

# The potential use of biochar to reduce nitrogen waste from farming systems in India

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## ABSTRACT

The aim of this paper is to quantify the impact of incorporating biochar into soils on net nitrogen waste from farming systems in India. It assumes only crop residues that are currently burnt in the fields are used to produce biochar. It accounts for losses of nitrogen occurring during pyrolysis, and the potential savings due to capture of reactive nitrogen from other parts of the farming system and from industry and energy sectors. In 2020, this could have been used to capture up to 67% of the nitrogen lost as nitrogen oxides and ammonia from the energy, industry and farming sectors. This is equivalent to 31% of the nitrogen that was applied as fertilisers and so could be an important tool in efforts to meet the United Nations target to reduce nitrogen waste by 50% by 2030. However, if the rate of nitrogen capture is low, alternative uses for crop residues are developed, or wasted nitrogen is successfully captured by other methods, the benefits of nitrogen capture on biochar could be much lower. Nevertheless, using biochar as a method to deliver wasted nitrogen to crops is likely to sequester more carbon than alternatives because pyrolyzed carbon is highly recalcitrant. It is also likely to be a more reliable method of capturing nitrogen emissions and delivering nitrogen to crops because emissions of ammonia during storage and spreading of compost or bioslurry can be high. Therefore, even if alternative uses of crop residues are favoured by farmers, it is recommended that nitrogen sorption on biochar should be part of the process, whether it is by direct capture of nitrogen from urine and industrial nitrogen oxide emissions, or by mixing of biochar with compost or the ammonium rich bioslurry produced by anaerobic digestion.

## 1. Introduction

A high proportion of the nitrogen (N) applied to farming systems in India is wasted due to overapplication of N fertilisers and the low productivity of soils. Since the 1960s and the Green Revolution in India, the provision of N to crops by application of synthetic fertilisers has sharply increased (Aryal et al., 2021; FAO, 2022; Fig. 1). India now has the second highest agricultural use of synthetic fertiliser N in the world (FAO, 2022), and although this (and other measures introduced in the Green Revolution) are correlated to significant improvements in crop yields (FAO, 2022; Fig. 2), these yields are not likely to be sustainable in the long term due to the declining productivity of soils (Bhattacharyya et al., 2015; Srinivasarao et al., 2021). Of the  $3.287 \times 10^8$  ha total land area in India, in 2010, 36.7% was classified as degraded and 79% of that degraded land had lost productivity due to processes associated with loss

of soil organic matter; 68% by water erosion, 10% by wind erosion and 1% by other physical degradation processes (ICAR (Indian Council of Agriculture Research), 2010).

The organic matter content of soils in India is already low, typically less than 0.6% carbon (C) by weight (Katyal et al., 2001), and it continues to decline (Srinivasarao et al., 2009, 2014). There is a strong relationship between crop yield and the organic matter content of the soil (Han et al., 2018). Data from long term experiments in India demonstrate an increase in yield of between 0.01 and 0.17 t ha<sup>-1</sup> for every 1 t ha<sup>-1</sup> increase in soil organic C (Srinivasarao et al., 2014; Table 1). Soil organic matter controls crop production through its impact on soil structure (Oades, 1993), which determines soil water holding capacity (Murphy, 2015), root aeration (Thomas et al., 1996) and structural stability (Keller and Dexter, 2012; Macks et al., 1996). It also impacts crop production through the release of nutrients as the

*Abbreviations:* C, Carbon; HCN, Organic nitrogen; N, Nitrogen; NH<sub>3</sub>, Ammonia; NH<sub>4</sub><sup>+</sup>, Ammonium; NO, Nitric oxide; NO<sub>x</sub>, Nitrogen oxides; PM<sub>2.5</sub>, particulate matter with a diameter of less than 2.5 μm.

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organic matter decomposes (Kirkby et al., 2011). Lal (2004) reported that, between 1960 and 2000 across India, there was a 30 to 60% reduction in soil organic C in cultivated soils compared to undisturbed systems. According to the long term experiments reported by Srinivasarao et al. (2014), this would be equivalent to a decline in yield of between 0.3 and 5.6 t ha<sup>-1</sup> (Table 2). This is supported by observations from a range of authors of yield declines associated with loss of soil organic matter in sub-tropical systems (e.g. Yadav et al., 2000; Das et al., 2014).

