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Understanding the fate of nitrogen compounds in the urban environment

Final report of the UNCNET project (D1/5)

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Urban nitrogen cycles:
new economy thinking to master the challenges of climate change

D1/5 FINAL PROJECT REPORT

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PU	Public	☒
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CO	Confidential, only for members of the consortium (including funding agencies)	☐

Table of Contents

SUMMARY

The project UNCNET successfully established the pathways of reactive nitrogen compounds (Nr) for urban and peri-urban regions. Test areas were the urban and peri-urban regions of four cities in China and Europe (Vienna, Austria, Zielona Góra, Poland, and Beijing and Shijiazhuang, China). As Nr is known to be responsible for a series of negative environmental impacts, understanding its cycling is essential for integrative approaches to address issues like air pollution, water pollution, soil acidification, eutrophication, biodiversity impacts, or climate change.

Focussing on the urban situation allowed to identify and separate two distinct strata of flows, which were termed the “agro-food chain” and the “combustion chain”. The well-known nitrogen cascade, creating interferences widely across nitrogen flows of a country or even continent, appears to be less relevant, simplifying source attribution of observed exceedances. Moreover, urban densities (population, energy and food consumption, pollution) simplified the analysis compared to a country scale, while many underlying data (statistics) were much more difficult if not impossible to obtain.

While differences in the industrial make-up, energy strategies, and the structure of peri-urban agriculture do have repercussions on the fate of Nr, some key results turn out to remain beyond the challenge of data harmonization based on very different availability. First of all, in terms of Nr, large flows include (where applicable) mineral fertilizer application, human/household consumption, and fuel oil. Especially relevant, for an urban area, is the magnitude of the agro-food chain that strongly determines the overall pattern of Nr. Secondly, the most general practice to reduce damage caused by Nr is in its conversion to molecular (atmospheric) N_2, a process involving potentially substantial losses of energy, e.g. in the form of denitrification in wastewater treatment, or the its selective reduction (with or without catalyst) at the end of combustion processes. Finally, out of considerable potentials for recycling, only minor shares are effectively implemented. Especially organic wastes may be reused as nutrients after processing (e.g., composting). Where relevant, this is the case for animal manure, but also for human excrements, which are to a small extent applied on fields either directly or in the form of sewage sludge form wastewater treatment. Considering quantities, extending application of human manure clearly offers great opportunities due to the general importance of the agro-food chain (recognizing the need to deal with sanitary issues and with heavy metal/medical pollution).

Taking account of the needs and the intentions of stakeholders as expressed in dedicated meetings, recommendations to urban decision makers were formulated. These include the need for an integrative view on Nr, the necessity to learn from practices applied in other cities via benchmarking, and the target to fulfil the requirements of the UN Sustainability Goals via limiting the release of environmentally detrimental nitrogen compounds.

1. Introduction

The element nitrogen (N), in its chemical compounds, is known to be essential for life. As a constituent of proteins, N is required for all metabolic processes. Yet on earth N is a scarce resource. Virtually all N is available in molecular from, as N_2 gas, the main compound of the atmosphere. In order to become available to plants, it needs to be converted into “reactive nitrogen”, Nr. Nr will strongly enhance plant growth, hence it is added to soils as fertilizer and is indispensable for human nutrition. By way of adding fertilizers to soils, but also releasing nitrogen oxides resulting from combustion processes, humans have doubled the global cycle on Nr compounds. The sudden easy availability of Nr has led to multiple effects on ecosystems and on the environment in general, including climate change with N_2O as the third most important anthropogenic greenhouse gas, and N deposition (and availability) also relevant for C uptake in the biosphere (carbon sequestration).

Nr is an example for strongly interlinked processes. While the effects of changing Nr flows are often difficult to predict, due to such interlinkages and chemical processes that allow rapid conversion between different chemical forms, the overall amount of Nr remains remarkably stable. Mass conservation and a material balance approach offer adequate opportunities to describe the overall principles of the environmental fate of Nr. Together with intermittent information that may be available for specific fluxes, an overall framework of N budgets can be created. Multiple uses of Nr then also allow to assess the potentials of recycling (as well as their implementation), becoming a model example for a circular economy.

Investigation of Nr issues traditionally has focused on agricultural settings. As most of Nr is being applied in agriculture as fertilizer, the countryside is the area that receives the by far largest part of such input. However, many of the N flows focus on urban areas. Cities are where people live in high density, and where food is being consumed and metabolized in human bodies. At the same time, fuel combustion also is concentrated in cities, due to energy needs of industry and humans. Finally, cities are places where N waste is being created, providing opportunities for recovery. Hence, in the UNCNET project, Nr flows in cities have been investigated by means of N budgets.

This final report describes the key project results in line with the overall project structure. Firstly, the sectoral elements are being discussed – atmosphere and hydrosphere as the essential environmental transport media, and agriculture and the household sector including waste as the essential focal points of N flows. Science developed here directly is integrated to the flux scheme methodology of the urban nitrogen budget, and it forms the backbone of its application in the four test cities (Vienna, Zielona Góra, Beijing, Shijiazhuang). These results allow detailed interpretation of our understanding of urban nitrogen cycles. Interaction with stakeholders accompanied the development of the methodology throughout the project duration and the policy impacts are described. The key conclusions of UNCNET complete the report. Details are covered in the individual project deliverables as well as in the scientific publications available (or still to be added once public) at the project web page www.uncnet.org.

Following UNCNET’s specification, this report covers the main results of the respective sectoral approaches (atmosphere and hydrosphere/soil as important pools, agriculture and waste as key sectors: sections 2-5), before reporting on the stakeholder interaction process (section 6) and the main results derived from comparing results across cities (sections 7 and 8), in part together with stakeholders. Spatially, system boundaries were selected to facilitate data availability, that means administrative borders proved useful. Moreover, separating areas into an urban core of highest population density, and the larger, peri-urban area that is strongly influenced by the city allowed to work out differences. Fig. 1 displays the boundaries of urban core and peri-urban area used in each of the four cities serving as test area.

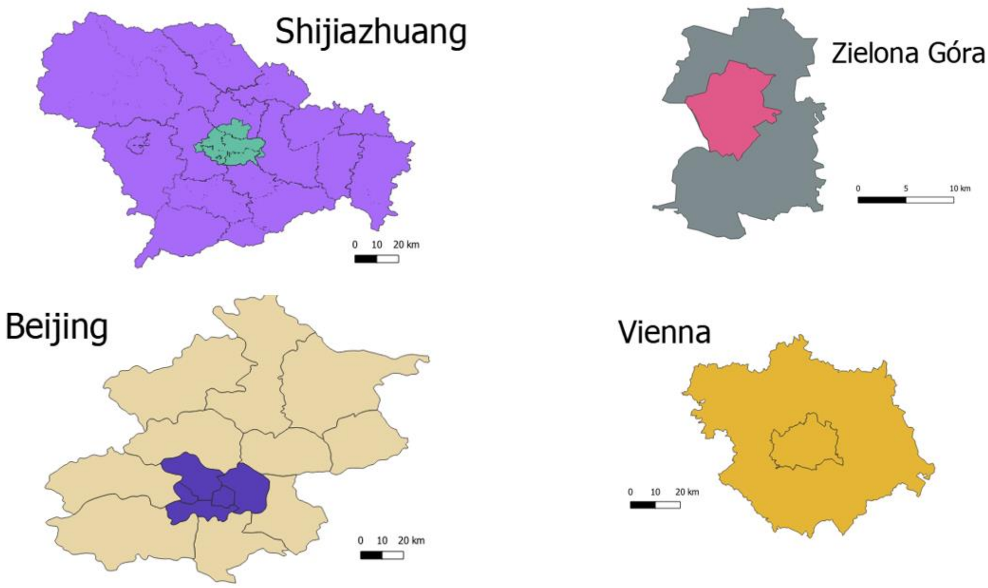

Figure 1. Spatial delineation of the four test areas into core area and peri-urban area

2. Nitrogen compounds in the urban atmosphere

2.1 Overview

$PM_{2.5}$ air pollution has been reported to cause millions of premature deaths annually in recent years, especially in densely populated urban areas. Among the major $PM_{2.5}$ precursors, NH_3 is very important for it can stabilize the nucleation and neutralize acidic aerosols (sulfate and nitrate) to form secondary inorganic particles, which are the main components of $PM_{2.5}$. As a result, NH_3 emission reduction plays an important role in reducing $PM_{2.5}$ concentration and improving air quality. However, NH_3 emissions have not been regulated yet in China and most European countries.

As the tasks of UNCNET WP3, we estimate the amount of ammonia (NH_3) emitted to the atmosphere from urban agricultural activities over China where NH_3 emissions inventory still exist highly uncertainty. We further investigate the impacts of NH_3 emissions on $PM_{2.5}$ air pollution over Beijing-Tianjin-Hebei (BTH) region and Vienna (Vienna having been 4th largest city globally about 150 years ago) using a regional air quality model (The Weather Research and Forecasting (WRF) coupled with Chemistry (WRF-Chem)). We conduct a series of model simulations to quantify the impacts of NH_3 emissions on urban $PM_{2.5}$ air quality and design practical scenarios for reducing agricultural ammonia emissions. Obtaining a quantitative estimate of the effectiveness of NH_3 emission reductions on $PM_{2.5}$ air pollution regulation will help policy makers to optimize emissions reduction strategies.

We find strong nonlinear responses of $PM_{2.5}$ air pollution to NH_3 emission reductions in North China and Vienna, with increasing effectiveness as deeper NH_3 emission reductions. The BTH mean PWC $PM_{2.5}$ concentrations in January would only decrease 1.4-3.8 $\mu g\ m^{-3}$ (1.1-2.9% of $PM_{2.5}$) when NH_3 emissions in North China were reduced by 20-40%, but the decreases would reach 8.1-26.7 $\mu g\ m^{-3}$ (6.2-20.5% of $PM_{2.5}$) with 60-100% NH_3 emission reductions. In Vienna, NH_3 emission controls become more effective than NO_x emission controls when the emission reduction percentage is higher than ~60%. Such nonlinearity reflects a switch of NH_3-saturated to NH_3-limited condition for SIA, in particular, aerosol nitrate formation. The $PM_{2.5}$ changes in July also show a nonlinear response, but the nonlinearity is much weaker than January. Combine the technologies of using proper feed, better manure management, optimize fertilizer application rates and adding urease inhibitors could reduce urban agriculture NH_3 emissions over North China by 56.3%

2.2 Objectives

Under the scope of the UNCNET project, we estimate the amount of ammonia releasing to the atmosphere from agricultural activities in the city core and extended areas over China. With the updated agricultural ammonia emission inventory, we will further conduct a series of model simulations to quantify the effectiveness of NH_3 emission reductions on $PM_{2.5}$ air pollution regulation. Finally, we will design a series of practical scenarios for reducing agricultural ammonia emissions from the application of chemical fertilizer and from different stages of livestock and poultry breeding.

2.3 Results

2.3.1 Abating ammonia is more cost-effective than nitrogen oxides for mitigating $PM_{2.5}$ air pollution

The metric called “N-share” of $PM_{2.5}$ pollution is developed to quantify the contribution of reactive nitrogen (Nr) compounds to total $PM_{2.5}$ concentration, which is determined by modeling with and without Nr emission. Here, we show that nitrogen accounted for 39% of global $PM_{2.5}$ exposure in 2013, increasing from 30% in 1990 with rising reactive nitrogen emissions and successful controls on sulfur dioxide. Controlling NH_3 tend to be more effective than NO_x for deep emission controls. The N-share caused by NH_3 emissions contributed an estimated 25% (range, 20 to 31%) to $PM_{2.5}$ pollution in 1990, increasing to 32% (25 to 39%) in 2013, whereas NO_x emission contributed 17% (14 to 20%) in 1990, increasing to 28% (23 to 33%) in 2013 (Fig. 2). Nitrogen emissions to air caused an estimated 23.3 million years of life lost in 2013, corresponding to an annual welfare loss of 420 billion United States dollars for premature death. The marginal abatement cost of ammonia emission is only 10% that of nitrogen oxides emission globally, highlighting the priority for ammonia reduction.

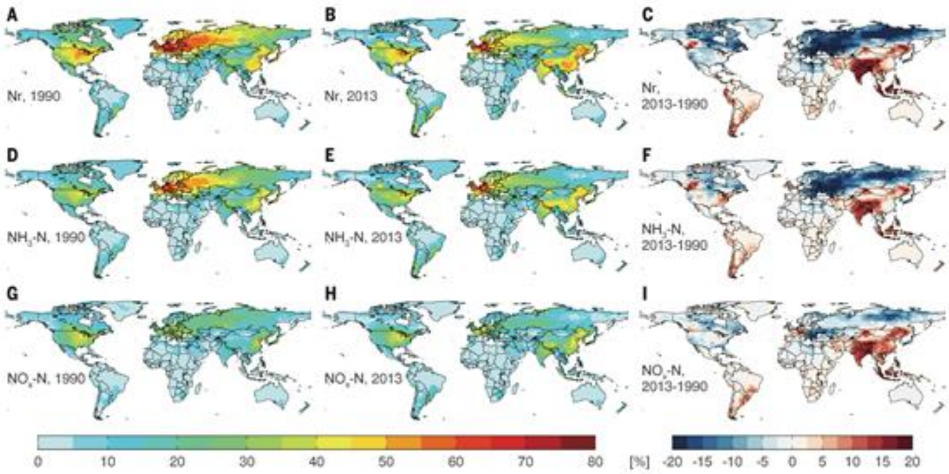

Figure 2. N-shares of $PM_{2.5}$ pollution and their changes between 1990 and 2013. Values represent the percentage contribution of each form of Nr emission to $PM_{2.5}$ pollution. (A) Total Nr-share in 1990. (B) Total Nr-share in 2013. (C) Change in total Nr-share between 2013 and 1990. (D) NH_3-N share in 1990. (E) NH_3-N share in 2013. (F) Change in NH_3-N share between 2013 and 1990. (G) NO_x-N share in 1990. (H) NO_x-N share in 2013. (I) Change in NO_x-N share between 2013 and 1990. (J) Change in NO_x-N share between 2013 and 1990.

2.3.2 Estimates of ammonia emissions from urban agriculture

We improve the bottom-up approach of Zhang et al. (2018) to develop a Chinese agricultural NH_3 emission inventory covering 2005-2016. The total anthropogenic emissions of NH_3 in China decreased from 14.4 Tg NH_3 yr^{-1} in 2005 to 13.1 Tg NH_3 yr^{-1} in 2007, and remained relatively stable in the range of 12.9 Tg - 13.9 Tg NH_3 yr^{-1} over 2008-2016 (Fig. 3). Total

Chinese anthropogenic NH_3 emissions from fertilizer are 5.0 Tg in 2005, then increase to 5.3 Tg in 2012, and further decrease to 4.5 Tg in 2016. As the fertilizer application amounts have increased and the ammonium bicarbonate (ABC) with a much higher emission factor than urea and compound fertilizer (NPK) has largely decreased during the past decade. NH_3 emissions from livestock reached 8.2 Tg in China in 2005 and decreased to 7.0 Tg in 2016. The trend could be largely explained by the switch of free-range raising to large intensive raising systems since 2007. We find that the recent increases in atmospheric NH_3 levels over North China as evident by satellite observations are not driven by the changes in NH_3 emissions, but to a large extent by the changes in SO_2 emissions over this region.

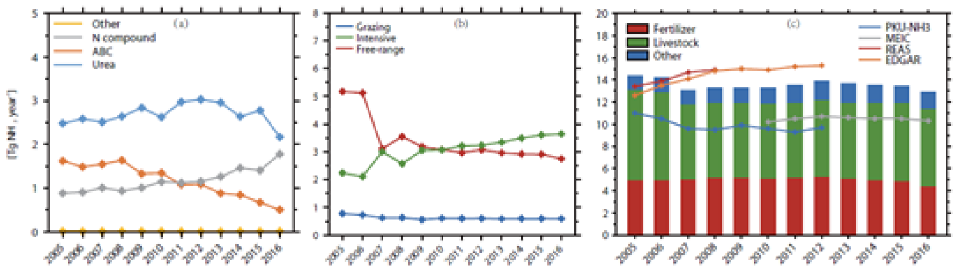

Figure 3. NH_3 emissions from (a) nitrogen fertilizer: urea, ABC, nitrogen compound, and other nitrogen fertilizers, (b) livestock: free-range, intensive, and grazing system, (c) NH_3 emissions from fertilizer, livestock, and other anthropogenic sources over 2005-2016 in China. The lines in (c) are emission inventories from other studies: PKU-NH_3, MEIC, REAS and EDGAR.

