24 IPCC, *Chapter 6 – Wastewater Treatment and Discharge*. 2019.
25 *AQUASTAT FAO's Information System on Water and Agriculture*, FAO, Rome, 1999.
26 X. Liu, *et al.*, Evidence for a Historic Change Occurring in China, *Environ. Sci. Technol.*, 2016, **50**, 505–506, DOI: 10.1021/acs.est.5b05972.
27 D. Cordell, J. O. Drangert and S. White, The story of phosphorus: Global food security and food for thought, *Global Environ. Change*, 2009, **19**, 292–305, DOI: 10.1016/j.gloenvcha.2008.10.009.
28 R. Nkoa, Agricultural benefits and environmental risks of soil fertilization with anaerobic digestates: a review, *Agron. Sustainable Dev.*, 2014, **34**, 473–492, DOI: 10.1007/s13593-013-0196-z.
29 J. R. Mihelcic, L. M. Fry and R. Shaw, Global potential of phosphorus recovery from human urine and feces, *Chemosphere*, 2011, **84**, 832–839, DOI: 10.1016/j.chemosphere.2011.02.046.
30 H. Heinonen-Tanski and C. van Wijk-Sijbesma, Human excreta for plant production, *Bioresour. Technol.*, 2005, **96**, 403–411, DOI: 10.1016/j.biortech.2003.10.036.
31 T. Karak and P. Bhattacharyya, Human urine as a source of alternative natural fertilizer in agriculture: A flight of fancy or an achievable reality, *Resour., Conserv. Recycl.*, 2011, **55**, 400–408, DOI: 10.1016/j.resconrec.2010.12.008.
32 P. M. Melia, A. B. Cundy, S. P. Sohi, P. S. Hooda and R. Busquets, Trends in the recovery of phosphorus in bioavailable forms from wastewater, *Chemosphere*, 2017, **186**, 381–395, DOI: 10.1016/j.chemosphere.2017.07.089.
33 D. Echevarria, J. T. Trimmer, R. D. Cusick and J. S. Guest, Defining Nutrient Colocation Typologies for Human-Derived Supply and Crop Demand To Advance Resource Recovery, *Environ. Sci. Technol.*, 2021, **55**, 10704–10713, DOI: 10.1021/acs.est.1c01389.
34 Z. Druilhe and J. Barreiro-Hurlé, *Fertilizer Subsidies in Sub-saharan Africa*, FAO, Rome, 2012.
35 M. K. van Ittersum, *et al.*, Can sub-Saharan Africa feed itself?, *Proc. Natl. Acad. Sci. U. S. A.*, 2016, **113**, 14964–14969, DOI: 10.1073/pnas.1610359113.
36 L. Lassaletta, *et al.*, Nitrogen use in the global food system: past trends and future trajectories of agronomic performance, pollution, trade, and dietary demand, *Environ. Res. Lett.*, 2016, **11**, 095007, DOI: 10.1088/1748-9326/11/9/095007.
37 D. I. Gregory and B. L. Bumb, *Factors Affecting Supply of Fertilizer in Sub-saharan Africa*, World Bank, 2006.
38 A. L. Moree, A. H. W. Beusen, A. F. Bouwman and W. J. Willems, Exploring global nitrogen and phosphorus

flows in urban wastes during the twentieth century, *Global Biogeochem. Cycles*, 2013, **27**, 836–846, DOI: **10.1002/gbc.20072**.
39 P. M. Vitousek, *et al.*, Nutrient Imbalances in Agricultural Development, *Science*, 2009, **324**, 1519–1520, DOI: **10.1126/science.1170261**.
40 S. Snapp, D. Rohrbach, F. Simtowe and H. Freeman, Sustainable soil management options for Malawi: can smallholder farmers grow more legumes?, *Agric. Ecosyst. Environ.*, 2002, **91**, 159–174.
41 H. Yang, Resource management, soil fertility and sustainable crop production: Experiences of China, *Agric. Ecosyst. Environ.*, 2006, **116**, 27–33.