Associated with the losses in soil organic matter, as well as increasing inputs of synthetic fertilisers, is a decline in the efficiency of fertiliser N use, defined here as the ratio of N in the harvested product to the fertiliser N applied. The average efficiency of fertiliser N use across India has sharply declined from over 9 kg harvested product per kg fertiliser N applied in the 1960s to only 0.4 in 2019 (Fig. 3). This represents average wasted fertiliser N over the decade 2010–2019 (calculated as fertiliser N applied – N in the harvested product) of 61 kg ha<sup>-1</sup> y<sup>-1</sup>, which, given the area of cropland in 2019 (1.69 × 10<sup>8</sup> ha, FAO, 2022), would be a total loss across India of (1.0 × 10<sup>7</sup>) t y<sup>-1</sup> (Fig. 4). Given an average price of urea-N between 2010 and 2019 of 625 \$ t<sup>-1</sup> (World Bank, 2022), this amounts to 38 \$ ha<sup>-1</sup> or 6.45 × 10<sup>9</sup> \$ across India. This is equivalent to 2.4% of the annual gross domestic product (GDP) from agriculture (\$2.73 × 10<sup>11</sup>; Trading Economics, 2022). Using the inflated 2022 price for urea-N (1982 \$ t<sup>-1</sup>; World Bank, 2022), this would increase to 121 \$ ha<sup>-1</sup> or 7.5% of the GDP from agriculture.

The process of organic matter loss may in part be related to easy access to inorganic fertilisers, which reduces the need to incorporate organic residues to provide nutrients to crops. This allows increased use of crop residues as fodder and burning of residues, either to provide fuel or as a means of timely disposal in the field (FAO and ITPS, 2015). The tight timing of the major highly productive crop rotations, such as rice-rice or rice-wheat rotations, means that farmers often have little time or available labour to collect crop residues, and so burn or dump them instead (Lohan et al., 2018; Venkatramanan et al., 2021); the sowing window between the summer rainy season (*kharif*) and the winter sown (*rabi*) crops in many parts of India is only a few days (Ravindra et al., 2019; Puri et al., 2021). Many farmers also believe that residue burning has a beneficial effect on yield, helps to control weeds and releases nutrients for the next crop (Junpen et al., 2018). However, over the longer term, the loss of soil organic matter and nutrients due to burning is likely to reduce soil productivity and increase N waste compared to management practices that incorporate organic residues and retain nutrients (Murphy, 2015).

Burning of crop residues is of concern to both human health and the environment. Because the production of crop residues is increasing, the

amount of residues burnt in the fields is also increasing (Ravindra et al., 2019). According to Jain et al. (2014), 40% of the crop residues burnt in 2008–09 were rice straw, 22% wheat straw and 20% sugarcane trash. Burning of rice straw is particularly high because of the short sowing window for the following *rabi* crop, and because it is not a favoured source of fodder, having a high silica content and low digestability and nutritional value (Na et al., 2014). Every tonne of rice straw burnt in the fields is estimated to release 1168 kg carbon dioxide, 27.8 kg carbon monoxide, 10.4 kg particulate matter, 3.2 kg non-methane hydrocarbons, 1.0 kg methane, 2.9 kg nitrogen oxides (NO<sub>x</sub>) and 0.06 kg nitrous oxide (Soam et al., 2017). Venkataraman et al. (2006) estimated that burning of crop residues in 2003 contributed 25% of India's emissions of black C and carbon monoxide and (9 to 13)% of the emissions of greenhouse gases and the particulate matter that causes respiratory illness (with a diameter of less than 2.5 μm; PM<sub>2.5</sub>). Ravindra et al. (2019) estimated that in 2017, PM<sub>2.5</sub> emissions due to burning crop residues was equivalent to (8.24 × 10<sup>5</sup>) t and was responsible for the highly publicised crisis in air quality (Shyamsundar et al., 2019), especially in the Indian capital, New Delhi (e.g. Singh et al., 2021). While air pollution has become an acute problem in India, removal of organic residues from the farming system is central to the more insidious problem of the decline in soil organic matter and reduced N use efficiency (Mandal et al., 2004).

