Science of the Total Environment 902 (2023) 166238

Contents lists available at ScienceDirect

Science of the Total Environment

journal homepage: www.elsevier.com/locate/scitotenv

Anthropogenic effects on global soil nitrogen pools

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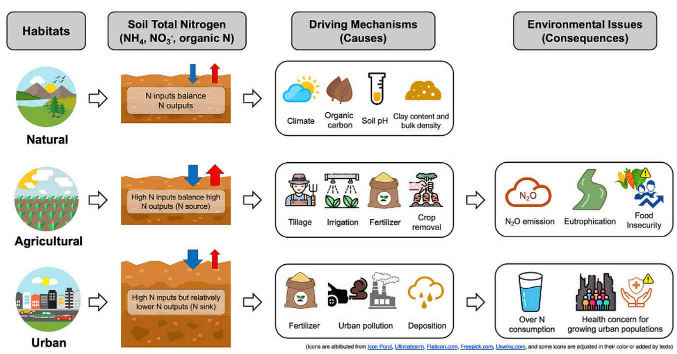
HIGHLIGHTS

- The nitrogen pools of agricultural and natural soils are similar in size.
- The nitrogen pool of urban soils is larger than natural soils.
- Mechanisms controlling soil nitrogen pool size vary with human use.

GRAPHICAL ABSTRACT



ARTICLE INFO

Editor: Dr. Elena Paoletti

Keywords:
Soil
Nitrogen cycling
Agricultural system
Urban system
Global change

ABSTRACT

The amount of nitrogen stored in terrestrial soils, its “nitrogen pool”, moderates biogeochemical cycling affecting primary productivity, nitrogen pollution and even carbon budgets. The soil nitrogen pools and the transformation of nitrogen forms within them are heavily influenced by environmental factors including anthropogenic activities. However, our understanding of the global distribution of soil nitrogen with respect to anthropogenic activity and human land use remains unclear. We constructed a meta-analysis from a global sampling, in which we compare soil total nitrogen pools and the driving mechanisms affecting each pool across three major classifications of human land use: natural, agricultural, and urban. Although the size of the nitrogen pool can be similar across natural, agricultural and urban soils, the ecological and human associated drivers vary. Specifically, the drivers within agricultural and urban soils as opposed to natural soils are more complex and often decoupled from climatic and soil factors. This suggests that the nitrogen pools of those soils may be co-moderated by other factors not included in our analyses, like human activities. Our analysis supports the notion that agricultural soils act as a nitrogen source while urban soils as a nitrogen sink and informs a modern understanding of the fates and distributions of anthropogenic nitrogen in natural, agricultural, and urban soils.

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https://doi.org/10.1016/j.scitotenv.2023.166238
Received 2 May 2023; Received in revised form 7 August 2023; Accepted 9 August 2023
Available online 15 August 2023

1. Introduction

Soil total nitrogen (STN), comprising both organic and inorganic forms (Bingham and Cotrufo, 2016), is determined by the balance between all nitrogen inputs and outputs to and from soil and often moderated by activity in the rhizosphere (Marty et al., 2017; Liu et al., 2018). The outcome of soil nitrogen cycling is globally consequential including effects on: primary production (Du and de Vries, 2018; Kicklighter et al., 2019) that provides food and energy supporting an ever-growing human population (Ladha et al., 2016; Kopittke et al., 2019), eutrophication that affects aquatic ecosystem health (Kaltenegger and Winiwarter, 2020), and the global carbon budget that regulates climate systems (Zhou et al., 2017). Naturally, without anthropogenic influence, soil nitrogen may be controlled by climatic conditions (*e.g.* temperature and precipitation) (Fagodiya et al., 2017; Chen et al., 2017; Li et al., 2020; Bytnerowicz et al., 2022), plant uptake and rhizosphere effects (Dotaniya and Meena, 2015; Dubey et al., 2021), microbial transformations (Kuypers et al., 2018; Myrold, 2021), and soil properties (*e.g.* aeration, organic carbon and pH) (Li et al., 2017; Neina, 2019; Zhou et al., 2019; Wang et al., 2020).

Human activities can be a significant factor altering soil nitrogen pools and cycling in anthropogenic soils (Fowler et al., 2015; Melillo, 2021). Indeed, anthropogenic nitrogen inputs have altered the global nitrogen balance over the last hundred years. This has consequences for soil and ecosystem health, including inefficient agricultural production (Khan et al., 2017), soil acidification (Raza et al., 2020), contaminated waterways (Ghaly and Ramakrishnan, 2015) and degraded ecosystems (Tully et al., 2015; Kopittke et al., 2019). All have been blamed on improper nitrogen management.

Most anthropogenic influences on the STN pool moderate nitrogen inputs and outputs (see Box 1).[1] For example, in agroecosystems, nitrogen fertilization and crop removal represent an additional but substantial and artificial soil nitrogen input and output (Sainju, 2017). Some agricultural practices have worked to improve nitrogen use efficiency and employ rotation with nitrogen fixing crops (Hossain et al., 2016; Yong et al., 2018) that can bring more nitrogen into soil environments, while others till or irrigate such that they trigger higher soil nitrogen loss through leaching (Bender and van der Heijden, 2015; Xu et al., 2016) and nitrous oxide (N_2O) emission to the atmosphere (Oikawa et al., 2015). The outcome being that agricultural soils may not retain the increased nitrogen for long since increased nitrogen outputs compensate for much of those additional nitrogen inputs. On the other hand, urban soils, that are also influenced by anthropogenic activities, can surprisingly accumulate more nitrogen in response to human associated nitrogen inputs, as has been seen across several large cities (Trammell et al., 2020).

Considering the growing human population (Crist et al., 2017), urbanization and food and energy demands (Ahmed et al., 2020), an accurate understanding of the STN distribution with respect to human influences across the globe will inform future ecological challenges and offer policy makers and managers a reference to develop mitigation strategies. We conducted a meta-analysis compiling the pools of STN across natural, agricultural and urban habitats from a globally distributed sampling of previous published works. Our analyses indicate that anthropogenic influence affects not only how much soil nitrogen is stored in that pool, but the mechanisms that moderate that pool size. The amount of nitrogen stored in anthropogenic *versus* natural soils varies with human use and has profound implications for sustainable agriculture and pollution remediation.