2.3.3 A quantitative estimate of the impacts of ammonia emissions on urban $PM_{2.5}$ air quality

We conduct a series of WRF-Chem simulations to investigate the impacts of ammonia emissions on urban $PM_{2.5}$ air quality over Beijing-Tianjin-Hebei (BTH) region and Vienna with 9 km and 3 km horizontal resolutions, respectively. The average $PM_{2.5}$ concentration over BTH regions (86.5/32.7 μg m^{-3} in January/July) is much higher than Vienna (13.8/7.1 μg m^{-3} in January/July) because of much higher air pollutant emissions over North China. The decreases of $PM_{2.5}$ concentrations associated with NH_3 emission reductions follow a power exponential function in January leading to small $PM_{2.5}$ changes under small NH_3 emission reductions. We find stronger $PM_{2.5}$ responses in heavy pollution episodes in both months. As shown in Fig. 4, for the highest 5% $PM_{2.5}$ concentrations, values over BTH can be decreased by 4.5/24.2/64.4 μg m^{-3} when NH_3 emissions in North China are reduced by 20%/60%/100% and values in Vienna can be decreased by 0.83/2.8/9.3 μg m^{-3} when NH_3 emissions in Europe are reduced by 30%/60%/100% in January.

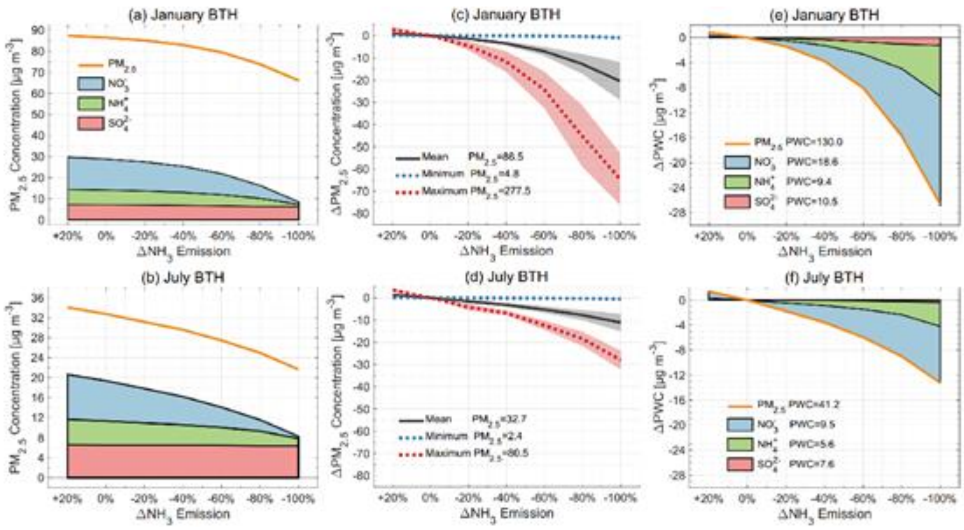

Figure 4. Effectiveness of NH_3 emission reductions in North China on BTH regional mean surface $PM_{2.5}$ pollution in January (top panels) and July (bottom panels) 2015. The left panels show BTH geometric mean $PM_{2.5}$ (orange lines), sulfate (red shading), ammonium (green shading), and nitrate (blue shading) levels. The central panels show reductions in monthly mean (black lines), minimum (blue dashed lines), and maximum (red dashed lines) $PM_{2.5}$ concentrations. The right panels show changes in population-weighted $PM_{2.5}$ (PWC) together with sulfate, ammonium, and nitrate contributions. Numbers inset are their values (ug m^{-3}) in the Base simulation.

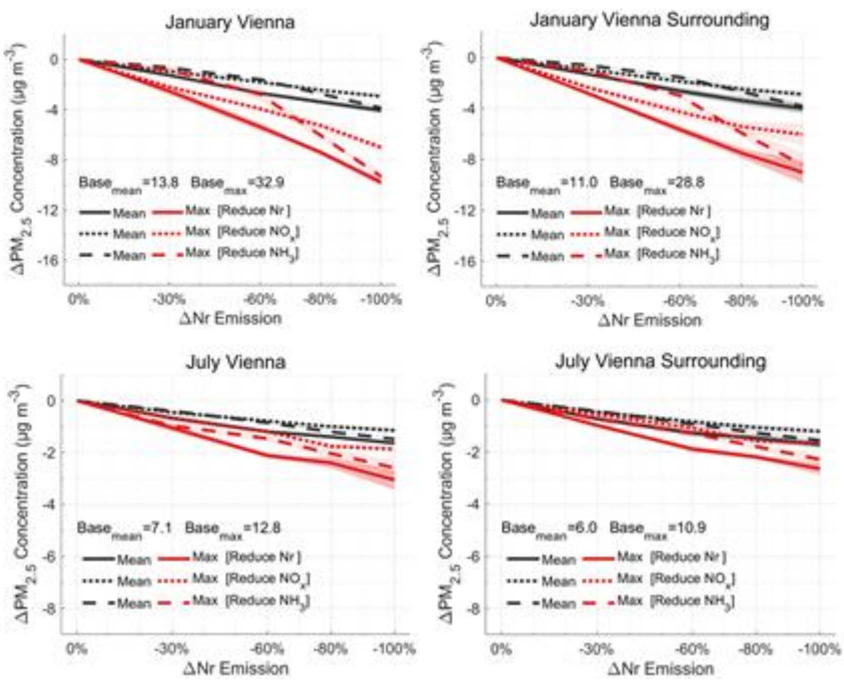

Figure 5. Effectiveness of nitrogen emission reductions in Europe on Vienna (left panels) and Vienna surrounding (right panels) regional surface $PM_{2.5}$ pollution in January (top panels) and July (bottom panels) 2015. The lines show reductions in monthly mean (black lines) and maximum (red lines) $PM_{2.5}$ concentrations by reducing reactive nitrogen emissions for both NO_x and NH_3 (solid lines), only NO_x (short dash lines), and only NH_3 (long dash lines). Numbers inset are their values (ug m^{-3}) in the Base simulation.

Fig. 4 also shows the responses of population-weighted $PM_{2.5}$ concentration (PWC) in North China as a metric more relevant to human health. PWC values show similar but larger responses than the regional geometric means. When NH_3 emissions in North China are reduced by 20-40%, monthly mean BTH PWC could be reduced by 1.4-3.8 μg m^{-3} (1.1-2.9% of PWC) in January and 1.8-3.6 μg m^{-3} (4.3-8.7% of PWC) in July. When NH_3 emissions are reduced by 60-100%, BTH PWC would be reduced by 8.1-26.7 μg m^{-3} (6.2-20.5% of PWC) in January and 5.9-13.2 μg m^{-3} (14.4-32.0% of PWC) in July, illustrating $PM_{2.5}$ air quality improvements we can achieve over BTH by the NH_3 emission controls under 2015 emission conditions.

Fig. 5 shows the responses of $PM_{2.5}$ over Vienna to emission reductions of NH_3, NO_x, and NH_3/NO_x together. It demonstrates that NH_3 emission controls become more effective than NO_x emission controls over Vienna when the emission reduction percentage is higher than ~60% in January, while the effectiveness is always similar or higher in July. We also find the nitrogen emission reductions have similar impacts on average $PM_{2.5}$ concentrations in Vienna and Vienna surrounding area.

2.3.4 A quantitative estimate of different urban agriculture mitigation pathways

The ammonia emitted from agriculture are mainly via livestock breeding and chemical fertilizer application. Therefore, the ammonia reduction potential is evaluated here based on the specific reduction measures in these two components. Eight mitigation scenarios and the baseline scenario are designed to evaluate the ammonia emission reduction potential in agriculture for the year 2015 in BTH. The eight agricultural ammonia reduction scenarios as shown in Table 2 are as follows: (1) feeding scenario, (2) housing scenario, (3) storage Scenario, (4) field application scenario, (5) reduce application rate scenario, (6) improved manage methods scenario, (7) addition scenario; and (8) combine scenario.

We estimate the agricultural ammonia emissions in BTH were 639.83 Gg in 2015. Tab. 1 shows the 2015 annual ammonia emissions in BTH in the baseline scenario and the emission mitigation scenarios. Each individual agricultural mitigation measure can significantly reduce BTH ammonia emissions, ranging from 8.30 Gg-106.36 Gg (1.3% to 18.1%). The combined scenario has the greatest emission reduction potential, with the reduction reaching 56.3%.

Table 1. Annual agricultural ammonia emissions (Gg) under baseline scenario and agricultural emission reduction scenarios

Scenario		Measures	2015 emission / Gg	reduction
baseline		No mitigation measures	639.73	/
Manure	feeding	Use suitable feed for pigs and poultry	594.37	7.1%
	housing	Floor management of the farm house	524.00	18.1%
	storage	Use cover materials and change the pH for slurry; Use compaction, static piling and covering for solid	631.70	1.3%
	field application	Band spreading, injection, incorporation digestate, and solid-liquid separate	560.56	12.4%
Chemical fertilizer	reduce application rate	Optimize nitrogen use rate	600.77	6.0%
	improved manage methods	Change spreading to deep fertilization	548.20	14.3%
	addition	Use urease inhibitor (LIMUS)	551.25	13.8%
combine		All measures above	279.63	56.3%

2.4 References

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3. Nitrogen compounds in soils & groundwater

3.1 Overview

Agricultural soils account for considerable nitrogen leaching to groundwater. In China, especially in the Beijing–Tianjin–Hebei (BTH) region, this exceeds 50%. This poses a risk of adverse health effects such as methaemoglobinaemia in infants and gastric and oesophageal cancers for both urban and rural populations (WHO, 2011). We will quantify regional-scale N leaching from agricultural soils at high spatial resolution and assess their potential groundwater N pollution risks, as well as to optimize agricultural management practices to mitigate the risks.

Based on massive surveys related to agricultural managements and county-scale statistic registers (Zhou et al., 2014), we first develop a high-resolution and real-time N inputs and irrigation datasets from targeted cities or regions (e.g., BTH). We then quantify soil N losses using both land surface model (ORCHIDEE-CROP, Wang et al., 2017) and flux upscaling approach (Gao et al., 2016, Hou et al., 2017). Based on the updated land surface model, we finally assess the mitigation potentials of different measures, and identify the matrix of measures for different regions.

We find that soil N leaching fluxes in Beijing and Shijiazhuang ranged from 3.4 to 4.4 kg $N \cdot ha^{-1}$, clearly lower than the threshold (22.6 kg $N \cdot ha^{-1}$) estimated by De Vries et al. (2013). There is a large potential to mitigate soil N losses in China. When jointly optimizing fertilizer management and food consumption, the total NH_3 volatilization and N leaching could be reduced by 65% relative to the business-as-usual scenario in 2050, and China would show a much lower soil N emission intensity (0.52 g N $kcal^{-1}$ yr^{-1}), comparable to that of the USA (0.49 g N $kcal^{-1}$ yr^{-1}) and the EU (0.45 g N $kcal^{-1}$ yr^{-1}). Half of the N loss mitigation could be achieved on 26% of crops, concentrated in Huaihe and lower Yellow River Basin. This result implies the importance of this region on food production and highlights the benefit of focusing on a small area that could deliver large soil N loss mitigation.

3.2 Objectives

Under the scope of the UNCNET project, we quantify regional-scale N leaching from agricultural soils at high spatial resolution and assess their potential health risks, and we present optimized agricultural management practices to mitigate such health risks. There are three tasks: (i) development of high-resolution N inputs and irrigation datasets, (ii) Land surface modeling simulation of N leaching and the associated N flow in aquifer groundwater under different agricultural management and climate change; (iii) Optimization of urban agricultural N management to mitigate groundwater N pollution under different climate changes.

3.3 Results

3.3.1 Development of high-resolution N inputs and irrigation datasets from agricultural soils

For N inputs, we first collected nationwide surveys of county-scale (the third-level administrative division) synthetic N fertilizer applied to croplands (FSN, kg N yr^{-1}) for ~ 2900 counties in Mainland China, Taiwan, Hong Kong, and Macau for the period 1990-2014. The other data was supplemented by the survey of the Program on Systematic Analysis and Comprehensive Treatment of Agricultural Environment or the Agricultural Information Institute of Chinese Academy of Agricultural Sciences (AII-CAAS; http://aii.caas.net.cn/). These data were further disaggregated by nine types of crop, based on the crop-specific, provincial data of Rijt from the Statistics of Cost and Income of Chinese Farm Produce (http://tongji.cnki.net/overseas). In addition, China has experienced changes of County-scale administrative divisions, such as aggregation, disaggregation, and name changes, so we harmonized the temporal evolution of FSN to fit the latest administrative divisions (http://geodata.pku.edu.cn), based on the historical trajectories summarized by the Ministry of Civil Affairs of China (http://xzqh.mca.gov.cn/). Second, we estimated annual N in livestock manure, human excreta, and crop residues returned to croplands by the Eubolism model at county scale (Chen, Chen, & Sun, 2010), based on county-scale activity data, such as the numbers of livestock by animal, rural population, and yields by crop type. The Eubolism model has been evaluated against multi-site observations in highly-fertilized cropping areas across China. Third, dry and wet deposition of N species were quantified by the global aerosol chemistry climate model LMDZ-OR-INCA at a horizontal resolution of 1.27° latitude by 2.5° longitude (Wang et al., 2017), in which wet N deposition fluxes have been validated by a recent global dataset (Vet et al., 2014). Finally, crop-specific N application rates (Rijt) were calculated as county-scale N input totals (i.e., synthetic fertilizers, manure, human excreta, crop residues, and N depositions) divided by the associated sowing areas that were obtained from the statistical yearbooks of 31 provinces (http://tongji.cnki.net/overseas). This new county-scale dataset of Rijt was then resampled into a 1-km grid map based on the dynamic cropland distributions (Liu et al., 2014). We assumed a maximum N fertilizer application rate of 700 kg N ha^{-1} based on a previous study (Carlson et al., 2017).

For irrigation water use, we reconstructed a new National Long-term Water Use Dataset of China (NLWUD). The NLWUD includes irrigation water use and irrigated area used to calculate WUI for 5 crop types (wheat, maize, rice, vegetables and fruits, and other crops) in 341 prefectures during the period 1965–2013. The data of water use by sub-sector and prefecture were obtained from two nationally-coordinated surveys: the 1st and 2nd National Water Resources Assessment Program for the period 1965–2000 and the Water Resources Bulletins of 31 provinces for the rest of period 2001–2013. Both of these surveys were led by the Ministry of Water Resources, and had an identical survey methodology including definition, survey unit, sector classification, field survey or measurements, and quality assurance. China experienced an extensive shift of prefectural boundaries during the past five decades, and these changes have resulted in discontinuities in the spatial data. We therefore harmonized the temporal evolution of water use to the 2013 administrative map of China, based on the history of boundary and name changes given by the Ministry of Civil Affairs of China.

The prefectural-scale irrigated area by crop were obtained from the statistical yearbooks of China's 31 provinces, given by the provincial statistics registers. It should be noted that the reconstructed data sets contain some uncertainties, but should not affect our findings unduly. Crop-specific irrigation rates (mm yr^{-1}) at the prefectural level were then calculated as cropland irrigation amounts divided by irrigated areas, and were resampled into the gridded irrigation maps of HYDE 3.2.1.

3.3.2 Land surface modeling simulation of N leaching and health effects assessment

The ORCHIDEE-crop model was used to predict how climate change and management practices regulate N losses from different soils. The model has been successfully applied to simulate hydrological and N processes and vegetation growth for both natural and managed systems across China. The model integrated meteorological module, combined transport module of soil water, heat and nitrogen, crop growth module, organic matter module, root water and nitrogen absorption module, inorganic nitrogen module and field management module. The Penman-Monteith equation recommended by FAO was used to calculated Potential evapotranspiration (ET). Crop parameters in the model (specific leaf weight, crop coefficient, distribution coefficient, etc.) are corrected according to observed values.

Additional model improvements had been conducted for soil water dynamics, heat dynamics, and solute transport. First, surface water runoff is simulated for daily rainfall using the SCS curve number equation (Mockus, 1972). Subsequently, soil water infiltration is computed using the Green-Ampt model (Green & Ampt, 1991). The process of soil water redistribution was incorporated into the model using the Richard's equation as described by Šimůnek et al. (2008). The reference crop evapotranspiration ET_0 is estimated using the Penman-Monteith equation (Allen et al., 1998) solved using standard grass with an assumed height of 0.12 m and a surface resistance of 70 s m^{-1}. The crop coefficient is used to calculate actual crop potential evaporation. And then, using leaf area index (LAI), separate potential crop transpiration and potential soil evaporation (Jones, 1986).

Second, the simulation of one-dimensional heat transfer was taken from the HYDRUS1D model, which is described with the convection-dispersion equation (Šimůnek et al., 2008). The top and bottom boundaries are set constant boundary conditions. The temperature of the top soil layer is estimated based on the daily air temperature and leaf area index, and the bottom boundary temperature is estimated used by the method used in the DNDC model (Li et al., 1992).