42 Centre I. F. D, *Malawi Fertilizer Assessment*, 2013.
43 Y. X. Li, *et al.*, An Analysis of China's Fertilizer Policies: Impacts on the Industry, Food Security, and the Environment, *J. Environ. Qual.*, 2013, **42**, 972–981, DOI: **10.2134/jeq2012.0465**.
44 C. Gough and P. Upham, Biomass energy with carbon capture and storage (BECCS or Bio-CCS), *Greenhouse Gases: Sci. Technol.*, 2011, **1**, 324–334, DOI: **10.1002/ghg.34**.
45 J.-F. Soussana, *et al.*, Matching policy and science: Rationale for the ‘4 per 1000 – soils for food security and climate’ initiative, *Soil Tillage Res.*, 2019, **188**, 3–15, DOI: **10.1016/j.still.2017.12.002**.
46 JMP, *Progress on Drinking Water, Sanitation and Hygiene: 2017 Update and SDG Baselines*, Geneva, 2017.
47 B. Guenet, *et al.*, Can N_2O emissions offset the benefits from soil organic carbon storage?, *Global Change Biol.*, 2021, **27**(2), 237–256, DOI: **10.1111/gcb.15342**.
48 R. L. Ray, *et al.*, Soil CO_2 emission in response to organic amendments, temperature, and rainfall, *Sci. Rep.*, 2020, **10**, 14, DOI: **10.1038/s41598-020-62267-6**.
49 D. Cordell and S. White in *Annual Review of Environment and Resources*, ed. A. Gadgil and D. M. Liverman, 2014, vol. 39, p. 161.
50 M. Capdevila-Cortada, Electrifying the Haber–Bosch, *Nat. Catal.*, 2019, **2**, 1055, DOI: **10.1038/s41929-019-0414-4**.
51 F. Z. V. Aloulou, in *World Oil* (2019).
52 B. J. Gu, *et al.*, The role of industrial nitrogen in the global nitrogen biogeochemical cycle, *Sci. Rep.*, 2013, **3**, 7, DOI: **10.1038/srep02579**.
53 J. W. Erisman, *et al.*, Consequences of human modification of the global nitrogen cycle, *Philos. Trans. R. Soc., B*, 2013, **368**, 9, DOI: **10.1098/rstb.2013.0116**.
54 R. Chai, *et al.*, Greenhouse gas emissions from synthetic nitrogen manufacture and fertilization for main upland crops in China, *Carbon Balance Manage.*, 2019, **14**, 20, DOI: **10.1186/s13021-019-0133-9**.
55 X. Liu, A. Elgowainy and M. Wang, Life cycle energy use and greenhouse gas emissions of ammonia production from renewable resources and industrial by-products, *Green Chem.*, 2020, **22**, 5751–5761.
56 L. K. Boerner, Industrial ammonia production emits more CO2 than any other chemical-making reaction. Chemists want to change that, *Green Chem.*, 2019, **97**.
57 R. L. Thompson, *et al.*, Acceleration of global N2O emissions seen from two decades of atmospheric inversion, *Nat. Clim. Change*, 2019, **9**, 993, DOI: **10.1038/s41558-019-0613-7**.
58 S. K. Pradhan, J. Nikiema, O. O. Cofie, H. Heinonen-Tanski and P. Drechsel, Fecal sludge-derived pellet fertilizer in maize cultivation, *J. Water, Sanit. Hyg. Dev.*, 2016, **6**, 474–481, DOI: **10.2166/washdev.2016.160**.
59 M. J. Kampschreur, *et al.*, Dynamics of nitric oxide and nitrous oxide emission during full-scale reject water treatment, *Water Res.*, 2008, **42**, 812–826, DOI: **10.1016/j.watres.2007.08.022**.
60 M. R. J. Daelman, E. M. van Voorthuizen, U. van Dongen, E. I. P. Volcke and M. C. M. van Loosdrecht, Methane emission during municipal wastewater treatment, *Water Res.*, 2012, **46**, 3657–3670, DOI: **10.1016/j.watres.2012.04.024**.