2. Materials and methods

2.1. Literature search and data collection

The goal of this work was to construct a meta-analysis quantifying the soil total nitrogen (STN) pool from a global distribution of natural, urban and agricultural soils. We conducted an extensive literature survey using Google Scholar from June 2021 to September 2021. We conducted the same literature review in Web of Science and Scopus. Results of all searches were similar, since Google Scholar provided more references, we used that output here. We selected published papers and works that met the following criteria: (1) the papers provided sufficient soil data which must include soil nitrogen, and (2) the publication year ranged from 2000 to 2021 to better reflect current soil nitrogen trends, and to ensure that we accumulated sufficient data sets from relatively limited urban studies for comparison. The searching keywords were [soil] and [nitrogen OR total nitrogen OR nitrogen pool OR nitrogen stock OR nitrogen storage OR nitrogen density OR nitrogen concentration] and [natural OR nature OR native OR agriculture OR agricultural OR farm OR farmland OR garden OR orchard OR urban OR city OR municipal OR metropolitan]. In total, we conducted 91 searches, and in each search, we collected the papers that met the above criteria. Within each query, we selected publications from the top 10 pages of results.

After paper identification and filtration (see Prisma Flow Diagram in Supplemental Fig. 1), in total, we collected the data from 204 papers. Because many papers conducted their research across multiple geographic locations or experiment treatments, we accumulated 913 data sets in total. The data extracted from publications represent a global distribution including all continents except Antarctica (see Fig. 1 in the Results section). The data used in this research can be found on the website of the *Knowledge Network for Biocomplexity via* https://knb.ecoinformatics.org/ upon publication.

From each paper, we recorded the data for climatic information and soil properties including: study location, mean annual temperature (MAT), mean annual precipitation (MAP), STN, soil organic carbon (SOC), soil carbon to nitrogen ratio, soil pH, soil bulk density (BD), and soil clay content. We collected the data of the soil profiles to 30 cm depth to account for shallow urban soils. If some soil data were missing in a paper, we referred to the global soil mapping data, in 30 cm depth, from Zhao et al. (2019) to acquire an approximate value. Most of the papers we included provided complete soil information, and we only included papers that describe soil nitrogen content in the forms of concentration (%) or pool, storage and stocks (g/cm^2, kg/m^2, Mg/ha, etc).

2.2. Criteria for defining habitat type

We compared the size of the nitrogen pool among natural, agricultural, and urban soils, and we used frequently used keywords to divide each data set into one of the three classifications. In each collected paper, the keywords would be found from its text, tables, or figures, while mostly from its method section or study site description. The keywords for defining a study location as an agricultural habitat were: farm, farmland, crop, cropland, harvest, fruit, vegetable, fertilizer, compost, tillage, agriculture, and agroecosystem. The study location was defined as urban if it is located in the range of a city, and the keywords for identifying a study location as an urban environment were: urban, city, civic, fertilizer, commercial, residential, and a city name. A few data sets were retrieved from studies in urban farms, and we categorized these data sets into agricultural habitats. Because fertilizer was a common keyword for identifying agricultural and urban habitats, we read carefully through the text of all papers with "fertilizer" as a key word to make the best habitat classification (agricultural or urban). Eventually, for those papers whose data was retrieved from neither urban nor

[1] In this box and box figure, please note that we do not fully incorporate the effect of animal manure on the soil N fluxes and STN pool in agroecosystems given that our cited papers including livestock farming only account for 4 %–5 % of our total agricultural datasets, but we acknowledge the application of livestock manure may alter the soil nitrogen pool and cycling, like increasing N_2O emission (Davidson, 2009) and influencing ammonia volatilization (Meisinger and Jokela, 2000; Smith et al., 2007).

agricultural land, we defined them as natural habitats. The keywords for identifying a study location as a natural habitat were: natural, intact, pristine, and undisturbed. However, if there was no keyword for habitat identification in a paper, we utilized the satellite function of Google Maps to decide the most appropriate habitat type for its study location.

2.3. Data processing in STN and other variables

In our analysis, we defined the STN pool as a unit of mass divided by unit area (*e.g.* kg/m^2, Mg/ha). The literature refers to soil nitrogen content by using the terms of soil nitrogen storage (Lin et al., 2000) and density (Yang et al., 2007), but also, usage of the term soil nitrogen pool is common (Jagadamma et al., 2007; Marty et al., 2017; Mo et al., 2020). When a publication presented their STN data in concentration rather than pool, storage, density, or stock, we used the following equation for data transformation (He et al., 2008; Barros et al., 2015):

$$STN_{pool}\left(g\ cm^{-2}\right) = STN_{concentration}(\%) \times BD\left(g\ cm^{-3}\right) \times Soil\ depth\ (cm) \quad (1)$$

where STN_{pool} is the soil total nitrogen pool (stocks/storage); $STN_{concentration}$ means soil total nitrogen concentration, BD is soil bulk density and soil depth corresponds to 30 cm depth of original $STN_{concentration}$ data. If the original $STN_{concentration}$ data did not reach to 30 cm depth, the data would be further standardized to 30 cm depth (see Eq. 4 and 5). Likewise, to study how SOC pool, a potentially-important measure, affect the STN pool, we used the following equation (Guo and Gifford, 2002; Li et al., 2012) to calculate SOC pool if a paper presents their SOC data in concentration instead of pool:

$$SOC_{pool}\left(g\ cm^{-2}\right) = SOC_{concentration}(\%) \times BD\left(g\ cm^{-3}\right) \times Soil\ depth\ (cm) \quad (2)$$

where SOC_{pool} denotes soil organic carbon pool (stocks/storage), $SOC_{concentration}$ indicates soil organic carbon concentration, and soil depth corresponds to the 30 cm depth of original $SOC_{concentration}$ data. If the original $SOC_{concentration}$ data did not reach to 30 cm depth, they would be further standardized to 30 cm depth (see Eq. 4 and 5). However, if a paper does not provide the data of soil BD, we then utilized the following equation to calculate soil BD (Post and Kwon, 2000; Guo and Gifford, 2002):

$$BD = \frac{100}{\frac{OM\%}{0.244} + \frac{100-OM\%}{MBD}} \quad (3)$$

where BD again indicates soil bulk density; OM% is the percentage of soil organic matter, and MBD denotes soil mineral bulk density. We assume that OM% equals to the percentage of soil carbon divided by 0.58 (Mann, 1986; Shi et al., 2018), and the value of MBD is 1.64 (Mann, 1986; Shi et al., 2018).

To increase the comparability of STN pools (and SOC pools) across natural, agricultural, and urban habitats, we normalized the data to 30 cm soil depth. Many data sets from urban soil papers only sample to 30 cm depth. This follows The Intergovernmental Panel on Climate Change (IPCC) guidelines that only consider soils to a maximum of 30 cm (see Penman et al., 2003). Nonetheless, for those papers whose data sets of STN and SOC pools did not reach to 30 cm, we extrapolated those data and adapted from literature by using the following asymptotic equation (Jobbágy and Jackson, 2000; Jobbágy and Jackson, 2001):

$$Y = 1 - \beta^{d} \quad (4)$$

where Y is the cumulative proportion of soil nitrogen/carbon pool from the soil surface to its depth (cm); β indicates the relative decrease of soil nitrogen/carbon pool as soil reaches to deeper depth; d indicates soil depth.