Third, the transport of soil NH_4^+-N and NO_3^--N was simulated using the convection-dispersion equation (CDE), and a generalized nonlinear adsorption isotherm was used to consider the adsorption-desorption process between the liquid and solid phase as described in HYDRUS1D model (Šimůnek et al., 2008). The model assumes an equal crop absorption ratio of NH_4^+-N and NO_3^--N. Each N transformation process was computed as a sink-source term in the CDE, and each of the processes are described into detail in the next two sections. The boundary conditions (Cauchy type) for the solute (NH_4^+-N and NO_3^--N) transport equation was used to describe the solute flux at the upper or lower boundary. This CDE was solved by the general upwind difference method, and this procedure effectively avoids numerical dispersion and oscillation even under the conditions of dramatic changes in solute concentration without using dense nodes (Chen, Guo, Hu, & Li, 2005). Surface broadcast and deep fertilizer applications are regarded as uniformly incorporated within the top 1 cm of soil or at the prescribed application depth (usually 5–10 cm) in the soil, respectively.

Field observations of N and P runoff losses and relevant variables were collected from an observation network containing six sites from April 2017 to November 2018. The sites are Fangzheng (FZ, Heilongjiang), Panjin (PJ, Liaoning), and Jingzhou (JZ, Hubei), Chaohu (CH, Anhui), Anlu (AL, Hubei), and Gaoan (GA, Jiangxi) (Group 1). To validate the performance of improved ORCHIDEE-CROP model, additional field observations were observed from April 2018 to November 2018 at the other five sites in south China. The sites are Nanxian (NX, Hunan), Nanchang (NC, Jiangxi), Changsha (CS, Hunan), Huizhou (HZ, Guangdong), and Nanning (NN, Guangxi; Group 2) for the double rice system. The observation network covered various cropping systems, and a broad range of climate and edaphic conditions in China. Finally, model parameters were calibrated based on the event-based observations from two-year rainfall simulation experiments, and further validated based on the daily observations from 11 sites (Groups 1 and 2). Soil water content was determined first, following by N concentrations and thereby N emission fluxes. The percent bias (BIAS) and Nash-Sutcliffe efficiency (NS) were ranged from –1.9% to 3.5% and from 0.87 to 0.96 for the daily soil water content observed at six sites of Group 1, respectively. This module performed well in producing the daily variabilities of N concentration, with BIAS of <11.0 % and NS of 0.3 ~ 0.7. The improved model well captured the spatial contrast and temporal variation of N leaching observed at six sites and from rainfall simulation experiment (n = 91), with the BIAS of 6.2% ~ 12.3% and the NS of >0.8. The accuracy of N leaching estimates was further confirmed by external cross-validation at five sites of Group 2 (n = 36, BIAS = −7.3 ~ 2.2%, NS > 0.9,). Overall, the model is proven to be capable of simulating daily N leaching fluxes under contrasting climate and edaphic conditions.

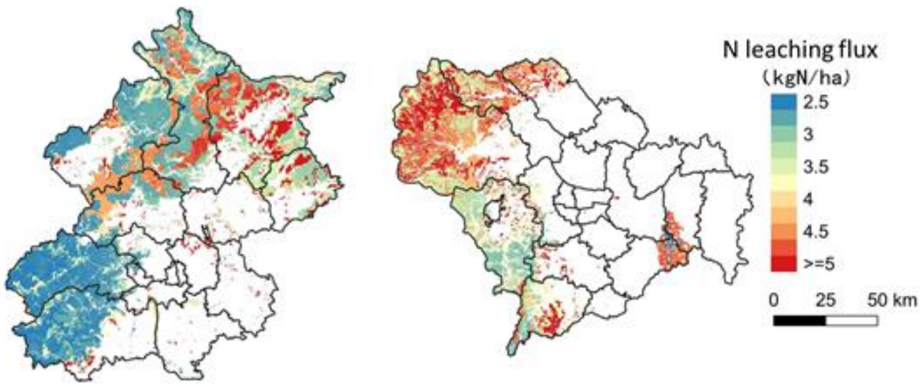

Figure 6. Patterns of soil N leaching fluxes in Beijing (left) and Shijiazhuang (right).

Maps of soil N leaching due to N inputs (i.e., synthetic fertilizers, manure, crop residues, and atmospheric depositions) were produced over Beijing and Shijiazhuang by the validated

ORCHIDEE-CROP model. The estimate of N leaching from urban agriculture was 2.8 kton N·yr-1 in Beijing and 1.8 kton N·yr^{-1} in Shijiazhuang. Accordingly, soil N leaching rates were 6.9% and 8.7%, respectively. Fig. 6 shows that the N leaching from the model varies across two cities. For Beijing, larger N leaching fluxes were mainly concentrated in the northern part (e.g., Miyun, Huairou, Pinggu), with mean N leaching fluxes of >4 kg N·yr^{-1}. However, higher mean N leaching rates were found in the suburban regions such as Chaoyang (11.8%), Tongzhou (11.8%), and Shunyi (9.5%). For Shijiazhuang, larger N leaching fluxes were found in both western and eastern parts such as Pingshan (847 kton N·yr^{-1}), Jingxing (289 kton N·yr-1), and Ningshou (206 kton N·yr^{-1}). Higher mean N leaching rates were found in Xingtang (10.1%), Pingshan (9.5%), and Ningshou (9.2%). In addition, soil N leaching fluxes in Beijing and Shijiazhuang ranged from 3.4 to 4.4 kg N·ha^{-1}, considerably lower than the threshold (22.6 kg N·ha^{-1}) estimated by De Vries et al. (2013).

3.3.3 Optimization of urban agriculture management to mitigate groundwater N pollution under different climate changes

To explore the future N loss mitigation potential, we performed four scenario projections in ten-year intervals from 2020 to 2050. In the business-as-usual (BAU) scenario, we only consider current (the year 2017) policies and national plans without any further intervention. However, the crop production will increase in line with projected increases of population and gross domestic product (GDP) as projected by Zhang et al (2020). Meanwhile, climate factors, i.e. air temperature and wind speed, changed following a conservative RCP2.6 (stringent mitigation scenario, predicts the global mean temperature increases of up to 2 °C by 2100) future climate change scenario (PICIR, 2021). Scenarios OFM and OFC predict the projections based on the same assumptions as BAU, but optimize fertilizer management (OFM) and food consumption (OFC), respectively. For scenario OFM, N fertilizer rate was set according to the "N Surplus Benchmarks in China" following Zhang et al. (2019). Meanwhile, the incorporation proportion of synthetic-N fertilizers will achieve 80% for three staple foods (i.e., wheat, maize and rice) according to the National Agriculture Mechanization Extension Plan (Zhang et al., 2020). For scenario OFC, the crop production will decrease by optimizing human diet structure following Zhang et al. (2020) and cut 50% of food loss and waste to achieve the Global Sustainable Development Goals (Clark et al., 2020; FAO, 2020; X. Li et al., 2021). To achieve the most ambitious mitigation target, the ALL scenario was proposed to combine all the mitigation options identified in OFM and OFC scenarios. It should be noted that for the intermediate year of scenario OFM, OFC and ALL, we assume linear adoption from 2017 until the adoption year (2050), at which point the technologies are entirely adopted (Clark et al., 2020).

China's crop demand is projected to increase by 140% by 2050 considering both economic development and population growth. This would require an additional sowing area of 35.4 Mha, with the total NH_3 volatilization and N leaching arriving at 4.9 and 2.5 Tg N by 2050 if maintaining the 2017 management practice under increasing temperature conditions. Under BAU, soil emissions of NH_3 volatilization and N leaching in 2030 (5.3 and 2.2 Tg N) would exceed the peak level in 1996 and 2015 and steadily increase until 2050. Soil N loss abatement through optimizing diet composition and cutting food losses and waste (OFC) could reduce emission by 20% in 2050 compared with BAU (Figs. 7a and 8a). When conducting optimal fertilizer management (OFM), N fertilizer consumption would reduce by 50.5%, inducing a subsequent N loss reduction of 63% compared with BAU in 2050. To achieve the most ambitious mitigation target, the ALL scenario combined all the mitigation options identified in OFW and OFC. The estimated total NH_3 volatilization and N leaching of the ALL scenario are 2.1 Tg N in 2050 (65% reduction relative to BAU). Under scenario ALL, China would show a quite low soil N emission intensity (0.52 g N $kcal^{-1}$ yr^{-1}) in 2050, which is closer to that of the USA (0.49 g N $kcal^{-1}$ yr^{-1}) and the EU (0.45 g N $kcal^{-1}$ yr^{-1}).

Spatially explicit information of soil N loss mitigation potential could help us to identify specific crops and hotspot areas, which may be attractive ‘mitigation targets’ (Figs 7c and 8c). We ranked gridded mitigation potentials from largest to smallest, and then added the value to the sum of its predecessors, resulting in cumulative mitigation potential up to a given point of sowing area. Soil N loss mitigation potentials were unevenly distributed across China. A half of the N loss reduction could be achieved on 26% for all crops together. Total mitigation potentials were concentrated in Huaihe and lower Yellow River Basin, which contributed about half of the total. This result implies the importance of this region on food production and highlights the benefit of focusing on a small area that could deliver large soil N loss mitigation. It should be noted that the Beijing-Tianjin-Hebei region is not central for soil N loss mitigation, where it accounts only for 8% of the total, because of relatively lower N losses in dry and cool areas.

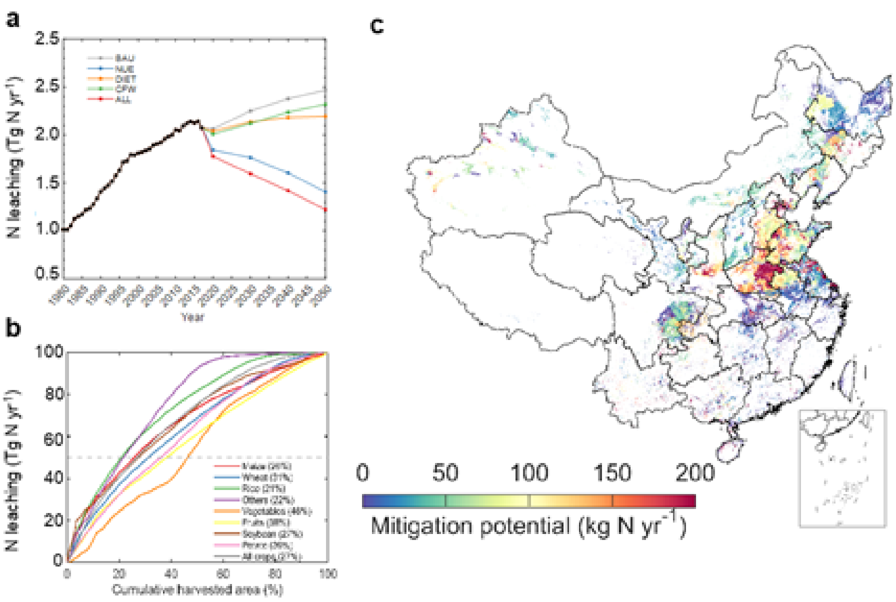

Figure 7. China’s cropland N leaching mitigation potentials. (a) Future N leaching under four scenarios; (b) China’s cropland N leaching mitigation potentials by crop under scenario ALL; (c) Spatial pattern of China’s cropland N leaching under scenario ALL.

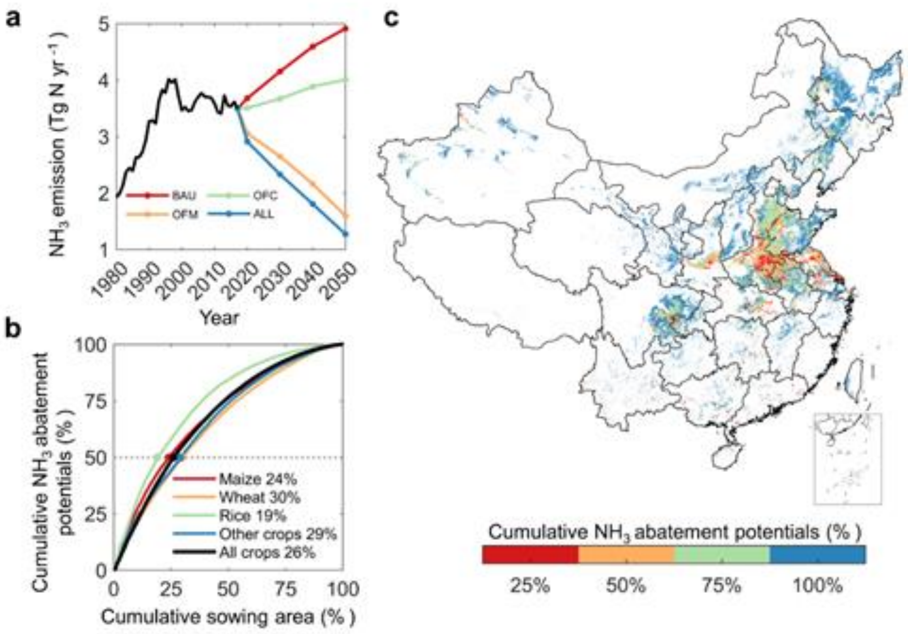

Figure 8. China's cropland-NH$_3$ mitigation potentials. (a) Future NH$_3$ emission under four scenarios; (b) China's cropland-NH$_3$ mitigation potentials by crop under scenario ALL; (c) Spatial pattern of China's cropland-NH$_3$ emission under scenario ALL.

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4. Nitrogen flows in Urban Agriculture

4.1 Concept for Urban Agriculture

Urban agriculture is an indispensable part of the urban N budget, though cities rely less on urban agriculture for food security. Here, the urban agriculture framework splits into 5 components, including cropland, urban green, horticulture, livestock, and pets mainly according to their functions and roles in the urban N cycle (Fig 9). In addition to these 5 main pools, there are 4 other main pools outside the urban agriculture N framework but included in the overall urban N budget, which relate to N exchanges from/to atmosphere and hydrosphere, waste of N and trade of N across the urban agriculture boundary. Urban agriculture plays a dominant role in animal food consumption. Hence, livestock production is more likely located around the urban area to be closer to the delivery market, mainly pig and poultry production, which is less reliant on the roughage feed due to high transportation costs. All nitrogen fluxes and their interactions were identified by the extended NUFER model (Ma et al., 2019), together with the linkages among sub systems within an urban and peri-urban areas.

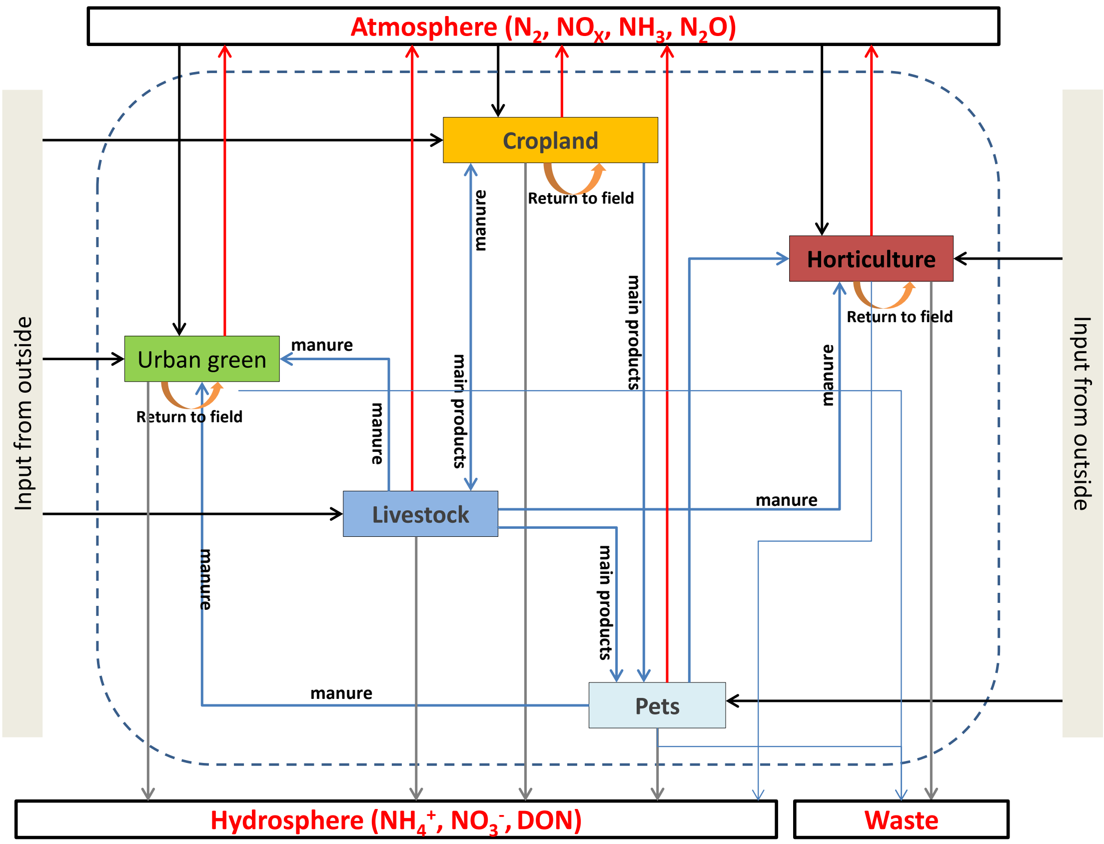

Figure 9. Detailed nitrogen cycle in the urban agriculture

N flows cycling across the 5 subsystems through the specific interactions are shown in Fig. 9. Each flow is indicated by the N output from one subsystem and the N input to another system. All N input will eventually become N output or accumulate within the test areas after cascading through multiple subsystems.