In order to reverse the long term decline in the productivity of soils in India and the associated loss of soil nutrients, it is essential to find practical ways to enhance retention of soil organic matter. Reducing burning of crop residues in the fields could be of benefit, both to the atmosphere and to soils and crop production. Pyrolysis of the crop residues that would otherwise be burnt and incorporation of the biochar produced into the soil has been proposed as a potential approach to increase soil organic matter, while avoiding problems with N deficiency due to incorporating low N crop residues (Han et al., 2018) and the associated need for extra fertilisers, water, time or space in the rotation (Mohan et al., 2018; Smith et al., 2019; Bhattacharyya et al., 2020).

If decomposable organic matter with a high C to N ratio is incorporated directly into the soil, mineral N can initially be immobilised by the micro-organisms responsible for decomposition and so become unavailable to crops (Sharma and Prasad, 2008). Depending on the C use efficiency of the decomposer communities, the C to N ratio over which N is immobilised ranges from anything over ~8 to ~15 (Manzoni et al., 2010). This has resulted in N deficiencies observed in crops following direct incorporation of C rich materials (Aulakh et al., 2000), such as the straw residues from wheat (C:N ~ 40 to ~80; Gan et al., 2011; USDA, 2011) or rice (C:N ~ 40 to ~75; Baruah et al., 2016; Goyal and Sindhu, 2011). Shyamsundar et al. (2019) demonstrated that using a cultivator,

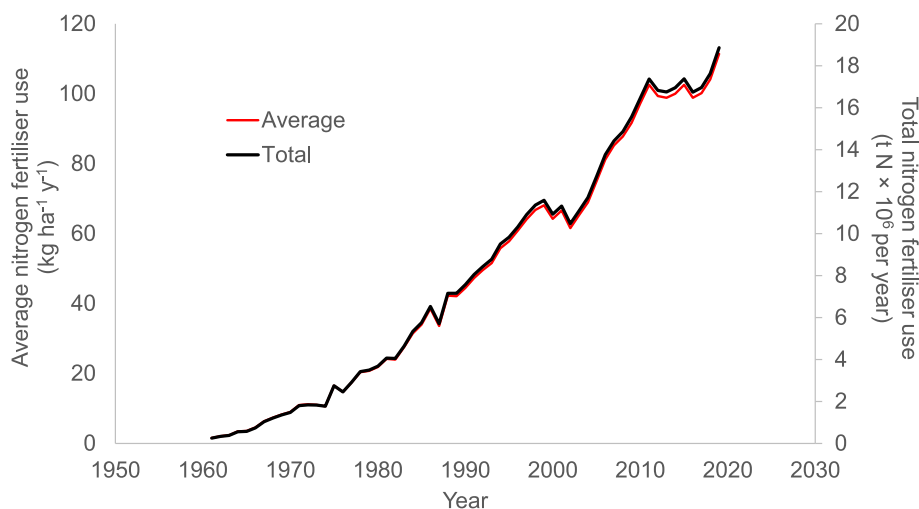


Fig. 1. Nitrogen fertiliser use for agriculture in India; data source (FAO, 2022).

such as the “Happy Planter”, to cut and lift rice straw, sow wheat directly into the soil and then use the rice stubble as a mulch, reduced N immobilisation and so increased profitability by an average of 20% (range – 7% to +44%) compared to burning. However, even this approach is likely to increase N immobilisation in the C rich mulch layer during the subsequent cropping seasons.

Additional fertilisers could be applied to compensate for N immobilisation associated with incorporation of the C rich straw (Zheng et al., 2019); the increased organic inputs with adequate compensation for immobilised N have been observed to increase both soil organic C and crop yields (Sharma et al., 2021), but also have the disadvantage of increasing N fertiliser use. This could further reduce overall N use efficiency, increase N waste and environmental pollution, and may also be a prohibitive extra cost for resource poor farmers (Aryal et al., 2021).