Based on the Eq. 4, we can extrapolate STN and SOC data, from soil depth <30 cm to 30 cm, through calculating the ratio of two Y values. The first Y value is for 30 cm soil depth (numerator), and the other Y value is for soil depth <30 cm (denominator):

$$X_{30} = \frac{1-\beta^{30}}{1-\beta^{d0}} \times X_{d0} \quad (5)$$

where X_{30} denotes the soil nitrogen/carbon pool in upper 30 cm; d_0 is the original soil depth (cm) in each soil study and is always lower than 30 cm; X_{d0} denotes the soil nitrogen/carbon pool from soil surface to d_0 soil depth.

We followed Shi et al. (2018) and valued β as 0.9831 and 0.9786 for nitrogen and carbon in Eq. 4, respectively. It was found that the soil depth distribution of nitrogen did not significantly change across a variety of ecotones (Yang et al., 2011), and neither did soil carbon (Jobbágy and Jackson, 2000; Yang et al., 2011). We maintained the β values for soil nitrogen and carbon constant throughout analysis. For the data of other important soil properties that did not reach to 30 cm soil depth, including soil pH, clay content and BD, again we referred to the global soil data in 30 cm depth from Zhao et al. (2019) to acquire an approximate value because we could not find relevant literatures to extrapolate those soil data with soil depth <30 cm, as we did in Eqs. 4 and 5 for SOC and STN data.

Box 1
Nitrogen fluxes moderating natural and anthropogenic soil nitrogen pools.

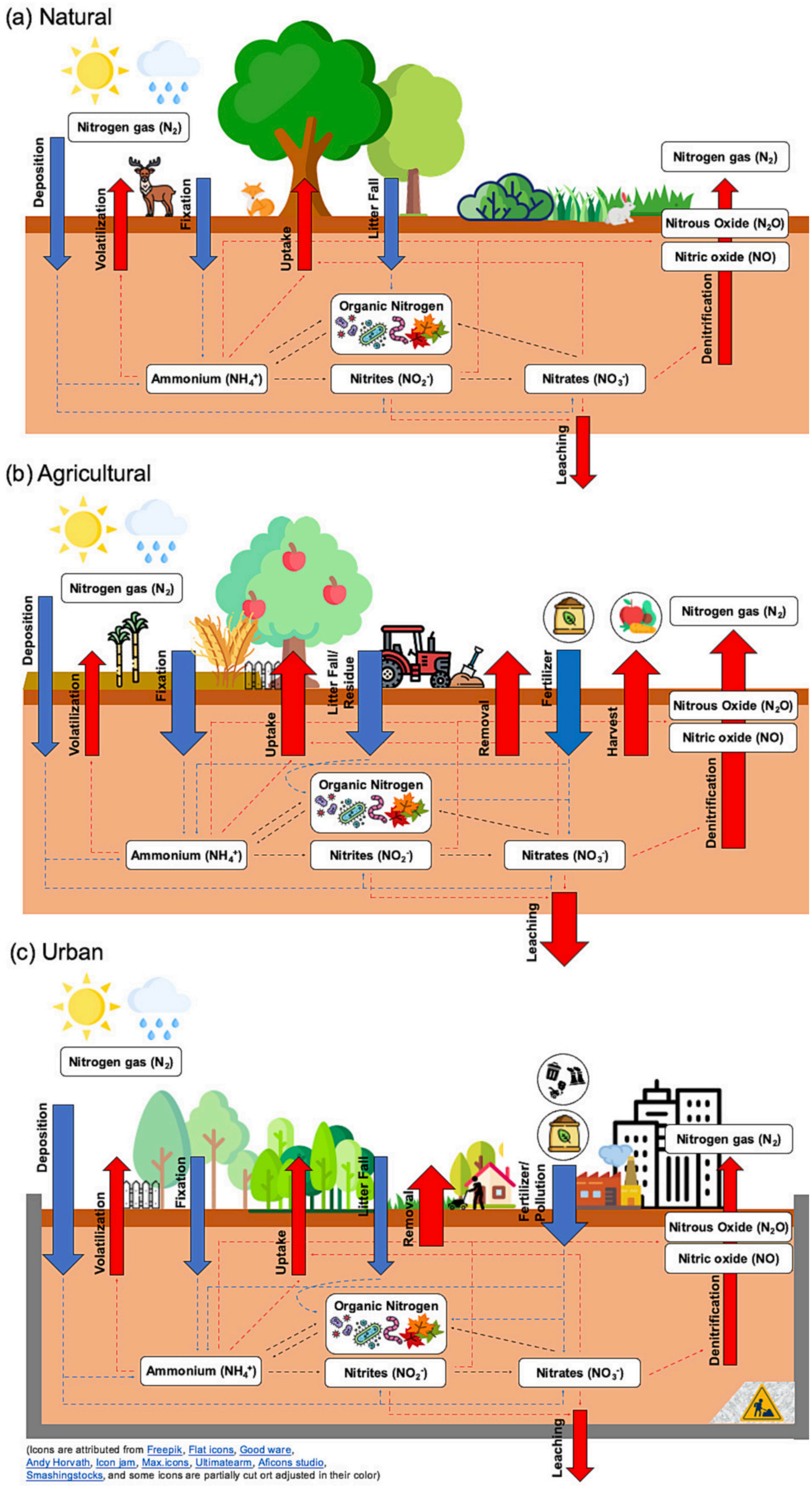
The pool of soil total nitrogen (STN) reflects the balance (or net difference) between soil nitrogen inputs and outputs. In natural habitats, common soil nitrogen inputs include biological nitrogen fixation, litter fall and atmospheric nitrogen deposition, while common soil nitrogen outputs include ammonia volatilization, plant uptake, denitrification, and nitrogen loss from leaching (Fig. A1 a).