61 M. R. J. Daelman, E. M. van Voorthuizen, L. van Dongen, E. I. P. Volcke and M. C. M. van Loosdrecht, Methane and nitrous oxide emissions from municipal wastewater treatment - results from a long-term study, *Water Sci. Technol.*, 2013, **67**, 2350–2355, DOI: **10.2166/wst.2013.109**.
62 A. Peal, *et al.*, Estimating Safely Managed Sanitation in Urban Areas; Lessons Learned From a Global Implementation of Excreta-Flow Diagrams, *Front. Environ. Sci.*, 2020, **8**, DOI: **10.3389/fenvs.2020.00001**.
63 T. Chaosakul, T. Koottatep and C. Polprasert, A model for methane production in sewers, *J. Environ. Sci. Health, Part A: Toxic/Hazard. Subst. Environ. Eng.*, 2014, **49**, 1316–1321, DOI: **10.1080/10934529.2014.910071**.
64 Z. X. Xu, *et al.*, Urban river pollution control in developing countries, *Nat. Sustainability*, 2019, **2**, 158–160, DOI: **10.1038/s41893-019-0249-7**.
65 *Our World in Data*, ed. Ritchie, H. R. M., 2021.
66 J. Johnson, *et al.*, Whole-system analysis reveals high greenhouse-gas emissions from citywide sanitation in Kampala, Uganda, *Commun. Earth Environ.*, 2022, **3**, 80, DOI: **10.1038/s43247-022-00413-w**.
67 J. Lynch, M. Cain, R. Pierrehumbert and M. Allen, Demonstrating GWP*: a means of reporting warming-equivalent emissions that captures the contrasting impacts of short- and long-lived climate pollutants, *Environ. Res. Lett.*, 2020, **15**, 044023, DOI: **10.1088/1748-9326/ab6d7e**.
68 K. Shaw, C. Kennedy and C. C. Dorea, Non-Sewered Sanitation Systems' Global Greenhouse Gas Emissions: Balancing Sustainable Development Goal Tradeoffs to End Open Defecation, *Sustainability*, 2021, **13**, 11884.
69 F. Mills, J. Willetts, B. Evans, N. Carrard and J. Kohlitz, Costs, Climate and Contamination: Three Drivers for Citywide Sanitation Investment Decisions, *Front. Environ. Sci.*, 2020, **8**, DOI: **10.3389/fenvs.2020.00130**.
70 *Our World in Data*, Ritchie, H. R. M., 2021.
71 K. Chartzoulakis and M. Bertaki, Efficient Irrigation Management and Its Effects in Urban and Rural Landscapes, in *Agriculture and Agricultural Science Procedia*, ed. P. E. Barouchas, Y. L. Tsirogiannis, and N. Malamos, Elsevier Science Bv, 2015, vol. 4, pp. 88–98.

72 M. C. Chrispim, W. A. Tarpeh, D. T. P. Salinas and M. A. Nolasco, The sanitation and urban agriculture nexus: urine collection and application as fertilizer in Sao Paulo, Brazil, *J. Water, Sanit. Hyg. Dev.*, 2017, **7**, 455–465, DOI: **10.2166/washdev.2017.163**.
73 L. Miller-Robbie, A. Ramaswami and P. Amerasinghe, Wastewater treatment and reuse in urban agriculture: exploring the food, energy, water, and health nexus in Hyderabad, India, *Environ. Res. Lett.*, 2017, **12**, 11, DOI: **10.1088/1748-9326/aa6bfe**.
74 D. Huang, *et al.*, Current state and future perspectives of sewer networks in urban China, *Front. Environ. Sci. Eng.*, 2018, **12**, 16, DOI: **10.1007/s11783-018-1023-1**.
75 J. Elliott, *et al.*, Constraints and potentials of future irrigation water availability on agricultural production under climate change, *Proc. Natl. Acad. Sci. U. S. A.*, 2014, **111**, 3239–3244, DOI: **10.1073/pnas.1222474110**.
76 C. Hamlin, *Public Health and Social Justice in the Age of Chadwick: Britain 1800-1850*, Cambridge University Press, 1998.