The following examples take advantage of the comprehensive quantification of agricultural flows in China using the NUFER model (Ma et al., 2019), in part extended with the CHANS model (Gu et al., 2015). This approach allows to take advantage of Chinese statistical information (taken from national and provincial statistical yearbooks) in an optimal way. Hence the examples below are limited to the Chinese test areas, in their spatial boundary as outlined in Fig. 1 and with core area and peri-urban area as separate entities. Approaches for Europe took advantage of similar strategies but were based on different statistical information available and included in the comparison sections only.

4.2 Test area Shijiazhuang

The total Nr input in agricultural land system in Shijiazhuang core area is 8.2 Gg. The dominating anthropogenic inputs included flows from household, animal manure and BNF, which account for 46%, 25% and 11%, respectively. Total Nr inputs in urban green and horticultural land are less than in agricultural land: the total Nr input of urban green is 1.1 Gg and total Nr input of horticultural land is 0.2 Gg. Fertilizer and deposition are dominating inputs, which account for 50% and 33%, respectively. Because Shijiazhuang barely plants horticulture and most horticultural products are imported from other areas, the imports are dominating N inputs in the horticulture system (88% of). For livestock, total Nr input is 2.7 Gg, and urban plant to livestock Nr amounted to 75% of total Nr input. The industrial Nr creation amounted to 18% of total Nr input. For the pet system, all N input comes from pet food.

The total N outputs are 5.9 Gg N in the agricultural land pool, among which about 33% is discharged into the environment, and 33% used by livestock as fodder. For urban green, 19% of total Nr output is lost as waste. Even larger is the share of waste from horticulture, as after use for decoration, most of ornamental plants end up in waste, accounting for 64% of total Nr outputs. In the city centre, the share of manure returned to fields is higher than in the surrounding area. Returning livestock manure to fields is the dominating output of animal husbandry, which accounts for 73% of total Nr output. Further 17% of total output is discharged to air, potentially becoming a pollution hazard.

In the peri-urban area of Shijiazhuang total Nr input in the agricultural land pool is much larger at 1136 Gg N. The dominating anthropogenic inputs include mineral fertilizer and animal manure, 72% and 13%, respectively. Total Nr input in urban green and horticultural sub-pools are far less than in agricultural land: they amount to 1.5 Gg N and 0.005 Gg N respectively. For the urban green sub-pool, fertilizer and atmospheric deposition are dominating inputs, which account for 45% and 30%, respectively. Also, the surrounding areas of Shijiazhuang barely have horticulture, hence most horticultural products have to be imported (61% of total Nr input). The total Nr input of livestock pool is 229 Gg N, among which the flow from urban plant to livestock (fodder production from agricultural land) and fodder import amount to 62% and 24% of total Nr input, respectively. For pets, the major share of N input comes from pet food. Still, 25% of total pet food comes from household residuals in the surrounding areas.

The total N outputs are 640 Gg N in the agricultural land pool, and approximately 54% is Nr that is discharged into the environment. The major reason is that the level of N management in the surrounding areas is lower than in the city area. 22% of total agricultural land output is returned to livestock pool as fodder. For urban green, 65% of total Nr output is emitted to air, including emissions of N_2O and NH_3. Outputs of the households pool are dominated by flows to wastewater and agricultural land, together accounting for 85%. Livestock manure returned to fields makes up for 65% of Nr output of the livestock pool. The discharge to air due to NH_3 and N_2O volatilization (14% of total Nr output) may potentially lead to serious environmental effects.

4.3 Test area Beijing

In the Beijing core area, total Nr input to agricultural land is 4.9 Gg N, with households, mineral fertilizer and atmospheric deposition accounting for 73%, 13% and 11%, respectively. Total Nr input to urban green is 2.9 Gg N, which is less than in agricultural land. Due to strong demand to decorate streets and squares with lots of flowers, the total Nr input to horticultural land has a similar value as for agricultural land, at 4.8 Gg N. Most ornamental plants are not produced locally, however, but need to be imported from outside the city borders. Hence N import is the dominating input in the horticulture pool, accounting for 87% of total Nr input (4.1 Gg). The total Nr input of the livestock pool is 0.5 Gg N, and the Nr flow from household to livestock (kitchen residues recovered from households as animal feed) amounted to 84% of total Nr input. The flow from agricultural land to livestock (harvested crops used as feed) amounted to 10% of total Nr input. For the pet pool, all N input comes from pet food.

The total N outputs are 2.3 Gg N in the agricultural land sub-pool, of which 46% and 48% are Nr losses to water and to air (N_2O, NH_3), respectively. Additionally, 2% of total agricultural land output is returned to the livestock pool as fodder. For the urban green sub-pool, 73% of total Nr output are losses to air. The atmospheric emissions and waste are the dominating outputs in the horticulture pool, accounting for 37% and 27% of total Nr outputs, respectively. Furthermore, the flow from horticulture to households and losses to water account for 25% and 11% of total Nr output, respectively. For the livestock sub-pool, the flow from livestock to industry is the dominating output, accounting for 58% of total N output. For dogs, we assume that their excretion in the core city area is deposited in urban greens, while for cats we understand this is the case only for half of the excreted amounts, the other half being transferred to waste. As a result of Nr inputs and outputs in the urban plant pool, the Nr accumulations amount to 6.4 Gg N. Both agricultural land and urban green areas contribute to the accumulation, with the agricultural land sub-pools accumulating 75% and urban green sub-pools 22% of the total accumulation.

With Beijing's peri-urban area consisting of much larger agricultural land, application of mineral fertilizer is the major N input, followed by animal manure. These account for 85% and 7%, respectively, of the total Nr input of 192 Gg N in agricultural land. Input to urban green and horticultural sub-pools are much smaller than in the agricultural land sub-pool, amounting to 14.7 Gg N and 11.2 Gg N of the total Nr input, respectively. For the urban green sub-pool, imported fertilizer and deposition are dominating inputs, which account for 78% and 19%, respectively. As for Beijing core area, the surrounding areas also need large quantities of horticultural crops to decorate streets and squares, which needs to be imported from other areas. Hence the imported N constitutes about 87% of total Nr input (9.7 Gg N). The total Nr input of livestock pool is 64.1 Gg N, from which imported fodder comprises 88%. For the pet pool, all N input comes from pet food.

The total N outputs are 99 Gg N in the agricultural land pool, from which 87% are Nr losses to the environment. The major reason is that the level of N management in the surrounding areas is lower than in the city area. 7% of total agricultural land output returns to livestock pool as fodder. For the urban green sub-pool, 68% of total Nr output is discharged to air. In horticulture, waste and atmospheric emissions are the dominating outputs (37% and 30% of Nr output, respectively). The flow from livestock to industry is the dominating output of the livestock sub-pool and accounts for 48% of total Nr output, while release to air and application on agricultural land (i.e., manure N) have similar magnitudes, 19% and 22%, respectively. For the pet pool, we assume that all excretion is treated as waste. As a result of Nr inputs and outputs in the urban plant pool, the Nr accumulations are 101 Gg N. Agricultural land and urban green sub-pools contribute to 92% and 8% of the total accumulation, respectively.

4.4 Agricultural N Indicators and the SDGs

Developing agricultural N budgets offers opportunities to extract meaningful indicators from such a dataset (Kanter et al., 2016). The UN Sustainable Development Goals (SDGs) require multiple such indicators to be developed from readily available data (see e.g. Schmidt-Traub et al., 2017). Hence, we establish relationships between data derived in the nitrogen budget and the SDGs.

To perform a quantitative analysis, we need to identify SDGs for which threshold values are available. Using the online databases for the Sustainable Development Report (SDR) 2022 and European Sustainable Development Report (ESDR) 2021 (Lafortune et al. 2021; Sachs et al. 2022), as well as references from the literature (Zhang et al. 2015, 2021; Winiwarter et al. 2020), we identified 8 nitrogen indicators for which thresholds have been quantified: Sustainable Nitrogen Management Index (SNMI), Nitrogen Use Efficiency (NUE), N surplus, NH_3 emissions from agriculture, NO_3^- concentration in groundwater, Recycling rate, Production-based N emissions, and N_2O emissions from agriculture. On the basis of the above literature, each of the 8 N indicators was connected to one of the 17 SDGs at a time, following a single-goal mapping framework.

To enable cross-comparison among indicators and to identify priorities for improvement in a city's performance, we designed a methodology for score calculation following the framework adopted in the latest Sustainable Development Report (Sachs et al., 2022). The score of each indicator provides an overview of the sustainability performance of each city. For some of the selected indicators, (E)SDR online databases (Lafortune et al., 2021; Sachs et al., 2022) provide an optimum and a lower boundary value. We understand the actual value of each indicator being in between, and we define the score between 0 (lower bound) and 100 (optimum) as the linear value (in %). For the remaining N indicators for which lower bound and optimum values are not available as such, available low ("red") and high ("green") thresholds from the literature are collected assuming that they correspond to scores of 33 and 67 respectively (i.e., one-third and two-third of the 0-100 score scale). In this case, the score is calculated according to equation (1), linearizing this time between the “red” and “green” thresholds:

$$Score_i = 33 + 33 * \frac{Raw_i - Red_i}{Green_i - Red_i} \qquad (1)$$

where Raw_i is the value of the indicator, Red_i is the value of the lower (red) threshold, and $Green_i$ the value of the higher (green) threshold for the i’th indicator. In both cases the score values $Score_i$ are set to 0 and 100 when they are lower than 0 or higher than 100, respectively. The lower bound, red, green, and optimum threshold values are summarized for each selected N indicator in Tab. 2.

Table 2: Green and red thresholds of the N indicators, plus lower bounds and optimum values. In a traffic light system, values below the green thresholds would be shown positively, in green, and values above the red threshold problematic, in red (except for the recycling rate, where the situation is opposite as a high value points towards benign practice)

Indicator	Unit	Lower bound	Red threshold	Green threshold	Optimum	Reference
SNMI		1.2	0.7	0.3	0	Sachs et al. 2022
NUE	%		0.68	0.42		Zhang et al. 2015
Nitrogen surplus	kg N/ha	200	100	50	10	Lafortune et al. 2021
NH_3 emission from agriculture	kg NH_3/ha	60	45	20	8	Lafortune et al. 2021
[NO_3^-] in groundwater	mg NO_3^-/L	60	50	25	10	Lafortune et al. 2021
Recycling rate	%		7	20		Winiwarter et al. 2020
Production-based N emissions	kg N/cap	30	20	10	2	Lafortune et al. 2021
N_2O emissions from agriculture	ton CO_2eq/ha		0.51	0.41		Zhang et al. 2021 + GAINS model

Based on the above framework, we designed an Urban SDG Index aggregating the various scores of the respective N indicators into a single number, ranging from 0 to 100. The index was calculated as the average of the score of the respective SDGs, considering an equal weighing for each goal. For the score goals calculated on the basis of several N indicators (i.e., SDG 2 and SDG 12), an equal weighing for each N indicator was similarly assumed.

Fig. 10 provides a comparison of the UNCNET test areas for their performance regarding the SDG’s, based on this indicator approach (both urban cores and peri-urban areas). The comparison indicates sustainability challenges in several aspects for Shijiazhuang that are much less relevant for the European cities, even though also in Europe the improvement of NUE and N surplus remains still to be solved also for urban areas.

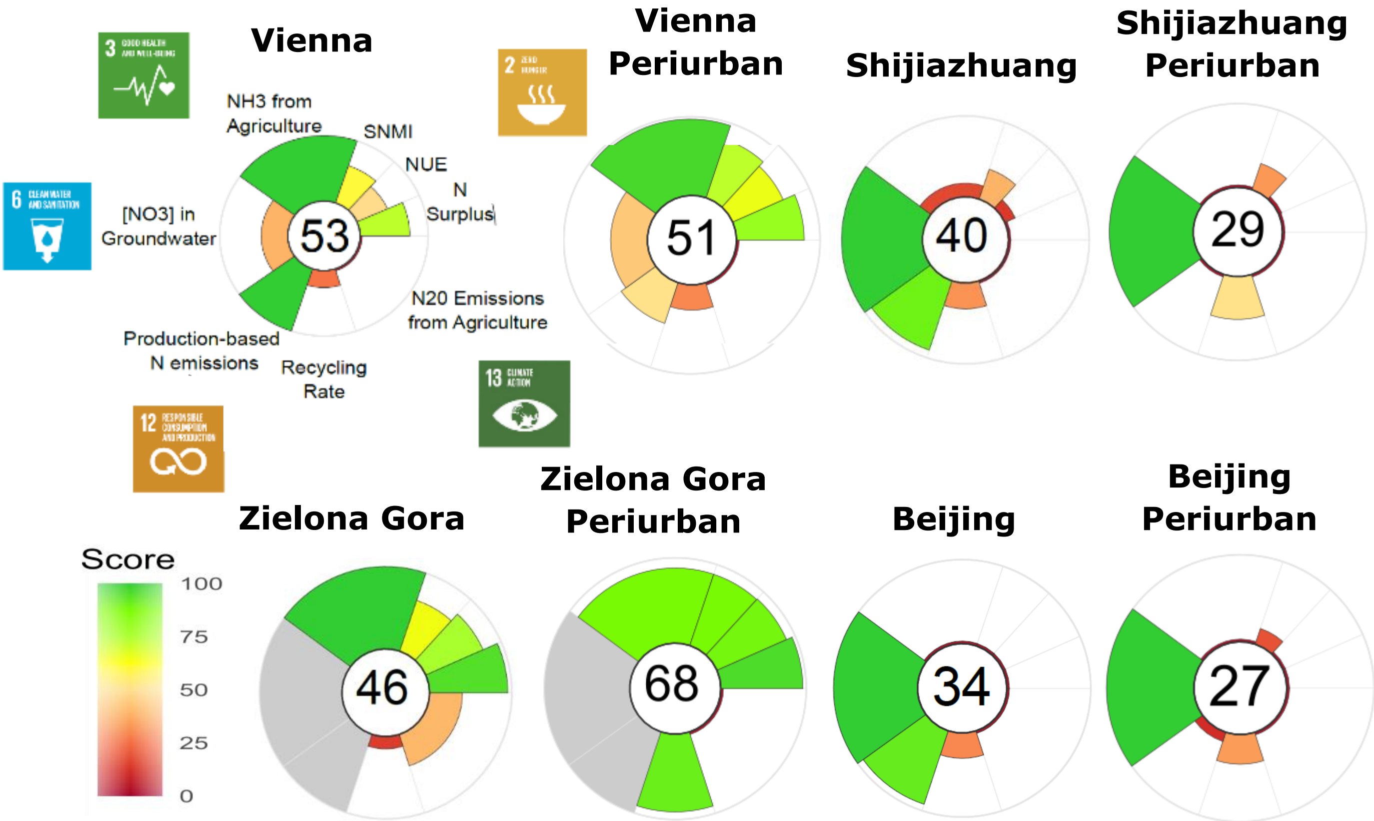

Figure 10. Urban SDG Index to benchmark achievements of SDG's for all of the UNCNET test areas, using a traffic light approach. Same width is provided for each SDG, thus narrower pies are shown where an SDG is characterized by more than one goal.

4.5 References

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5. Human dwellings and waste

5.1 Nitrogen compounds and opportunities for recycling

The human consumption system is a central element for cycling Nr in an urban environment. This clearly is the case for the agro-food chain, with households receiving (and people consuming) food, and with Nr being passed towards waste or wastewater. But also industry and combustion serve households via fuels and materials, that potentially after use are disposed of as waste. Significant amounts of Nr in waste and wastewater are generated. Sewage sludge and food waste constitute the largest share of nitrogen in waste generated in the urban environment (Buckwell and Nadeu, 2016). Apart from being the source of large amounts of N emissions, waste and wastewater treatment also offer a great opportunity for N recycling within the urban system. Households and the waste system are thus intrinsically connected, with waste fluxes typically providing more detailed information.

The physical and chemical components of waste/wastewater and their mass depend on eating habits, standard of living, degree of commercial activity, season, dynamics of economic development, cultural conditions, technical and sanitary equipment, type of industry and its efficiency. The type of waste and wastewater treatment technology used, how it is collected and transported, and the amount of nitrogen recovery and reuse have a significant impact on nitrogen emissions. Main sources of N_2O emissions are wastewater treatment, sewage sludge incineration, municipal solid waste incineration, biomass combustion for energy production, incineration of waste-based fuels with high content of nitrogen. Adequate urban waste management is an essential component of proper management of nitrogen flows in urban and peri-urban areas aimed at nitrogen recovery and reducing negative environmental and health impacts.