At first glance, the soils of agricultural habitats appear to support the same amount of nitrogen as is found in the soils of natural habitats. However, that nitrogen pool is not stable over short time scales; the nitrogen dynamics in agricultural soils are driven by anthropogenic management and activities as opposed to naturally occurring flows through decomposition, mineralization, and nitrogen cycling (Fig. A1b). When we consider how nitrogen enters agroecosystems, nitrogen fertilizer represents a substantial input, and higher nitrogen fixation in legumes and rhizobial-host plants brings even more nitrogen into soil environments and can improve growing conditions (Van Kessel and Hartley, 2000). When plant residues are allowed to remain, they return nitrogen *via* decomposition and mineralization. However, many agricultural practices include the removal of plant residue and subsequent soil nitrogen loss, and thus requiring higher nitrogen inputs. Common agricultural practices, like sulfur addition or other trace nutrients, can increase soil nitrogen uptake by crops (Salvagiotti et al., 2009), but they magnify nitrogen loss when plant residues are removed (Fig. A1 b). Further soil nitrogen loss is blamed on nitrous oxide (N_2O) emission. Globally, agriculture is responsible for almost 60 % of total anthropogenic N_2O emissions (López-Aizpún et al., 2020), and N_2O concentration is growing with increasing fertilizer use and agricultural land area (Reay et al., 2012). Additional nitrogen is lost from agricultural soils *via* leaching and riverine transport. This is especially the case of nitrate (NO_3^-), a critical nutrient for plant growth, that is highly soluble and can represent a substantial loss. Loss due to leaching (particularly NO_3^-) can contaminate both surface and groundwater, causing eutrophication. Therefore, the size of the STN pool may be similar to natural soils, but the magnitude and stability of this pool is moderated by human activity with fluxes that are costly to farmers and ecosystem health.

The soil nitrogen pool in urban environments is influenced by a very different set of factors, especially with respect to nitrogen inputs. Urban soils receive more nitrogen inputs from atmospheric deposition *via* industrial air particles and vehicle emissions. Further, the soils of urban landscaping and recreational spaces receive nitrogen from chemical fertilizer and are subject to pollution from man-made wastes (Fig. A1 c). With irrigation and fertilizer application, lawns are subject to receive and accumulate more nutrients and water. However, if urban soil nitrogen outputs (*e.g.* soil or plant removal from managed green spaces and runoff and leaching of nitrogen fertilizer) do not compensate for the additional nitrogen inputs, the STN pool in urban environments will likely accumulate more nitrogen over time (Fig. A1 c). For example, intensively managed and fertilized urban lawns increase soil nitrogen retention because common soil nitrogen outputs, like N_2O emission and NO_3^- leaching, do not offset the increased nitrogen inputs from fertilizer (Pickett et al., 2008).

(*continued on next page*)

Box 1 (*continued*)



Box Fig. 1. The fluxes to and from soil nitrogen pools in (a) natural, (b) agricultural, and (c) urban environments. Blue solid arrows represent soil nitrogen inputs, and red solid arrows are soil nitrogen outputs. Within the soil, dashed blue arrows indicate entering nitrogen fluxes within the pool, and dashed red arrows indicate exiting nitrogen fluxes within the pool; dashed black arrows indicate local nitrogen fluxes inside the pool. In agricultural and urban environments, the weight of the solid arrow, when appropriate, reflects the magnitude of that flux as it is less/more than the corresponding flux in natural environments. Note, for brevity, anaerobic ammonia oxidation (anammox), which transforms ammonium and nitrite into dinitrogen gas and water, and dissimilatory nitrate reduction to ammonia (DNRA) are not represented in this figure.

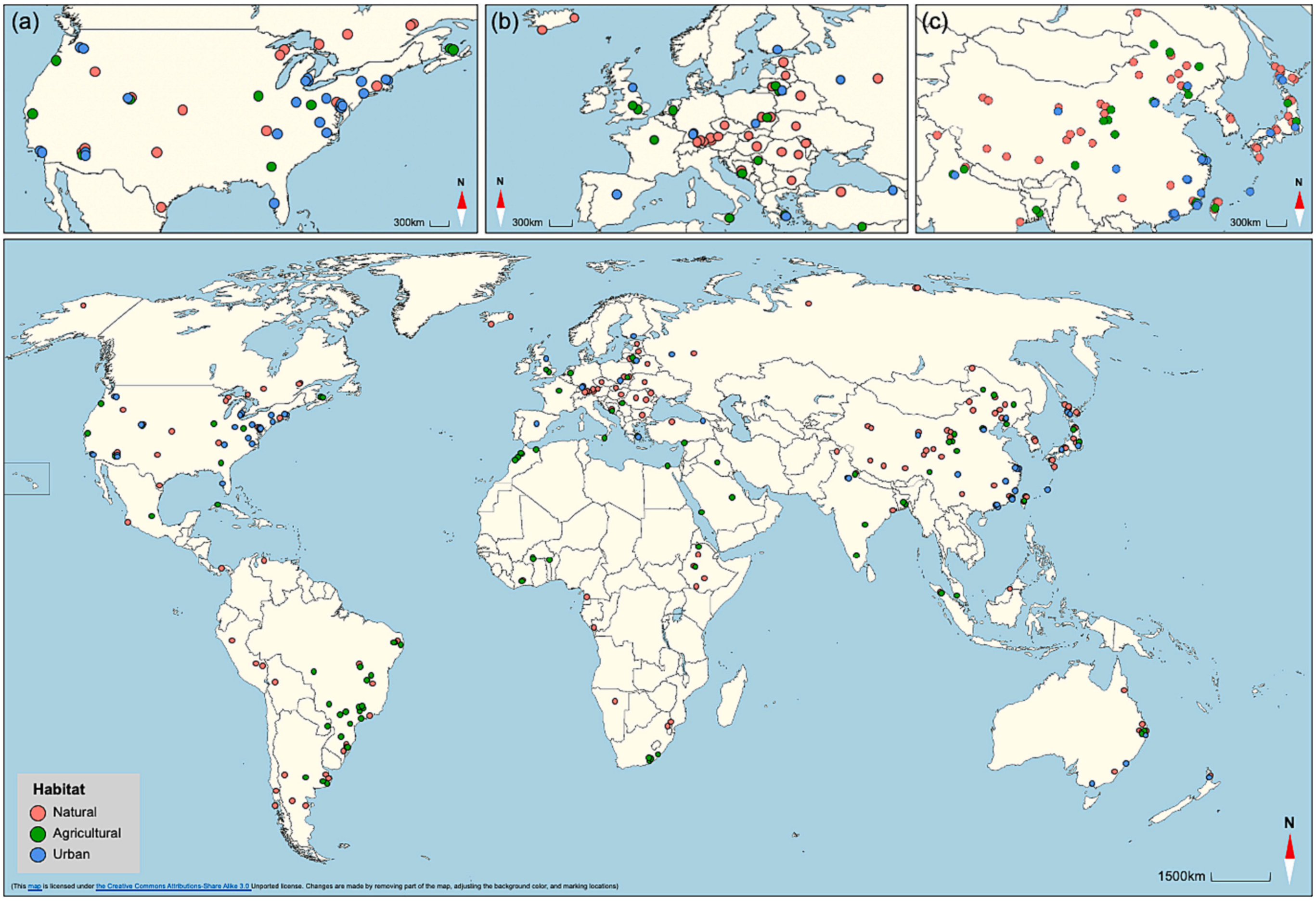


Fig. 1. The global distribution of all study locations. To make some sampling points clearer, the areas with higher density of study locations, including (a) North America, (b) Europe, and (c) East Asia, are enlarged. The World Map is licensed under the Creative Commons Attribution-Share Alike 3.0 Unported license; adjustments were made by cropping, altering background color, and marking locations.

2.4. Statistical analysis

To identify the significant difference in STN among natural, agricultural, and urban habitats, we first utilized a Shapiro-Wilk test to determine normality of our data, then we used a Kruskal–Wallis test to identify significant difference among the three habitats. Because the analysis revealed significant differences (p-value <0.05), we utilized Wilcoxon rank-sum test with a Bonferroni correction (Verhoeven et al., 2005) to examine which pairs of groups were significantly different.

Before examining the effect of different climatic and soil factors on STN through Structural Equation Modeling (SEM), we performed Principal Component Analysis (PCA) to correlate them to the component scores revealing the interdependence of each variable on that habitat (see Supplemental Fig. 2 and 3). Moreover, we performed linear regression to examine the effect of each single climatic or soil factor on the nitrogen pools. Ultimately, based on the first principal component, the correlations among variables from the PCA, and the linear regression analyses, we constructed hypothetical pathways in SEM (Wei et al., 2012; Chen et al., 2013; Vestergård et al., 2018).

We used two R (v4.1.1) packages, FactoMineR (v2.8; Lê et al., 2008) and factoextra (v1.0.7; Kassambara and Mundt, 2021) for PCA and data visualization, respectively. Moreover, we utilized lavaan package (v0.6.15; Rosseel, 2012) to perform SEM in each habitat type. All the statistical analyses were performed in R (v4.1.1).

3. Results

3.1. STN pool and habitat

Across all data sets, the size of the STN pool in urban habitats (7.03 ± 5.50 Mg/ha) is significantly larger (chi-square = 26.25, p-value <0.001) than the pools of natural (5.22 ± 3.49 Mg/ha) and agricultural habitats (4.72 ± 2.64 Mg/ha). The magnitudes of the nitrogen pools for both natural and agricultural soils are comparable. However, a portion of the agricultural STN data is distributed within the interval of −1–0 Mg/ha in logarithmic scale (Fig. 2), suggesting extremely low soil nitrogen retention.

3.2. Correlations among climatic variables, soil factors, and nitrogen pools

We used PCA to identify the interdependence of STN influencing variables across the three habitat types. The first two principal components (PCs) explain over 50 % of total data variation in all the habitat types (natural: 53.7 %; agricultural: 51.8 %; urban: 56.1 %; see Supplemental Fig. 2), and the PCA analyses (Supplemental Fig. 2) show little difference in how climate and soil variables are related to each other across all soils as they have similar loadings in the first two PCs. Furthermore, we construct correlation plots showing the interrelationships among variables in more detailed over PC1–PC5 for each habitat type (Supplemental Fig. 3). Overall, all three habitats reveal complex relationships among climatic and soil variables.

To understand how different factors determine the STN pool simultaneously between natural and anthropogenic soils, we construct SEM frameworks that are based on PCA and regression analyses (see the regression analyses in Supplemental Fig. 4). All three models show good fit and compatibility between the modeling structure and our data. Comparative Fit Index (CFI) and Tucker-Lewis Index (TLI) are close to or higher than 0.95, and the Root Mean Square Error of Approximation (RMSEA) is all lower than 0.08 (Fernandez et al., 2018; Tubbs-Cooley et al., 2018).

When all soil nitrogen influencing parameters are incorporated to determine the STN pool, natural habitats show a complicated network including many factors influencing the nitrogen pool (Fig. 3 a). However, in agricultural and urban environments, the complexity of significant relationships in soil nitrogen is decoupled from climatic and soil factors (Fig. 3 b and c), suggesting that agricultural and urban STN pools are moderated to a lesser degree by naturally occurring environmental factors. Factors like agricultural activities and urban human influences may be moderating nitrogen pools in anthropogenic soils, though those factors are not included in our SEM frameworks. This is seen in agroecosystems, where the significance levels of the relationships between soil organic carbon and STN pools and environmental factors are weakened (Fig. 3 b). Our SEM analysis also suggests that agricultural soil nitrogen pools seem to be more subject to anthropogenic impacts than the nitrogen pools in urban soils, where the effects of environmental drivers on the STN pool still play a meaningful role.

4. Discussion

4.1. *Soil nitrogen in natural ecosystems*

The magnitude and high variability of total soil nitrogen in pools across the globally diverse natural ecosystems we sampled reflects varied climatic conditions and vegetation types (Fig. 2) (Yang et al., 2007; Xu et al., 2013). In contrast, the lowest variability in STN pools of agroecosystems reflects industrialized and standardized management practices for agriculture. In our analysis, the soil nitrogen pools of natural ecosystems are significantly moderated by many climatic and soil factors (Fig. 3 a), and these factors control the soil nitrogen pools by influencing soil nitrogen inputs and outputs. For instance, higher precipitation can result in higher soil water content and lower oxygen availability, and this may cause the activity changes of biological nitrogen fixation (Smercina et al., 2019) and denitrification (especially N_2O reductase is very sensitive to aeration) (Owens et al., 2016) that require anaerobic conditions. Furthermore, when soil pH changes, the process of nitrification is carried out by different microbial communities (*e.g.* ammonia-oxidizing archaea or bacteria) (Jiang et al., 2015), and this may influence subsequent soil nitrite (NO_2^-) and nitrate (NO_3^-) availability which can be subject to leaching or plant uptake as soil nitrogen loss.

Our analysis shows that the effects of climatic and soil factors on nitrogen pools are reduced in anthropogenic soils, and this may suggest anthropogenic soil nitrogen pools are subject to other factors that are not easily measured. The presence of those critical factors, like diverse human nitrogen uses, is influencing the nitrogen pools in anthropogenic soils, even interacting differently between agricultural and urban environments (Table 1). In agreement with the data of nitrogen fluxes (Table 1), our analyses (Fig. 2 and 3) suggest that the fates of anthropogenic nitrogen inputs and use are different. Whether or not they are agricultural or urban can determine their fate as a source or a sink of nitrogen. Moderating and managing nitrogen in these two anthropogenic systems requires very different strategies.

4.2. *Agricultural soil nitrogen pools: Decoupled mechanisms, potential nitrogen source and environmental consequences*

The magnitude of the soil nitrogen pool in agricultural ecosystems is not different than that of natural soils, even across a global sampling of diverse environments (Fig. 2). However, the complexity of mechanisms driving the agricultural nitrogen pools is decoupled from climatic and soil factors. This suggests the nitrogen pools in agricultural soils may be subject to other factors that are not easily or uniformly quantified from the literature, like high nitrogen loading and the level of tillage.