The urban nitrogen cycle associated with waste and wastewater management is shown in Figure 11. According to the EU Waste Framework Directive (2008/98/EC, updated as 2018/851) and its “waste hierarchy”, proper waste management first involves waste prevention, followed by preparation for reuse, recycling, recovery (e.g., recovery of energy, materials, or raw materials), and, if none of the initial four steps can be achieved, environmentally sound disposal. This approach is an integral part of the ongoing process over the past few years of rebuilding the waste management system from a linear, resource-flow-based economy to a closed-loop economy. According to the Directive, recycling is understood as recovery in which waste is transformed back into products, materials, or substances for primary or other use, in addition to energy recovery purposes. Recycling also includes the conversion of organic material (organic recycling) to a substance or material with the properties of a primary or other substance/material but does not include energy recovery and reprocessing into materials to be used as fuels or for pit filling purposes.

Currently, one of the most economically viable ways to recycle waste is to use it as fertilizer (Pesonen and Rautio, 2019). A similar situation applies to sewage sludge generated at wastewater treatment plants. Composting and digestion processes are counted as recycling, on the condition that the resulting product (stabilized matter, compost, digestate) is no longer waste and can be used as a fertilizer or soil improver in agricultural applications or the reclamation of degraded land.

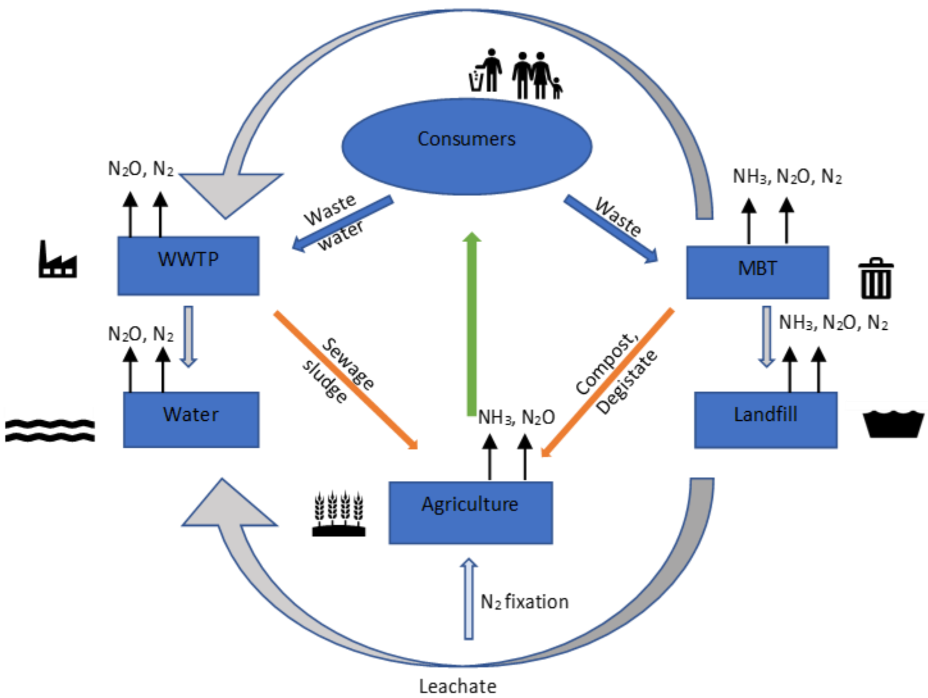

Figure 11. Nitrogen cycle in the city related to waste and wastewater management (WWTP denotes a wastewater treatment plant, MBT stands for mechanical-biological waste treatment plant)

Considering that the main objectives of waste management are aimed at maximum reduction of landfilling with effective separate collection of raw waste (for recycling), mechanical-biological treatment (MBT) with stabilization of residual waste before landfilling will not be a desirable solution in the future. MBT will be transformed into facilities that treat selectively collected waste and produce fertilizer from biowaste. The stabilized waste and ballast resulting from residual waste treatment will be subjected to recovery processes before being discharged to landfill (e.g.: by increasing the separation of recyclable and combustible materials or using them for road construction or the production of concrete (Bernat et al., 2021; Połomka and Jędrczak, 2020; Boer, 2018). However, new technologies in this area are in the research and development phase.

The road to a circular economy requires the development of new technologies aimed at the treatment of bio-waste. Technologies for the production of organic-mineral fertilizers from bio-waste allowing highly efficient return of nitrogen to soils or recovery of nitrogen from leachate, filtrate water or digestate are already known. However, these methods are not yet commercially applicable on a large scale.

5.2 Waste recycling in Zielona Góra

Recycling and recycling rates are a function of economic attractiveness, technical potentials, and legal requirements. Legal obligations for the management of municipal waste (household waste and similar waste) are defined in the EU Waste Framework Directive. Such obligations include the goal of preparing 50% of municipal waste for reuse/recycling by 2020. Using data from Zielona Góra, Poland, we illustrate observed and projected developments.

Tab. 3 shows estimated levels of recycling and reuse (according to the definitions in the Directive) of different fractions of municipal solid waste (MSW) in Zielona Góra. Given that selectively collected waste needs to be cleaned to remove contaminants and non-recyclable waste, more of each type of waste needs to be collected than the required recycling levels in order to achieve the required recycling levels after cleaning at the sorting plant and taking into account material losses during treatment processes.

Table 3. Estimated levels of recycling of municipal waste collected selectively in the city and municipality of Zielona Góra from 1995 to 2015 and the levels under current law, from 2020 to 2030. Projected values are marked in gray.

	Level of reuse and recycling of MSW components [%].									
	Values achieved							Values required by law		
	1995	2000	2004	2005	2010	2014	2015	2020	2025	2030
Bio-waste	0.0	4.6	7.9	8.8	12.6	17.6	20.2	25.0	60.0	65.0
Paper and cardboard	0.2	2.0	17.5	17.2	15.6	15.9	17.2	50.0	80.0	85.0
Multi-material packaging	0.0	0.0	0.0	0.0	0.0	1.0	1.8	10.2	30.0	45.0
Plastics	0.6	1.1	3.4	3.8	8.2	12.1	13.1	50.0	55.0	55.0
Glass	1.5	5.9	11.4	11.9	4.9	13.9	28.4	50.0	70.0	75.0
Metal	0.0	11.3	17.3	19.7	21.4	20.9	21.3	50.0	75.0	80.0
Clothing, textiles	0.9	0.8	0.4	0.4	0.3	0.4	0.6	15.3	50.0	70.0
Wood	0.0	3.0	2.5	1.5	0.7	0.7	0.7	7.5	25.0	30.0
Hazardous waste	0.0	0.0	0.0	0.0	0.0	12.4	26.1	20.2	60.0	70.0
Ash fraction <10 mm	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Mineral waste	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Other	0.0	2.5	2.7	2.7	5.0	9.8	12.9	5.1	5.0	5.0
Bulky waste	1.4	76.0	84.8	81.5	80.0	83.7	85.5	80.8	95.0	95.0
Total level of reuse and recycling	0.3	5.5	10.6	10.7	11.9	15.5	18.4	39.8	55.5	60.5

The total mass of waste streams that must be collected separately to ensure the required recycling levels for 2025 and 2030 are very high, at 64.9% and 70.5% of the total mass of generated waste, respectively. It seems that such levels are unattainable in practice and separating the missing amounts of waste for recycling from residual mixed waste is impossible, as the quality of this waste will not allow it to be recycled, especially bio-waste, paper and cardboard (which are the fractions most relevant for Nr). The situation is similar with sewage sludge generated at the Zielona Góra Wastewater Treatment Plant, which is entirely used for natural or agricultural purposes outside of Zielona Góra.

5.3 Methods of recovering nitrogen from wastewater

The use of nitrogen recovery methods in a wastewater treatment plant (WWTP) means that nutrients are recycled within the system as fertilizer. This reduces energy consumption and minimizes eutrophication as well as greenhouse gas emissions.

Legal requirements on recovery of treated wastewater, first of all, focus on the water use. Reclaimed water from WWTP's thus can be used for agricultural irrigation, and for certain other purposes (like in industry) when health and environmental legislation are met. In terms of reintegrating nutrients, irrigation (fertigation) may be able to reduce mineral fertilizer needs. Still reusing effluents of a WWTP will have limited effects on Nr, as such plants optimize towards denitrification to remove Nr rather than use it. Several approaches may be considered more useful to specifically recover Nr:

- Struvite ($MgNH_4PO_4 \cdot 6H_2O$) may form spontaneously in WWTP's posing operational problems. Using fluidized bed reactors struvite can be successfully separated from the process as a directly usable product – a mineral fertilizer. High ammonia concentrations in the liquid phase help precipitation. Typical nutrient content in struvite are: 4.7-5.6% N (w/w), 28-29% P_2O_5 (w/w), <1.0% K_2O (w/w) (Siciliano et al., 2020).
- Ammonia recovery from wastewater and digestate liquids may use a membrane filtration system combined with an ion exchange process. NH_3-N recovery of 37.5% has been reported (Hui et al., 2017).
- Ammonia stripping in stripping towers requires application of high pH or elevated temperatures (up to 70°C) and is able to remove 55-77% of ammonia (Guštin et al., 2011; Kim et al., 2021). Recovery is in water (to yield an aqueous ammonia solution) or in acids (to result in concentrated ammonium sulfate or ammonium nitrate), products can be used as fertilizers or in industry.
- Using wastewater urea as a catalyst in water electrolysis for hydrogen generation, instead of pure water, may save ~16% energy (Geng et al., 2021; Logan et al., 2008).
- A microbial fuel cells obtains electricity by microbial decomposition of organic compounds while recovering some of the products of this decomposition (ammonium and nitrate nitrogen) can be recovered. At an output voltage of 600-700 mV and an external load of 500 Ω, 47% of NH_4^+ is recovered in the anode chamber and 83% NO_3^- in the cathode chamber (Xiao et al., 2016; Cheng et al., 2014; Dewan et al., 2009).

The most widely used method of nitrogen recovery from WWTPs is the application of sludge for fertilizer purposes. Protection of the environment, specifically of soils, needs to be central in all considerations and legal requirements (Directive 86/278/EEC on the protection of the environment, and in particular of the soil, when sewage sludge is used in agriculture) need to be observed. While in some EU countries, untreated sludge can be used in farming if it is injected or worked into the soil, there are also certain cases where sludge cannot be used at all (e.g. for fruit and vegetable crops). The goal of management is to use the valuable agronomic properties and fertilizer potential of municipal sewage sludge, i.e. the organic matter contained in it and plant nutrients such as nitrogen (N), phosphorus (P) and micronutrients. However, there are also potentially harmful and persistent compounds that have a potential to accumulate in soils, such as heavy metals or persistent organic pollutants. Such compounds may endanger soil's capability for food production.

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6. Integrating stakeholders: experience and practice

6.1 Specific activities involving stakeholder

An important aspect of the project is the applicability of the results in practice. For this reason, it is important to involve stakeholders and potential users at an early stage in the process in order to jointly develop practical approaches. In connection with the elaboration of nitrogen budgets a dialogue process with relevant stakeholders was started in the city of Vienna and the small town of Klagenfurt in Carinthia. In both cities / regions stakeholders have been identified during an intense stakeholder mapping process.

The first two workshops were held virtual in November 2020 using the communication platform Zoom and the virtual whiteboard Miro. At this stage of the project, primarily the background and objectives of the project as well as first project results were presented and discussed. In addition, it was important to identify the stakeholders' requirements for the project. Based on the results of two workshops, the next workshop series took place in 2021, on October 11 in Klagenfurt and on October 14 in Vienna. In addition to a review of the already further developed nitrogen budgets, approaches for practical implementation were primarily identified and further elaborated.

On May 31, 2022, all results have been compiled in an international conference in Vienna. To this event, the stakeholders who were already involved in the process during the previous workshops were invited in the first place. However, some new stakeholders also participated. In addition, scientists from the Expert Panel on nitrogen budgets under the UNECE Air Convention who had a meeting in Vienna the next day also took part. Especially this mixture of practitioners and scientists led to interesting results.

At the conference the promising outcomes of the UNCNET project were presented. In particular, the comparison between the Viennese nitrogen budget with those of Zielona Góra in Poland and the cities of Shijiazhuang and Beijing in China provided much material for discussion. In a fishbowl discussion, the project results were jointly analysed and discussed. In the process, existing approaches to practical implementation were further developed and new ideas and solutions were identified. In general, the stakeholders were very interested in the project and showed a high willingness to actively participate.

6.2 Stakeholder structure

In the course of the stakeholder mapping, attention was paid to a balanced distribution of stakeholders both thematically (agriculture, waste management, energy, mobility, etc.) and structurally (authorities, NGOs, companies, etc.). A total of 104 people participated in the stakeholder workshops with the following distribution among the different stakeholder groups: Authorities 22%, NGOs 12%, Research and science 46%, Companies 10%, Stakeholder organisations 7%, Urban planners 5%.

It can be seen that scientists and authorities in particular have the strongest connection to the topic of nitrogen and were therefore recruited in greater numbers for the stakeholder process. The rather low number (compared to the invitations) of companies that came to the workshops is striking. Only companies where nitrogen plays a role in the core business, such as the wastewater treatment plant operator of the City of Vienna or the fertilizer manufacturer Borealis are interested in the topic.

It is therefore also important to make other companies with a strong connection to nitrogen, for example in the area of energy production or mobility, more aware of the topic, for example by better highlighting problems and correlations. A business circle in cooperation with an economic interest group could be a suitable measure

6.3 Stakeholders' demands on the project

The majority of stakeholders were impressed by the idea and implementation of nitrogen budgets. Especially in the first two workshops (both in Klagenfurt and in Vienna), the stakeholders formulated their wishes and demands on the project, which could partly be considered in the project elaboration. Stakeholder and especially city authorities expected valuable inputs from nitrogen budgets in different areas of circular economy. For example, urban mining from sewage sludge is a major topic and is currently being implemented as part of a pilot project in Vienna. Perhaps the nitrogen balances can provide new inputs and impulses in the direction of N-recycling.

Data, facts and arguments gained by the project should also be useable for different existing tools and strategy processes like the city's greenhouse gas balance. Nitrogen balances allow a new level of observation. In this way, connections or problems can be recognized and analysed from another perspective. The need to focus not only on large metropolitan areas and cities was also emphasized. Findings for smaller cities and regions, which have other infrastructure. than Vienna would be appreciated.

Finally, stakeholders also want concrete recommendations, conclusions and proposals for action that can be derived from the nitrogen balances.

6.4 What practical approaches for nitrogen budgets were identified?

An important focus of the stakeholder engagement was the "reality check" of the nitrogen budgets. Therefore, the question should be answered: can nitrogen budgets be practically applied or support stakeholders in their daily work? Almost all stakeholders answer yes to this question, but to varying extent. According to Experts of the Austrian Federal Environment Agency nitrogen budgets could make a valuable contribution to biodiversity management. Information on N-release and N-input via air plays an important role for certain ecosystems, e.g., for critical loads of nitrogen oxides and ammonia. Input of N into ecosystems often shifts the species spectrum. For instance, in the Danube National Park, grassland is a priority-habitat very sensitive to nitrogen inputs.

In the implementation of nitrogen budgets, the regional approach is of great importance. For certain stakeholders in Austria one option could be to focus on existing structures like KEMs (climate and energy model regions) or KLARs (climate change adaptation regions) or e5 (energy regions).

Especially for public servants nitrogen budgets could also deliver meaningful information for the Austrian climate council especially in the field of waste management and circular economy. City authorities also highlight the Importance of nitrogen budgets for the food waste prevention action plan. Data and results would be underpinning the work of Vienna's municipal departments. For strategy development in other municipalities or cities, the further development into a user tool as a basis for the elaboration of measures or prioritization of them could be interesting.

Almost all Stakeholders mentioned the importance of awareness raising according to this topic. The nitrogen problem is hardly perceived as a threat potential by the population, but also by politicians. Although water eutrophication, overfertilization; ammonia emissions (source of particulate matter) have major negative effects on health and the environment. The main task here is to create awareness and sensitize people, for example through campaigns, public relations work or educational measures. Microplastics can serve as a successful example for raising awareness.

According to NGOs and agricultural Stakeholder Organizations the cooperation with agriculture is of high importance. Currently, there is already 25% organic farming in Austria. To raise awareness and to help agriculture to get out of its defensive position, it is extremely important to cooperate with the educational sector, e.g., with the University of Natural Resources and Life Sciences, Vienna and University of Applied Sciences in Wieselburg. Above all, research is challenged to offer solutions to agriculture, but also agriculture to understand its own activities in such a way that they become part of an environmentally sound solution.

Above all, scientists emphasized that currently the process of nitrogen extraction and utilization is a one-way street and leads to end-of-pipe solutions. It is therefore important to close the loops and to recycle or reuse reactive nitrogen. However, there are still many open questions and a great need for research.