For instance, the fluxes of nitrogen in and out of agroecosystems are far greater than that of natural ecosystems, and as it is moderated by anthropogenic influences, it varies with different regions and continents

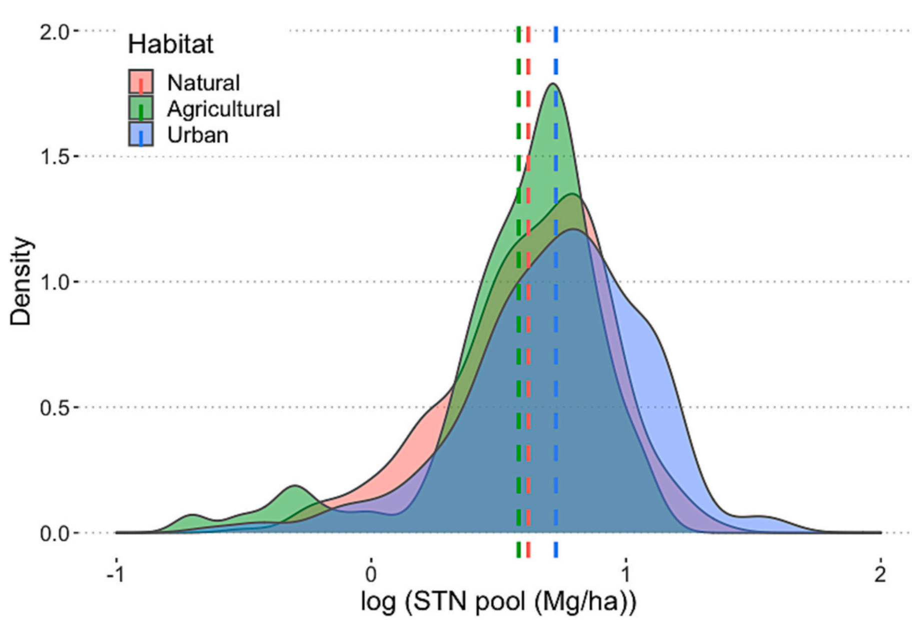


Fig. 2. The density plot of log STN pool among natural, agricultural, and urban habitats in 30 cm soil depth. The mean values are represented as colored dashed lines, respectively. The STN pool in the urban habitats (7.03 ± 5.50 Mg/ha, $n = 222$) is significantly higher (chi-square = 26.25, p-value <0.001) than natural (5.22 ± 3.49 Mg/ha, $n = 346$) and agricultural (4.72 ± 2.64 Mg/ha, $n = 345$) habitats.

(Bouwman et al., 2009). It is estimated that world nitrogen fertilizer production is 123 Tg per year (Kopittke et al., 2019). When that fertilizer is applied liberally to agriculture, it can change soil biogeochemistry (Han et al., 2015) and reduce residence time (or low nitrogen retention) when conditions favor more labile forms (NO_3^-) (Kopáček et al., 2013; Sebilo et al., 2013). Moreover, some of the applied nitrogen is taken up by crops, while the rest is only retained in agricultural soils temporarily and subject to loss at cost to farmers and environmental quality (Bodirsky et al., 2014; Kanter et al., 2020). In agreement with nitrogen flux data from recent literature (Table 1), our analysis (Fig. 2) suggests that agricultural soil nitrogen pools are a potential nitrogen source because most of nitrogen loading is not likely retained in soils.

The effects of high nitrogen inputs to soil biogeochemistry in agroecosystems are varied yet highly interdependent. Heavy use of nitrogen fertilizer can lead to soil acidification *via* several mechanisms including: higher nitrification rates (reducing pH) (Shi et al., 2019) and the leaching of base anions to maintain charge balance (Fageria and Nascente, 2014). In spite of natural soil buffering, nitrogen addition can influence soil pH and carbon quality (Ball and Virginia, 2014), and simultaneously, soils can undergo a buffering transition from base (Ca^{2+} and K^+) to non-base anions (Al^{3+} and Mn^{2+}) (Tian and Niu, 2015). Subsequently, soil acidification can cause increased soil inorganic carbon losses, primarily as gaseous carbon dioxide release and leaching of bicarbonate (Zamanian et al., 2018; Raza et al., 2021). Moreover, changes in soil pH can impact soil biota (Zhalnina et al., 2015) affecting soil physical and chemical properties, like soil aggregations (Lehmann et al., 2017) and SOC stocks (Mueller et al., 2015). The changing soil biogeochemistry resulting from anthropogenic nitrogen addition in farms may loosen the intrinsically bonded relationships among soil properties (*e.g.* relationship between soil pH and carbon stocks), as we show in our analysis that many significant relationships are reduced in agricultural relative to natural soils (Fig. 3 b).

Although 68 million tons of nitrogen fertilizer is applied to agriculture worldwide each year (Chen et al., 2014), only about half of that fertilizer is taken up by crops on average (Kanter et al., 2020). The result of this inefficiency is that agricultural soils act as a source for reactive nitrogen (N_r) (*e.g.* NO_3^- and nitrogen oxides NO_x) and can cause environmental damage to waterways as nitrogen retention time is often short. When nitrogen fertilizer and nitrogen-rich sediments enter freshwater and coastal ecosystems, biodiversity may increase due to higher nutrient availability in oligotrophic environments (Oczkowski and Nixon, 2008). However, this is likely temporary and not sustainable. Eventually excess nitrogen will cause harmful algal blooms, oxygen depletion, and eutrophication in downstream aquatic habitats (Kramer et al., 2006; Huang et al., 2017; Byrnes et al., 2020). Furthermore, if N_r enters our drinking water, NO_3^- consumption may lead to human pathology like elevated blood pH, kidney failure or cancer (Erisman et al., 2013). Nitrogen can also be lost from agricultural soils *via* aeration and N_2O emission (Cavigelli et al., 2012; Kaltenegger and Winiwarter, 2020). During the past five decades, the N_2O emission from the use of synthetic nitrogen fertilizer in global cropland has increased by 180 % (Xu et al., 2020), the consequences of which are stratospheric ozone layer depletion (Kanter et al., 2016) and regional warming effects (Oikawa et al., 2015).