6.5 Challenge of digital stakeholder engagement in times of Covid 19

Due to the Covid-19 epidemy, two stakeholder workshops were held online using the communication software Zoom and the Miro whiteboard. Also, the final stakeholder conference was held as a hybrid event. Generally, the involvement of stakeholders in form of online meetings and workshops works well, but also has its limits. Especially in a smaller context, when the participants come from one region or city, it makes sense to hold at least the kick-off event physically. The advantages, especially in terms of variety of methods and interpersonal interaction, clearly outweigh the disadvantages here.

Hybrid meetings are a good option for larger, supra-regional or international events. Ideally with a joint introductory part and differentiated interaction tools between the physical participants and the online community. At the end of the event, both groups can present their results to each other.

7. Methodology for urban N budgets

To allow a closer investigation and analysis of Nr flows through the urban and peri-urban environment and to facilitate the comparison between different cities, a framework was developed and implemented in a stock and flow model. The framework is based on the framework for the development of national N budgets (UNECE, 2013) but has been adapted to the urban environment. We identified 10 pools (waste, wastewater, households, urban animals, urban plants, import/export, combustion, industry, air, water) as important sources, sinks or places of N transformation in the urban and peri-urban environment. Subsystems were added to the two pools related to urban agriculture (urban plants and urban animals) to allow the display of more detailed processes (horticulture, urban greens and agricultural land for the urban plant pool as well as livestock and pets for the urban animal pool). A detailed description of the development can be found in Winiwarter et al. (2020).

The framework was implemented into a stock and flow model (https://www.stan2web.net) which allowed not only to show uncertainties related to certain flows but also enabled the calculation of unknown flows through an integrated balancing approach.

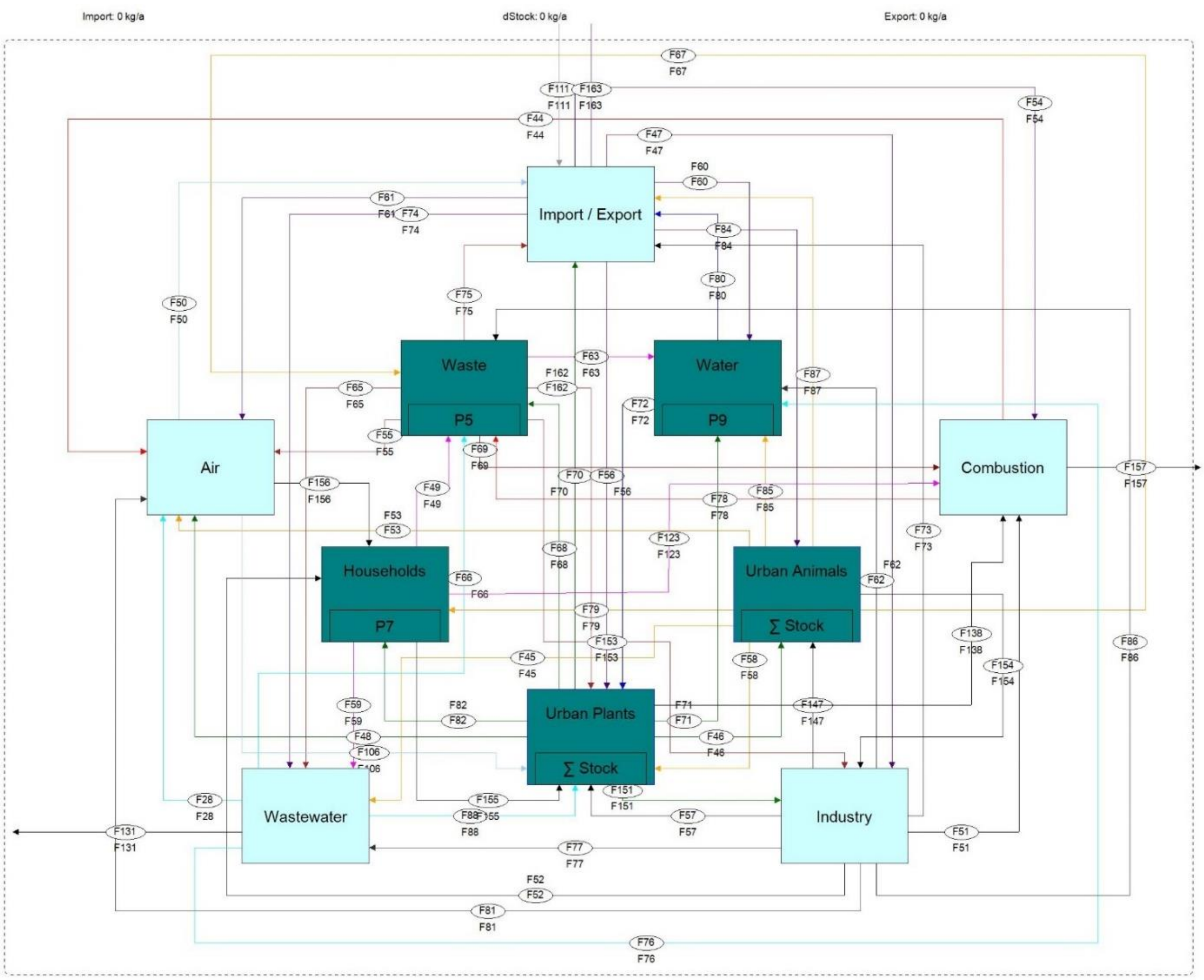

Figure 12. Framework of the urban N budget

Data collection for the calculation of the different flows was making use of the respective national/urban data in each test area. Other resources were the "guidance document on national nitrogen budgets" (Winiwarter & EPNB, 2016) for the European data and the CHANS and NUFER model for the Chinese test areas (Gu et al., 2015, Ma et al., 2019).

To simplify data collection and introduce 'check-points' for data compatibility, a concept for the separation of the urban N flows into two pathways was introduced. The two pathways are the 'Urban Agro-Food Chain' (Fig. 13) including urban agriculture, households, trade, wastewater and waste, and the 'Urban Combustion Chain' (Fig. 14) including industry, combustion and air. This division allowed more detailed discussions of the concept and its applicability as well as of the results and possible data contributions.

Slight differences occur between cities, despite efforts of harmonization, due to structural discrepancies and sometimes also due to data availability. In all cases, the agro-food chain is much more complex than the combustion chain, as the number of pools and flows between pools is considerably larger.

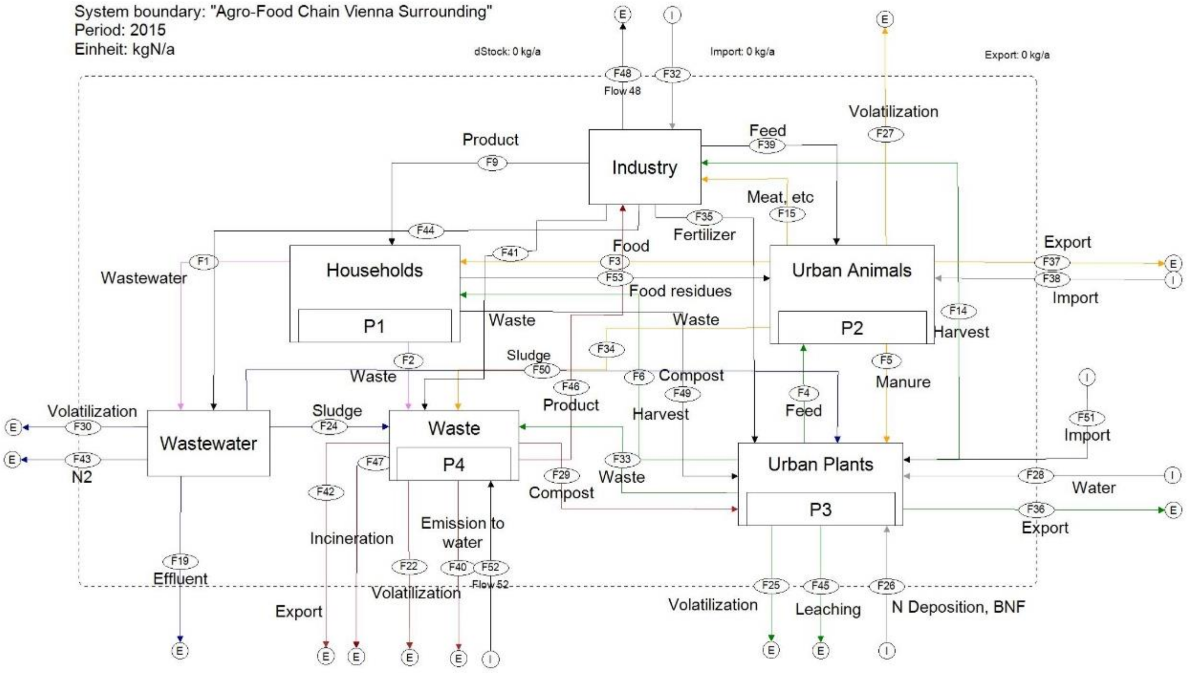

Figure 13. Implementation of the Agro-Food Chain for Vienna, peri-urban area. Visualization takes advantage of the STAN software for stock&flow analysis. Flows are depicted in oval shapes (imports/exports in circles), and rectangles stand for pools. Internal rectangles ('P1') show pools that allow stock changes.

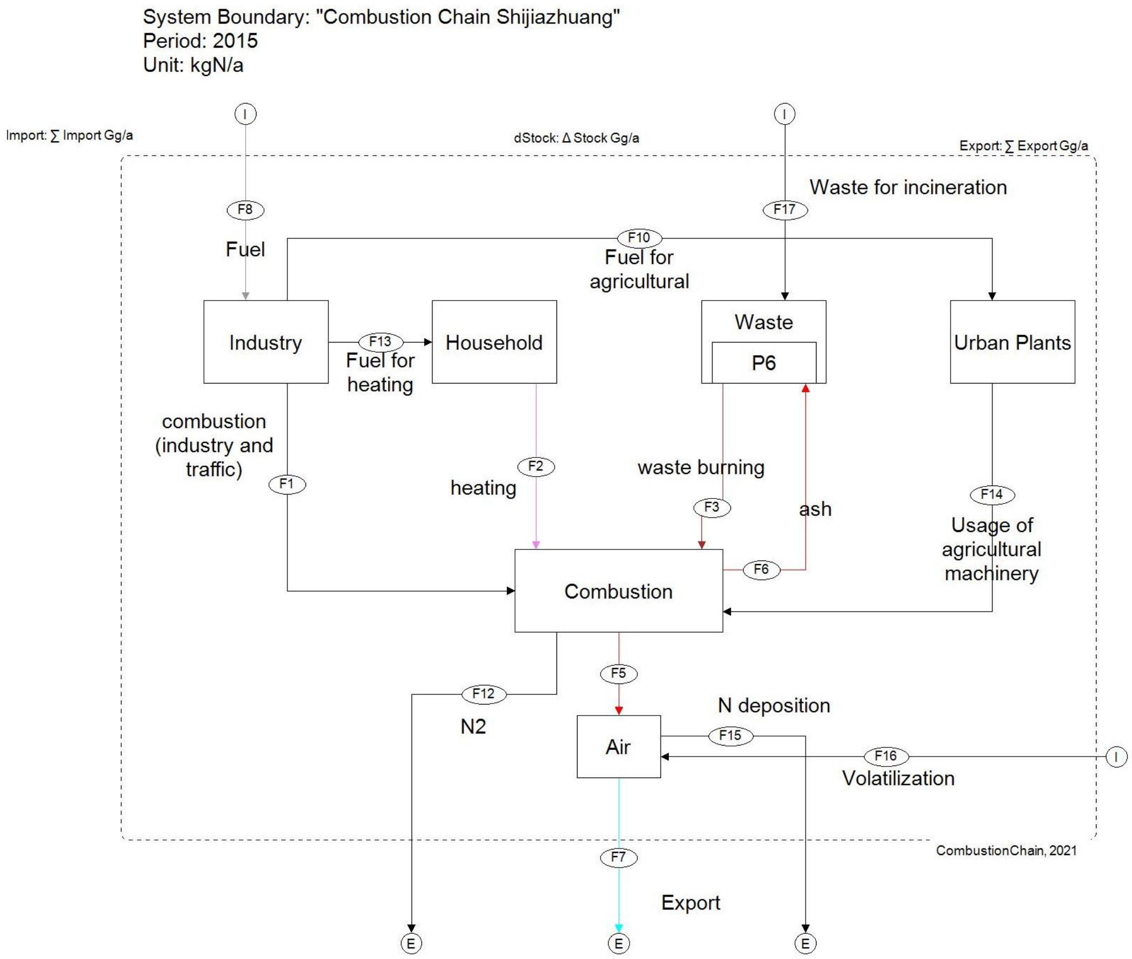

Figure 14. Implementation of the Combustion Chain for Shijiazhuang, city center. Visualization takes advantage of the STAN software for stock&flow analysis. Details as in Fig. 13.

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8. Comparison between cities – Differences and similarities

Once filled with data, the urban N budgets for the core and surrounding areas of Vienna, Zielona Góra, Shijiazhuang and Beijing were compared to discover common patterns, similarities and differences.

In general, urban as well as peri-urban areas are transformation sites for reactive nitrogen (Nr). Nr enters these areas as food, products, fuel, fertilizers, or livestock. These inputs are then transformed to products (e.g. food produced via fertilization, or meat from imported livestock) that are either consumed within the city, accumulated or exported as well as directly or subsequently turned to waste or emissions to water or air (including non-reactive N_2). However, it is possible to recycle Nr waste (=unintended by-products such as manure, sewage sludge or compost). Depending on the test areas, different patterns of these flows and transformations can be observed. Different approaches are applied to analyze the urban and peri-urban systems and discover similarities and differences between test areas. These approaches include:

- Analyzing the general flow pattern through the test areas
- Separating flows into the agro-food chain (encompassing flows related to households, waste, wastewater and agriculture) and the combustion chain (encompassing flows related to industry and combustion)
- Analyzing flow composition
- Analyzing emission composition

8.1 Flow patterns through the test areas

Differences in the Nr flow patterns can be observed between the urban and peri-urban region for all four cities (Vienna, Zielona Góra, Beijing and Shijiazhuang) with biggest flows more often being linked to human consumption or industrial production in the core areas and biggest flows linked to agricultural production in the surrounding areas. In the core area, the biggest share of Nr entering the area, mostly as consumer goods, is transformed through consumption or industrial production. It is then exported as product or emitted to the atmosphere in the form of N_2 from wastewater treatment or emission control in combustion processes but also as NO_x from combustion processes (Tab. 4, 5). In the surrounding area, the biggest share of Nr entering the area is more often found in synthetic fertilizers and transformed through agricultural production to goods that are exported as production exceeds local consumption. Subsequently, surrounding areas are more self-sufficient in plant and livestock-based food than core areas while also showing higher Nr accumulation in soils or water.

Table 4. Parameters & Indicators to describe urban N budget in Vienna and Zielona Góra

	Vienna Core	Vienna Surrounding	Zielona Gora Core	Zielona Gora Surrounding
General				
In [kgN]	19,339,227.17	58,199,522.22	1,914,082.52	1,028,181.90
Per area [kgN/ha]	466	139	328	47
Per person [kgN/cap]	11	89	16	52
Out [kgN]	4,070,165	45,797,759	1,432,926	637,427
Per area [kgN/ha]	98	109	246	29
Per person [kgN/cap]	2	70	12	32
Air (% of import)	20%	1%	6%	0%
N2 - sink (% of import)	57%	20%	28%	4%
Products Out (% of import)	1%	57%	31%	46%
Waste Out (% of import)	1%	0%	12%	12%
dStock (% of import)	22%	21%	23%	38%
Plant Stock (% of import)	3%	3%	4%	26%
Animals Stock (% of import)	2%	1%	5%	6%
Household Stock (% of import)	2%	4%	2%	4%
Waste Stock (% of import)	1%	0%	7%	0%
Industry Stock (% of import)				
Water (% of import)	15%	13%	4%	2%
Recycling (% of import)	4%	6%	0%	24%
Agri-Food Chain				
Self-sufficiency Plant Food	3%	317%	6%	74%
Self-sufficiency Livestock Products	0%	38%	0%	63%
Self-sufficiency Feed	728%	276%	0%	49%
NUE on agricultural land	55%	68%	70%	80%
N surplus [kgN/ha]	62	46	26	23
Emission and Deposition				
N deposition per hectare [kgN/ha]	17	13	16	17
Emission per hectare [kgN/ha]	110	16	35	1

Table 5. Parameters & Indicators to describe urban N budget in Shijiazhuang and Beijing

	Shijiazhuang Core	Shijiazhuang surrounding	Beijing Core	Beijing surrounding
General				
In [kgN]	23,857,053	1,145,480,661	150,100,311.72	308,539,380.10
Per area [kgN/ha]	589	811	1,083	201
Per person [kgN/cap]	18	162	18	59
Out [kgN]	16,222,869	253,714,827	31,235,020	91,064,559
Per area [kgN/ha]	401	180	225	59
Per person [kgN/cap]	12	36	4	18
Air (% of import)	10%	18%	12%	21%
N2 - sink (% of import)	18%	3%	47%	25%
Products Out (% of import)	42%	4%	0%	0%
Waste Out (% of import)	5%	0%	0%	0%
dStock (% of import)	25%	75%	42%	54%
Plant Stock (% of import)	8%	52%	3%	33%
Animals Stock (% of import)	0%	0%	0%	0%
Household Stock (% of import)	1%	5%	25%	7%
Waste Stock (% of import)	0%	0%	0%	0%
Industry Stock (% of import)		4%		
Water (% of import)	16%	15%	14%	14%
Recycling (% of import)	24%	17%	5%	7%
Agri-Food Chain				
Self-sufficiency Plant Food	59%	69%	9%	66%
Self-sufficiency Livestock Products	41%	84%	0%	65%
Self-sufficiency Feed	48%	88%	0%	49%
NUE on agricultural land	46%	19%	49%	82%
N surplus [kgN/ha]	191	961	62	20
Emission and Deposition				
N deposition per hectare [kgN/ha]	37	37	21	45
Emission per hectare [kgN/ha]	80	18	132	14

Structural differences between cities and their surrounding areas influence the Nr flow patterns. While the Vienna core area exports only 1% of incoming Nr, the Zielona Góra core area exports around 31% of incoming Nr due to a wood factory located within its boundaries. In the Shijiazhuang core area over 40% of Nr input is exported as product due to a high level of local industrial production (Fibre, wood fiberboard, furniture, tableware, textiles). In the European surrounding areas, most Nr is either exported as goods (agricultural goods like food or feed) or accumulates in the test area, often in water due to N leaching and runoff from agricultural land. In the Chinese peri-urban areas, only small shares of Nr are exported as product, most remains within the test area as stock change, often remaining in agricultural soils due to excessive fertilizer use (200-1000 kgN/ha).