Soil nitrogen and carbon cycles are tightly coupled (Li et al., 2014) and jointly moderated by microbial communities, plants, and soil properties (Ball and Virginia, 2014; Deng et al., 2016). Establishing global goals that address atmospheric carbon concentration targets and the climate-soil feedback systems to mitigate climate change may be impossible without considering soil nitrogen dynamics. For example, the 2015 Paris Agreement of the United Nations states global goals of “4 per 1000”. That agreement aims to increase global organic carbon stocks in agricultural soils by 0.4 % per year to remove a substantial portion of atmospheric carbon dioxide (http://4p1000.org) (Rumpel et al., 2020). Nevertheless, soil nitrogen content may hinder abilities to achieve this goal because increasing soil organic carbon stocks will require stoichiometric balance with nitrogen (*e.g.* the formation of new soil organic matter require both carbon and nitrogen) (Van Groenigen et al., 2017). Our analysis reveals that the magnitude of the soil nitrogen pool in agriculture is not larger than that of natural systems across our global sampling. Therefore, although anthropogenic nitrogen loading is higher in agricultural soils, it is also lost at a faster rate. Agricultural nitrogen additions may limit the potential for higher soil carbon stocks when standard farming practices like plant residue removal and tilling are employed. In agreement with recent literature (Van Groenigen et al., 2017; Giller et al., 2021), our results suggest that the “4 per 1000” goals of the UN may need to be reconsidered with respect to nitrogen availability in agricultural soils.

4.3. *Urban soil nitrogen pool: Potential nitrogen sink and environmental consequences*

In contrast to agricultural soils, urban soils support a significantly higher nitrogen pool than natural soils (Fig. 2) in agreement with what has been seen previously (Vodyanitskii, 2015; Vasenev and Kuzyakov, 2018; Trammell et al., 2020). Moreover, in accordance with data from recent literature (Table 1), the finding that urban soils support a higher amount of total nitrogen suggests they serve as a nitrogen sink, gradually accumulating nitrogen mainly from deposition and fertilizer use. Unfortunately, urban soils as a sink for soil nitrogen may not reflect a healthy increase in ecosystem services (see Midgley et al., 2021) because the origin of the increased nitrogen content in urban soils is mostly anthropogenic. Moreover, a greater nitrogen pool may not ultimately be beneficial to soil and human health.

The magnitude of the urban soil nitrogen pool is primarily attributed to urban atmospheric nitrogen deposition and fertilizer use. A recent global analysis reports that urban ecosystems are hotspots of atmospheric nitrogen deposition related to industrial and other anthropogenic activities (Decina et al., 2020). Moreover, poor air quality resulting from particulate matter and the residues of burning fossil fuels increase the nitrogen level of precipitation in urban environments (Houdeshel et al., 2015) that can contribute to a higher amount nitrogen flowing into urban soils. For instance, a recent study in Boston, Massachusetts area found a positive relationship between vehicle NO_x emission and soil nitrogen deposition, and the NO_3^- in urban soil solution is positively correlated to the NO_3^- input from throughfall (Decina et al., 2017). However, nitrogen deposition is highly variable across urban ecosystems, and fertilizer can also be an important component of increased soil nitrogen in cities (Wielemaker et al., 2019). It is estimated that urban turfgrasses can receive 50–300 kg of nitrogen fertilizer per hectare each year (Branham, 2008; Carey et al., 2012), and the nitrogen addition rate of urban lawns and golf courses can be close to the fertilization rate of agronomic row crops (Gu et al., 2015). Therefore, when common soil nitrogen outflows, like NO_3^- leaching, do not reconcile with additional nitrogen inputs from atmospheric deposition and fertilizer, over time urban soils are likely to be a sink of nitrogen (Raciti et al., 2008).

A large portion of the nitrogen in urban soils, however, may not be biologically available or beneficial to some microorganisms and plants. Increasing nitrogen availability in soils allows fast-growing species of plants and soil microbes to outcompete slow-growing organisms which are historically adapted to nitrogen-limited environments (Johnson et al., 2008; Revillini et al., 2016). Moreover, high loading of anthropogenic nitrogen can reduce the abundance of diazotrophs (Leigh, 2010) and mycorrhizal fungi (Treseder, 2004) because the plants are now less dependent on symbiotic relationships for water and nitrogen uptake. This negative plant-soil feedback transforms otherwise mutualistic microorganisms in to less cooperative or even competitive antagonists (Kiers et al., 2003; Leigh, 2010), and opportunistic pathogens proliferate (Wehner et al., 2010; Revillini et al., 2016). In addition to symbiotic microorganisms, many free-living fungi and nitrogen-fixing bacteria are sensitive to nitrogen inputs, and soil fauna respond to the increased