Reducing Nr flows can be achieved in part by increases recycling – here it is relevant to understand in which situation Nr is being recycled, and benchmarking between cities can be valuable. Only 4% of Nr is recycled in Vienna core area (mostly compost from the waste pool) and no Nr is recycled in Zielona Góra. Also in the surrounding areas of these cities recycling remains low, with only 6% in the Vienna surrounding and 13% in Zielona Góra surrounding (mostly manure). In Beijing and its surround, the recycling rate is higher with 8%. Here human excreta are being collected and directly applied to agricultural land. This practice constitutes the biggest share of recycled Nr in the core area (55%), while manure N application is prime contributor to recycling in the surrounding area (67%). In Shijiazhuang, Nr recycling in the core area is the highest with over 30% of Nr input being recycled. Again, direct application of human excreta on agricultural land makes up the biggest share of Nr recycling (64%). In the surrounding area, 18% of inflowing Nr is being recycled, most of it being manure N (77%).

The extreme values of Nr recycling as displayed in Tab. 4 and 5 are specifically striking for Zielona Góra and Shijiazhuang core areas. Due to local specificities (i.e., no livestock, wastewater sludge application, advanced waste collection system, or composted waste in the area), we observe that Zielona Góra does not recycle any Nr in its core area (while some is exported outside where recycling takes place, but cannot be expressed in this system). As the other extreme, Shijiazhuang core area reaches the highest recycling rate (32%) of all test areas. This is notably due to the significant use of human excreta being directly collected from households and applied on agricultural land. This practice, a specificity of the Chinese test areas, contributes to the respective recycling rates to considerable extent.

8.2 Environmental Nr flow chains

Displaying flows separately as agro-food and combustion chain shows more clearly how flows cascade through the urban environment, aiding in the identification of emission sources. Here we focus on the agro-food chain of the Vienna core area (Fig. 15) to identify the most important features. Clearly the household pool as place of human consumption stands in the center of all flows. This feature, with some variation, is evident also for the other core areas. In the surrounding area, where agricultural activities become visible, the largest flows are associated with fertilizer application, or in the Chinese peri-urban areas, where animal husbandry is important, with manure application.

The combustion chain, also encompassing industry, tends to be determined by Nr content of fuels (no figure). During combustion and exhaust gas cleaning devices much of Nr is converted to molecular N_2, thus no direct link between Nr in fuels and in air pollution can be drawn. Fuels rich in Nr are waste in incinerators (relevant for Vienna core area), but also fuel oil and biofuels, where relevant. Air pollution from the core areas is mostly exported, as atmospheric residence times especially of NO_x are sufficiently large to facilitate transport across the system boundaries.

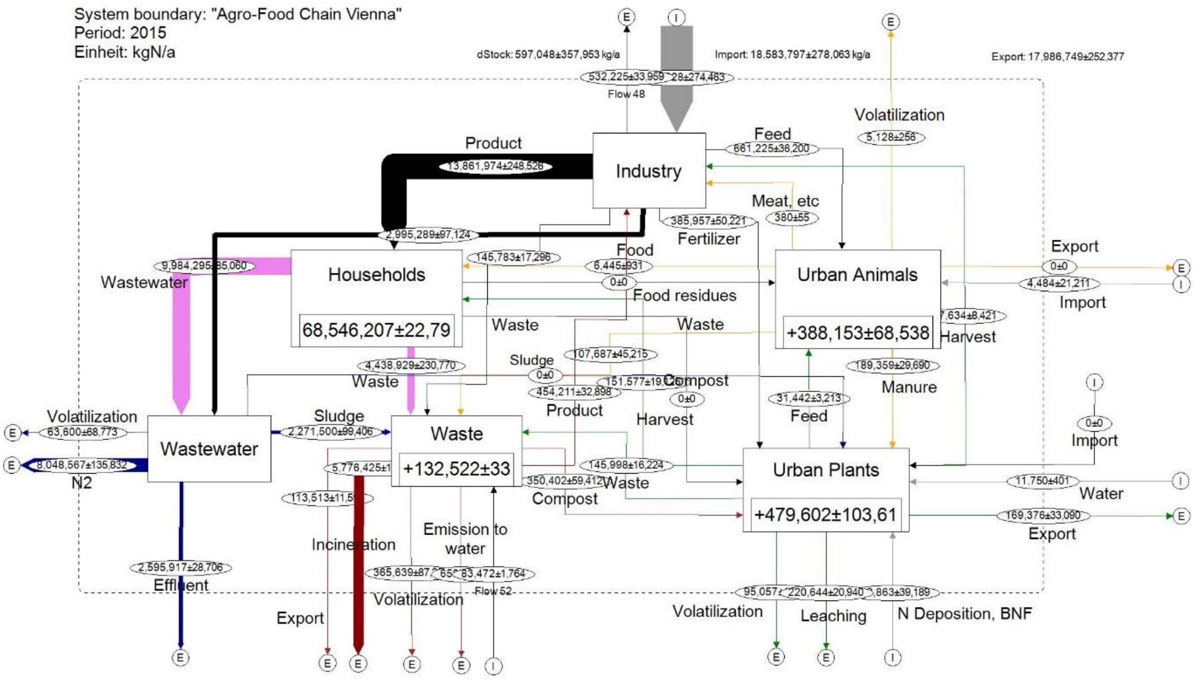

Figure 15. Agro-food chain Vienna core area

Differences between the urban and peri-urban area also appear when inspecting the composition of flows. While imported Nr to the urban area is mainly composed of consumer goods (mostly food), synthetic fertilizer import constitutes a large share in the surrounding (Fig. 16). In the Vienna core area, 55% of imported Nr can be found in food and 41% in non-food products directed to households. In the surrounding area, 39% of Nr import is fertilizer, while 24% of Nr import is directed to industry for fuels/fuel production as there is an oil refinery located in the Vienna surrounding area. In Zielona Góra, 51% of imported Nr can be found in fuels – mostly used for industrial production, 21% of Nr import is found in food as self-sufficiency is higher in this test area. Also in the surrounding area of Zielona Góra, synthetic fertilizer constitutes a larger share of Nr import. However, as Nr input to soils is comparably low in this area, the share of synthetic fertilizer remains lower compared to the Vienna surrounding area.

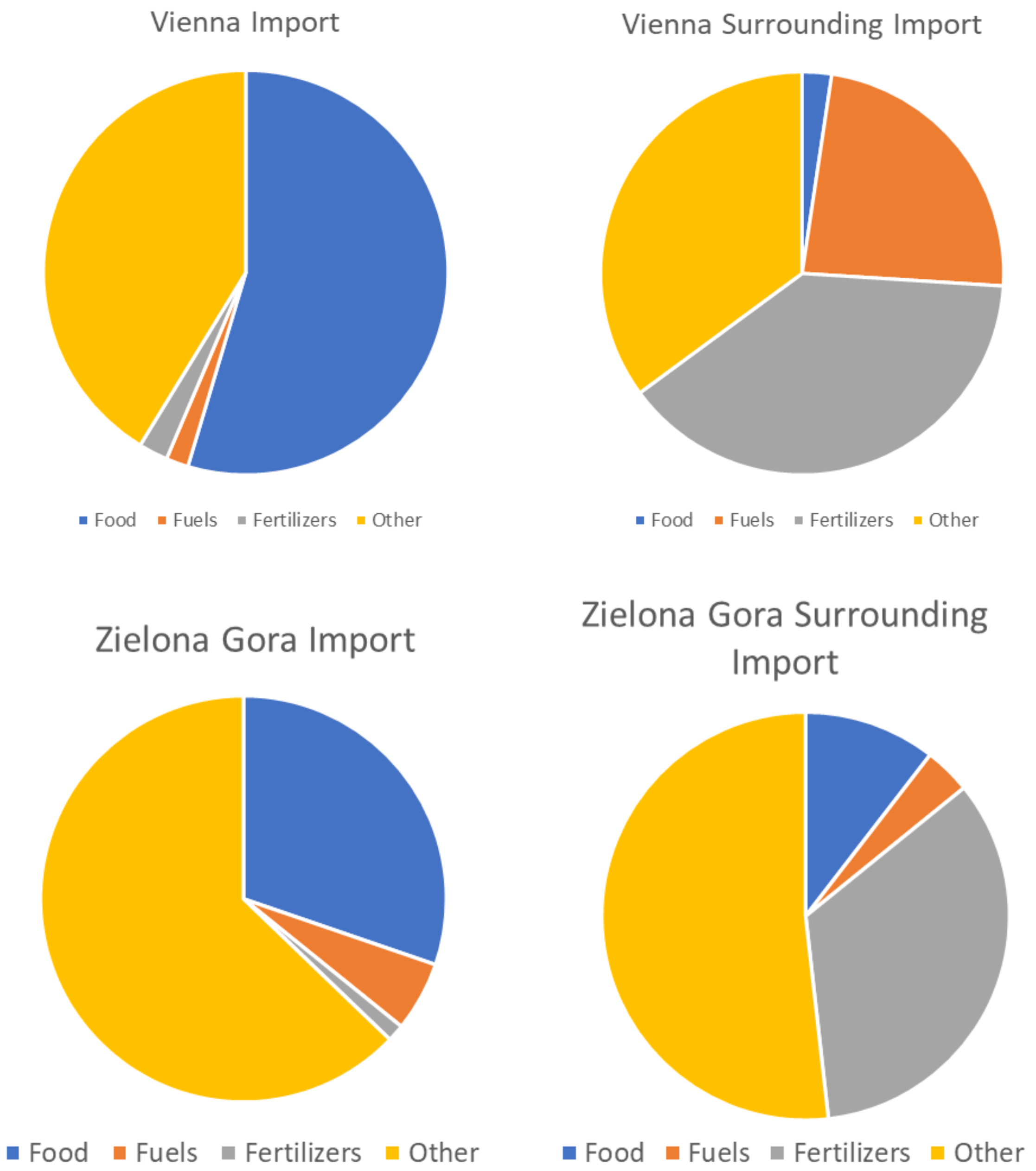

Figure 16. Composition of N import to test area for Vienna and Zielona Góra

In some cases, differences between the test areas not only just become visible when looking at certain flows' composition but also lie in their existence. Direct flows from household to livestock and agricultural land will only exist in the Chinese test areas where food residues are used as feed and human excreta is collected to be re-used on agricultural fields. In the Beijing core area, N input to agricultural land from human excreta has the highest share with over 70% of total N input, followed by Shijiazhuang core with over 50%. In the surroundings of the Chinese test areas, it plays a smaller role with 1%-3% of total Nr input.

8.3 Emissions to air and water

Human consumption and industrial production in the core area require wastewater treatment and combustion processes, respectively, which are the main processes causing the release of Nr to air and water. In the peri-urban areas, activities on agricultural land (leaching/run-off) and in animal husbandry add to or exceed these processes (Tab. 6). Although these patterns of emissions are largely similar in the test areas, differences in magnitude remain.

Leaching and run-off per area is up to four times higher in the Chinese compared to the European surrounding areas. This is the result of excessive fertilization. The same is true for volatilization per hectare, which is over 25 times higher in the Beijing surrounding area than in the Vienna surrounding area.

Table 6. Source shares of emissions to water and air per test area

	Vienna Core	Vienna Surrounding	Zielona Gora Core	Zielona Gora Surrounding	Shijiazhuang Core	Shijiazhuang surrounding	Beijing Core	Beijing surrounding
Emission Shares - Air								
From livestock	0%	6%	0%	23%	11%	14%	0%	16%
From agricultural land	1%	21%	1%	58%	9%	76%	6%	58%
From combustion	89%	52%	67%	15%	65%	6%	66%	13%
From waste	8%	11%	29%	0%	1%	4%	8%	1%
From wastewater	1%	0%	1%	0%	4%	0%	5%	0%
From urban greens	1%	10%	3%	5%	10%	0%	5%	7%
From horticulture	0%	0%	0%	0%	0%	0%	9%	5%
Emission shares - Water								
From wastewater	92%	11%	98%	0%	61%	2%	91%	6%
From agricultural land	5%	86%	1%	98%	36%	97%	5%	88%
From horticulture	0%	0%	0%	0%	0%	0%	2%	3%
From urban green	3%	3%	0%	2%	3%	0%	1%	3%
From waste	0%	0%	1%	0%	0%	0%	0%	0%

Differences in certain derived parameters can be traced back to the respective source patterns in the test areas. Cities can have different preference of livestock husbandry. E.g., while the overall number is quite small, the share of horses in total livestock for Vienna core area is high. These animals have fairly high Nr excretion, and in consequence high volatilization of Nr available in feed. Therefore, the share of NH_3 emissions from animal feed is high for Vienna, despite of not contributing much to air pollution. In Vienna surrounding, Nr leaching is high compared to other European test areas. However, in Beijing surrounding, it is even more than 6 times that high. But only for Vienna surrounding, an impact on groundwater nitrate levels (exceedance of threshold of 45 µg/m3) can be observed. A reason for this is the depth and size of the groundwater body. In the Chinese test areas, the groundwater bodies lie deep below the soil surface and are not yet affected by excessively high (up to 860 kg/ha) Nr input to agricultural soils.

8.4 Challenges and Potentials

While this analysis of flows and flow patterns helps identifying general patterns and analyzing differences between the test areas, it also shows where biggest flows leave the system and where they originate from as well as where they accumulate.

These patterns indicate challenges where Nr management and reduction is necessary, but also show potentials to increase Nr recycling in places it might otherwise be lost as N_2. An example for a challenge is the flow from agricultural land to water (leaching and run-off). Leaching can lead to increased NO_3 levels in groundwater, as observed in the Vienna surrounding, posing a threat to human health (WHO 2011; Feichtinger, 2013). While Nr flows from agricultural land in the Chinese peri-urban test areas do not flow to water but accumulate in soil, this could still affect groundwater quality in the future. Zhou et al. (2016) show that there is a potential future threat to groundwater from this high Nr accumulation in soils especially when considering increasing extreme rainfall events due to climate change. Additionally, this accumulation can disrupt the soil balance, can lead to higher Nr emissions and soil acidification (Velthof et al., 2011). Improved Nr management could improve these situations (Hansen et al., 2017). As was shown in Section 3, a combination of dietary change, reduced synthetic fertilizer application and a higher rate of synthetic fertilizer incorporation in soils could reduce N losses up to 63% compared with business as usual (BAU) in 2050.

In the European core areas, largest flows are related to wastewater treatment. While the biggest flow related to this process (N_2 to air) does not pose a challenge as it has no environmental effect, it has potential to reduce negative effects elsewhere. The re-use of sewage sludge could be seen as a potential to reduce emissions from synthetic fertilizer production when used as a substitute. It is estimated that around one ton of CO_2 is emitted for every ton of NH_3 produced using Haber-Bosch powered by natural gas as it is common practice in 90% of western European countries (UBA, n.d.). This, but also additional potentials become even more apparent when taking a closer look not only at the biggest flows but at recycling rates.

8.5 References

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9. Reducing pollution, increasing recycling rates: expectations and surprises

The previous sections of this report describe in much detail the flows of Nr compounds in the human compartments as well as in the environment and include information on possible impacts. The nitrogen flow model developed in UNCNET also allows to imply certain expectations on changed strategies, strategies that would allow for reducing Nr pollution and increasing recycling rates. Here we start out from best practice examples of comparing the test areas. Furthermore, we address potentially significant pathways for Nutrient Reuse and Recovery (NRR). Such key actions for enhanced NRR often advocated in literature (Sutton et al., 2013; Buckwell and Nadeu, 2016). Within the framework of UNCNET, we will discuss (i) the recycling of human excreta and/or urine to agricultural fields (with a focus on European test areas), (ii) the recovery of nutrients from wastewater and sewage sludge streams, and (iii) the recovery of the municipal waste organic fraction.

9.1 Comparing city information on recycling rates

Recycling takes an important role in improving the environmental performance of Nr compounds, because (i) every ton Nr recycled is a ton that cannot be released to the environment, and (ii) recycled Nr may replace Nr that otherwise needs to be fixed using large amounts of energy. Hence, collecting information on recycling is quite significant.

As data in Tab. 4 and 5, and the comparisons provided in section 8.1 show, recycling rates can be quite different in the respective areas. The low recycling rates of Zielona Góra core area, attributed to the limited experience with separate collection of waste fraction, and the high recycling rate of Chinese core city areas, assumed to be a consequence of reusing human excreta as plant nutrients, provide first explanations.

This pattern may be refined by identifying specific flows that contribute to recycling. Tab. 7 provides the share of seven specific materials (representing flows) to the total amount of recycling in each of the test areas. The urban core of Zielona Góra is missing as no recycling takes place.

Table 7. Recycling rate source apportionment per test area (% of total recycling rate) and linkages with Urban N Budget. Legend: VIE = Vienna; ZG = Zielona Góra; SJZ = Shijiazhuang; BEI = Beijing. The symbol “+” refers to the surrounding of the concerned test area.

Source	VIE	VIE+	ZG+	SJZ	SJZ+	BEI	BEI+	N Flow in budget
Manure (%)	2	61	100	32	77	1	67	Livestock to Agricultural land
Industrial waste (%)	55	8		1	2	43	16	Waste to Industry
Straw residues (%)				1	2	0.1	6	Waste to Agricultural land
Human excreta (%)				64	18	55	10	Households to Agricultural land
Kitchen residues (%)				2	2			Households to Livestock and Households to Pets
Sewage sludge (%)		5						Wastewater to Agricultural land
Compost (%)	43	26						Waste to Agricultural land and Waste to Urban greens

We note that manure application is an important part of recycling everywhere, but it constitutes the primary contribution to recycling rates in all four test areas' surroundings, ranging from 61% up to 100% in Vienna and Zielona Góra, respectively. Trends in core areas are more contrasted: while the application of human excreta on agricultural fields is predominant in Chinese test areas, Vienna essentially recycles waste. The other sources (straw and kitchen residues, and sewage sludge) represent minor contributions (1-7%) to the recycling rates of the respective areas. However, we observe that potentials exist to improve sustainability of N management, reuse and recovery, as discussed in the next Section.

9.2 NRR via the use of human excreta and urine recycling

Human excreta reuse as manure in agricultural practices has a long-standing and geographically varied history (Ferguson, 2014; Svirejeva-Hopkins et al., 2011). In South-East Asia and China specifically, human excreta derived fertilizer (HEDF) has withstood the era of chemically synthesized fertilizers and remains widely used (Liu et al., 2014). If such practices were still adopted in some of the European territories until the last century, they have now been fully replaced by modern wastewater treatment plants, mainly due to newly-made nutrient availability (mineral phosphates, the Haber-Bosch process for nitrogen, potash mines) and increased urban population (Esculier & Barles, 2019; Svirejeva-Hopkins et al., 2011).

If comprehensive guidelines on the adequate use of human excreta in agriculture now allow reducing the risk of pathogen contamination and enhancing nutrient recovery (Jönsson et al., 2004), social acceptance and higher ammonia volatilization than compared to Wastewater Treatment Plants (WWTP) remain obstacles to its technological expansion. (Moya et al., 2019; Spångberg et al., 2014)

In the ecological sanitation – or *ecosan* – field, the use of human urine as fertilizer for crop production is additionally being investigated to close nutrient loops. Containing the largest proportion of N (90%), P (50-65%), and K (50-80%) of households blackwater, as well as fewer enteric microorganisms and a reduced risk to human health in comparison to feces, urine offers a promising alternative and/or complement to HEDF and conventional WWTP. (Hilton et al., 2021; Rose et al., 2015) Specifically, urine diversion and source separation, whether that is through novel decentralized systems (Kavvada et al., 2017) or nutrient recovery on-site within the individual toilets (Wald, 2022) are key aspects of new design concepts (Randall & Naidoo, 2018). Among other uses of urine along an enhanced NRR, struvite precipitation in fluidized bed reactors with P and K recovery (Wilsenach et al., 2007), use of microbial fuel cells (MFCs) to generate electricity (You et al., 2016) or NH_3 recovery following urea hydrolysis through electrochemical stripping (Tarpeh et al., 2018) or ion-exchange (Tarpeh et al., 2017) constitute recent noteworthy technological advances.

Considering a urinary N excretion rate of 4 $kg.cap^{-1}.year^{-1}$ (Jönsson et al., 2004; Rose et al., 2015), the sole N recovery from urine as fertilizer using the above technologies would fully replace – excluding N losses of the respective recovery systems – the N mineral fertilizer use in Zielona Góra core and surrounding areas after redistribution of the N surplus in the core to the surrounding (Tab. 8). Similarly, such recovery would fully replace the N mineral fertilizer requirements in Vienna core area and about 42% of the N mineral fertilizer need in the surrounding area following N surplus redistribution.

Table 8. Gross estimates of recycled amounts of N from urine in Vienna and Zielona Góra (kt N.year^{-1}) and their percentages compared to mineral fertilizer requirements before and after N surplus redistribution. Legend: VIE = Vienna; VIE+ = Vienna surrounding ; ZG = Zielona Góra; ZG+ = Zielona Góra surrounding.

	VIE	VIE+	ZG	ZG+
N potential (kt N.year^{-1})	7.31	2.61	0.44	0.11
% Mineral fertilizer use	1828%	12%	1426%	26%
% Mineral fertilizer use after redistribution	100%	42%	100%	124%

As part of the European Green Deal, the European Commission (EC) implemented the Farm-to-Fork strategy striving for the reduction of nutrient losses to the environment by at least 50%, while ensuring no deterioration in soil fertility, while mineral fertilizer use should decrease by at least 20% by 2030. This would imply the joint development between EC and Member States of an Integrated Nutrient Management Action Plan specifically aiming to stimulate the markets for recovered nutrients and to address nutrient pollution at source, building on the framework on the new Common Agricultural Policy. In spite of such commitments, an integrated EU nutrient directive regulating the agricultural use of both N and P, including the potential use of human waste, is amiss. The latter would a.o. recognize regional differences between nutrient loads and the optimal balance between elements for an improved nutrient stewardship (Wassen et al., 2022).

9.3 NRR via reclaiming wastewater and sewage sludge

Estimates indicate that annually 2.3-3.1 Gg N sewage sludge in the EU27 is generated (Leip et al., 2014; Sutton et al., 2011) which also contain 0.23 Gg P (van Dijk et al., 2016). This is, respectively, representing up to 50% and 90% of N and P inflows to WWTP operating enhanced biological nutrient removal (Van Drecht et al., 2009).

Following treatment in WWTPs aiming to reduce water content and increase stability, the main disposal routes for sewage sludge include sanitary landfills, incineration, composting, or land-based applications, including structural soil improvement and soil amendment. The latter practices are framed according to various EU thresholds and directives. On the one hand, the EU Waste Framework Directive (91/156/EEC amending 75/442/EEC on waste) regulates the disposal of incinerated, composted, or landfilled sludge. On the other hand, both the Sludge Directive (86/278/EEC) and Nitrates Directive (91/676/EEC) aim to set minimum quality standards for the soil and sludge used in agriculture and prevent agricultural nitrate pollution in water, respectively. The overall sludge treatment in WWTPs is underpinned by the Urban Wastewater Treatment Directive (91/271/EEC) (European Commission, 2001). (Emerging) harmful substances in sludge such as heavy metals, Persistent Organic Pollutants (POPs), or pathogens remain a significant concern for land-based applications and a source of requests by some Member States for setting more stringent limits on their allowed concentration. Yet, recent work by Lamastra et al. (2018) indicated that the proposed EU limits are cautious enough to limit the deleterious impact on soil fauna.

The fate of sludge widely varies across the EU Member States. On average, about 40% is applied as a soil organic amendment on agricultural land (Milieu et al., 2010). In UNCNET, the disposal routes also differ greatly. While NRR from sewage sludge in European areas is more widely adopted, Chinese test areas fare worst by integrally disposing of sludge in sanitary landfills. In Zielona Góra, the sewage sludge is treated (aerobic mineralization or fermentation, followed by hygienisation with lime) in the city and then applied on agricultural land outside the city borders following its export. Vienna surrounding area applies a smaller fraction (16%) of its sewage sludge on agricultural land (0.2 ktN), equivalent to about 5% of its N recycling rate (Tab. 7) or 1% of its mineral fertilizer use. The remaining 84% are divided into composting (74%), thermal treatment (8%), and landfilling (2%). Vienna core area incinerates its sewage sludge, thereby recovering energy.

Other promising technologies for NRR from wastewater streams identified by Van der Hoek et al. (2018) include struvite precipitation, air treatment from thermal sludge drying, and digester reject water treatment through membrane filtration devices or air stripping (see also section 5.3 above). The latter achieve 1.1%, 2.1%, 20%, and 24% of N recovery from the WWTP system's total N inflow, respectively, whereas sludge reuse and urine treatment yield about 11% and 60%, respectively. While struvite precipitation and thermal sludge drying are mature technologies already applied in practice, others performing better may require additional effort before implementation, such as higher N concentrations in the digester reject water (membranes) or new/improved sanitation infrastructure (separate urine collection).

Maximum NRR from wastewater streams is attained when introducing alternatives in parallel throughout the WWTP process with sludge reuse and N recovery from urine, as well as from digester reject water. In a best-case scenario, the application of 90% of sewage sludge on agricultural lands (the highest percentage among the Member States, that is, in Portugal: Milieu et al., 2010) would suffice to fully replace the N mineral fertilizer requirements in Vienna and Zielona Góra core area, as well as about 12% and 28% of the respective needs of the surrounding areas, following N surplus redistribution. Due to both smaller quantities of sludge produced and larger quantities of mineral fertilizer applied on fields, Chinese test areas would profit less from that NRR pathway (Tab. 9).

Table 9. Gross estimates of recovered amounts of N from sewage sludge in all test areas (kt N.year^{-1}) and their percentages compared to mineral fertilizer requirements before and after N surplus redistribution. Legend: VIE = Vienna; VIE+ = Vienna surrounding; ZG = Zielona Góra; ZG+ = Zielona Góra surrounding; SJZ = Shijiazhuang; SJZ+ = Shijiazhuang surrounding; BEI = Beijing; BEI+ = Beijing surrounding.

	VIE	VIE+	ZG	ZG+	SJZ	SJZ+	BEI	BEI+
N potential (kt N.year^{-1})	2.04	0.94	0.15	NA	4.71	0.55	1.50	0
% Mineral fertilizer use	511	4	445	NA	32	0	2	0
% Mineral fertilizer use after redistribution	100	12	100	28	32	0	2	0

9.4 NRR from food and garden waste

NRR of bio-waste from Municipal Solid Waste (MSW) represents a third significant N potential pathway identified within the framework of UNCNET. Including green (garden and park) waste, household waste, food waste from food service (e.g., restaurants, caterers), and retail, the MSW organic fraction in the EU amounts on average to 37% of total MSW or about 88 Mt/yr (Saveyn & Eder, 2013). Despite the large quantities produced, recovery from such waste streams is typically more challenging, owing to the reduced nutrient concentrations and the more heterogeneous waste composition. MSW organic fractions in Zielona Góra and Vienna core and surrounding area amount to about 24%, 40%, and 37%, respectively (Tab. 10).

Bio-waste management is covered in the EU by the Landfill Directive (1999/31/EC) and the Waste Framework Directive (2008/98/EC), aiming to reduce landfilling progressively and develop sustainable waste management policies protecting the environment and human health, respectively. Additionally, specific EU policies, such as the Bioeconomy Strategy (European Commission, 2018) or Circular Economy Action Plan, advocate for the need to move towards a circular economy. If the Member States are requested to optimize the municipal organic waste treatment according to the *waste hierarchy* principle, they are in principle not yet (starting in 2023) legally bound to separately collect and recover/recycle it (European Commission, 2010). This consequently led to widely distinctive management strategies developed across Member States.

Table 10. Gross estimates of recovered amounts of N from composting in all test areas (kt $N.year^{-1}$) and their percentages compared to mineral fertilizer requirements before and after N surplus redistribution. Legend: VIE = Vienna; VIE+ = Vienna surrounding; ZG = Zielona Góra; ZG+ = Zielona Góra surrounding; SJZ = Shijiazhuang; SJZ+ = Shijiazhuang surrounding; BEI = Beijing; BEI+ = Beijing surrounding.

	VIE	VIE+	ZG	ZG+	SJZ	SJZ+	BEI	BEI+
MSW Biodegradable fraction (%)	40	37	24	NA	NA	NA	NA	NA
Composted or reused organic waste fraction (%)	38	87	0	0	17	0	3	3
N potential (kt $N.year^{-1}$)	0.32	0.005	0.026	NA	0.17	0.77	3.82	2.36
% Mineral fertilizer use	79	0.02	77	NA	23	<0.01	614	1
% Mineral fertilizer use after redistribution	79	0.02	77	NA	23	<0.01	100	3

Such management options combine primary bio-waste disposal routes that are landfilling, incineration, materials recycling, and nutrient recovery through Anaerobic Digestion (AD) and/or composting. Specifically, NRR from the latter 2 pathways comprise about 30% on average in the EU of the MSW organic fraction (Corrado et al., 2020). The composting option is already widely used in Vienna test area where bio- and green fractions of MSW are almost integrally (> 95%) composted. While the other test areas do not compost organic waste, Beijing area and Shijiazhuang core area however recover 3% and 22% of their household food waste as livestock and pet food, respectively (Tab. 10). Assuming a composting yield from organic waste of 37% (Buckwell & Nadeu, 2016) and a best-case scenario where 90% of biodegradable waste gets composted or reused in all test areas would integrally cover the mineral fertilizer need of Beijing core area, as well as more than 75% of those of European core areas (Tab. 10). Potentials for surrounding areas are globally much lower given the comparatively larger requirements of mineral fertilizer to be applied on fields.

9.5 References

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10. List of UNCNET deliverables

The results presented in this report are based on extensive material that has been prepared in the UNCNET project. Most of this material is publicly available on the UNCNET web site (www.uncnet.org) for download, with few exceptions (for reasons of data privacy or to avoid prepublication issues when planning to submit part of the material for peer review). The list below identifies all of the deliverables.

D1/1: Kick-off meeting report including project inception report (not publicly available).

D1/2: Report from the first project plenary.

D1/3: Report from the second project plenary.

D1/4: Report from third project plenary & stakeholder event.

D2/1: Draft concept of urban nitrogen flows.

D2/2: Final concept of urban nitrogen flows including uncertainty considerations.

Erratum D2/2: Correction of the final concept of urban nitrogen flows including uncertainty considerations.

D2/3: Using probability approaches to inform, revise and improve contributions on the respective nitrogen flows.

D3/1: Estimates of ammonia emissions from urban agricultural activities.

D3/2: A quantitative estimate of the impacts of ammonia emissions on urban PM2.5 air quality.

D3/3: A quantitative estimate of the impacts of different urban agriculture mitigation pathways on PM2.5 air pollution (not publicly available).

D4/1: Development of high-resolution N inputs and irrigation datasets from agricultural soils.

D4/2: Land surface modeling simulation of N leaching and health effects assessment (not publicly available).

D4/3: Optimization of urban agriculture management to mitigate groundwater N pollution under different climate changes.

D5/1: Draft concept of urban agricultural nitrogen flows.

D5/2: Clear concept of urban agricultural nitrogen flows.

D5/3: Final urban agricultural N flows with uncertainties and relationship with urban sustainable development goals.

D6/1: General guidance to quantify N in waste and waste water.

D6/2: Waste uncertainty.

D6/3: Extrapolation of methods on waste to human dwellings/constructions and circular economy concepts.

D7/1: Conceptual nitrogen budget.

D7/2: Final urban nitrogen budget.

D7/3: Report on comparison between cities and alternative nitrogen flows.

D8/1a: Report from stakeholder workshops I.

D8/1b: Report from stakeholder workshops II.

D8/2, D8/3, D8/4: Combined report on the Final Stakeholder workshop.

D9/1: Dissemination concept.

D9/2: Policy considerations of nitrogen budgets (not publicly available)

D9/3: Outreach report and summary.

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