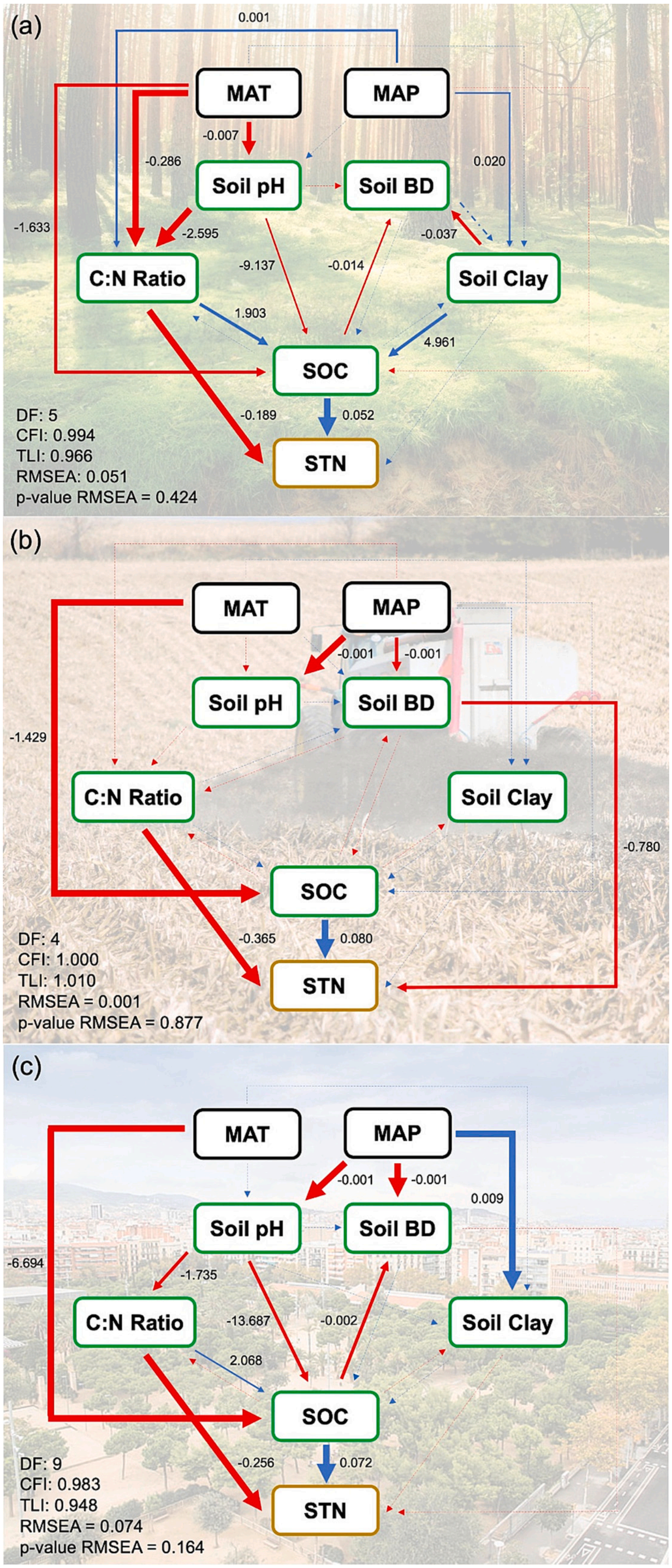


Fig. 3. The SEM analyses for the effects of climatic and soil variables on STN pool in (a) natural, (b) agricultural, and (c) urban habitats. Solid and dashed arrows indicate significant cause-and-effect relationships and non-significant relationships, respectively; blue and red arrows denote positive and negative effects, respectively. The numbers adjacent to the solid lines represent path coefficients, and the path coefficients of dashed lines (non-significant relationships) are not shown. The thickness of solid arrows represents the level of significance (from thin to thick arrows: p-value <0.05, p-value <0.01, and p-value <0.001, respectively). Black boxes indicate climatic variables; green boxes denote soil variables; brown boxes mean the variable of STN. (DF: Degree of Freedom, CFI: Comparative Fit Index, TLI: Tucker-Lewis Index, RMSEA: the Root Mean Square Error of Approximation) (Background images courtesy of the Creative Commons online).

Table 1
Global Estimates of Nitrogen (N) Fluxes and Balance (kg ha^{-1} yr^{-1}) in Agricultural and Urban Soils.

N Fluxes in and out of Soils	Agricultural Soils		Urban Soils		References (agricultural; urban)
	N Inputs	N Outputs	N Inputs	N Outputs	
N Deposition	3.9	–	99.0	–	Ladha et al. (2016); Decina et al. (2020)
N Fixation	75.4–115.1	–	17.8–78.9[@]	–	Herridge et al. (2008), Kakraliya et al. (2018); Zheng et al. (2019)
Crop Residue and Manure (mainly in agricultural soils), Litter Fall (mainly in urban soils)	9.8	–	50.0–100.0[#]	–	Ladha et al. (2016); Hill et al. (2022)
N Fertilizer	72.6–197.0	–	50.0–300.0	–	Schütz et al. (2018), Wang et al. (2018); Branham, (2008), Carey et al. (2012)
NH_3 Volatilization	–	19.1	–	15.8	Pan et al. (2016);
NO Emission	–	3.1	–	2.3[$]	van Lent et al. (2015); Hall et al. (2008), Zhan et al. (2023)
N_2O Emission		8.6–12.0	–	3.0	van Lent et al. (2015), Qasim et al. (2021); Zhan et al. (2023)
Denitrification (N_2 as Final Product)	–	61.7–86.1[*]	–	2.3[#]	van Lent et al. (2015), Qasim et al. (2021), Scheer et al. (2020); Shun et al. (2018)
Crop Uptake and Harvest (mainly in agricultural soils); Plant Uptake and Clipping (mainly in urban soils)	–	62.9	–	120.0–150.0[#]	Ladha et al. (2016); Kaye et al. (2005)
Leaching (mainly in NO_3^-)	–	297.0	–	35.1[#]	Qasim et al. (2021); Nidzgorski and Hobbie (2016)
Estimated Balance (Total N Inputs – Total N Outputs)	−318.5 to −126.6 (−0.3 to −0.1 Mg/ha yr^{-1})		8.3–399.4[!] (0.0–0.4 Mg/ha yr^{-1})[!]		–

Note: values are based on the average numbers provided by the literatures.
[*] : values are estimated by adopting the numbers of N_2O emission from van Lent et al. (2015) and Qasim et al. (2021) into the denitrification ratio ($N_2O/N_2O + N_2$) from Scheer et al. (2020).
[@] : values are estimated by global nitrogen fixation in soil in response to nitrogen enrichment due to the lack of direct data of urban soil nitrogen fixation from literatures.
[$] : value is estimated by adopting the numbers of N_2O emission from Zhan et al. (2023) into the nitrous oxide to nitric oxide ratio (N_2O/NO) from a local study Hall et al. (2008).
[#] : values are obtained from local or regional studies due to the lack of urban soil data from literatures.
[!] : values may not be well representative for urban soil nitrogen balance at a global scale because not all urban data included in this table are across global sampling.

nitrogen indirectly through their relationships with the microbial community and vegetation (Erisman et al., 2013).

Elevated nitrogen in urban soils can have direct and negative consequences to human health. NO_x is one of the main sources of tropospheric ozone, smog, and particulate matter (Ban-Weiss et al., 2008; Wong, 2017; Wang et al., 2019). With environmental exposure to NO_x, people are at a higher risk of mortality (Orellano et al., 2020), chronic kidney disease (Bowe et al., 2017), and lung cancer (Hamra et al., 2015). Globally, urban areas are expected to expand by 0.6–1.3 million km^2 between 2015 and 2050 and urban populations are estimated to increase 2–3 billion by 2050 (Huang et al., 2019), and nitrogen use correlates with human population density (Kobus, 2000; Ren et al., 2014). As anthropogenic nitrogen accumulates in urban soils, it is important to consider human health and urban quality of life in addition to ecological and ecosystem health.

5. Conclusions

The present meta-analysis reveals that the mechanisms moderating agricultural and urban soil nitrogen pools is decoupled from climatic and soil factors. We conclude that anthropogenic soil nitrogen pools are subject to a variety of factors that are not easily or frequently measured in the literature we gathered, like the amount of nitrogen loading in agriculture or the level of nitrogen pollution in a city. When we retrieved data of moderating factors with respect to human nitrogen use from recent large-scale studies, our analysis about nitrogen pool size, in agreement with the literature, suggests agricultural soils serve as a nitrogen source (nitrogen pool size similar to natural soils) while urban soils serve as a nitrogen sink (larger size of nitrogen pool). This is important because the fate and distribution of anthropogenic soil nitrogen affects important ecosystem services, human health and even climate stability as soil nitrogen and carbon dynamics are tightly coupled. This compiled data set and analysis can inform a better nitrogen use and management with lower environmental impact and socio-economic loss.

Author contribution statement

Mr. Chien proposed the question, conducted the analyses, and both authors wrote the manuscript. We acknowledge funding from the National Science Foundation (DEB#2120677).

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

Data will be made available on request.

Acknowledgements

We acknowledge funding from the National Science Foundation (DEB#2120677) and Richard V. Pouyat for valuable discussions that improved this research and manuscript. We also acknowledge two reviewers for very thoughtful comments and suggestions that greatly improved this manuscript.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2023.166238.

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