To maintain yields and increase soil C without increasing the demand for N fertiliser, N rich crop residues could be included in rotation with the C rich straw. In trials in the rice-wheat rotations of Punjab, a leguminous green manure, *Sesbania aculeata* L., was grown in the fallow period during May and June, between harvesting of wheat and planting of rice (Aulakh et al., 2000). Rapid release of N was observed after incorporating the *Sesbania*, and this was of immediate benefit to the following rice crop, reducing N fertiliser requirements by 50%, with additional benefits in the subsequent wheat crop, further reducing fertiliser requirements by 25%. This suggests that green manures could indeed be a viable method to compensate for N immobilisation associated with rice straw incorporation. However, growing a green manure requires either a period of fallow that is not used for production cropping, or the nitrogen fixing crops to be grown alongside the main crop, either in a regular pattern (intercropping) or by mixing the seeds and growing the two crops together (co-cropping); this may not be feasible for all regions, crops or rotations, and in some situations can result in increased greenhouse gas emissions (Bhattacharyya et al., 2012).

Microbial decomposition could be used to stabilise crop residues, reducing the C to N ratio and avoiding immobilisation of available soil N (Manzoni et al., 2010). This has been done both before application to the soil, such as by composting or anaerobic digestion (Smith et al., 2014a, 2014b; Sfez et al., 2017), or in-situ by microbial priming using inoculations with fungal micro-organisms (Gaiind and Nain, 2007; Kumar et al., 2019). Incorporation of composted or digested organic wastes has been widely demonstrated to increase the organic matter content of the soil and the availability of nutrients to crops (e.g. Mandal et al., 2007; Sodhi et al., 2009; Bhattacharya et al., 2016). However, composting is labour intensive (Smith et al., 2015; Chander et al., 2018) and requires addition of nutrient rich materials to co-compost with the straw (Roca-

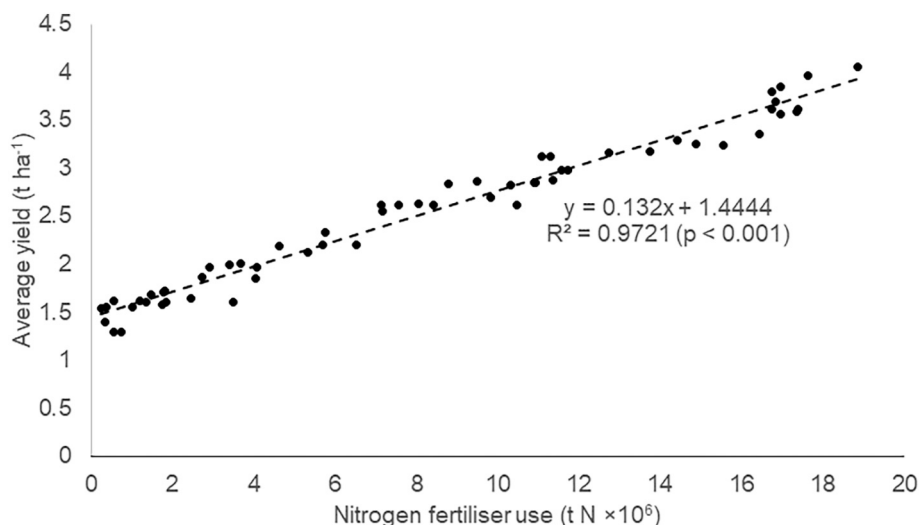
**Table 1**

Changes in yield with soil organic carbon for different crops in different soils and regions of India.

Region	Soil type	Crop	Change in yield (t ha <sup>-1</sup> per t ha <sup>-1</sup> increase of soil organic carbon)	Sources
South India	Alfisols	Groundnuts	0.01	Srinivasarao et al., 2012b, 2012f
		Finger millet	0.10	Srinivasarao et al., 2012e, 2012f
Indo-Gangetic plains	Deep alluvial sandy clay loam	Lentils	0.02	Srinivasarao et al., 2012d
		Rice	0.16	Srinivasarao et al., 2012d
Central India	Vertisols	Sorghum	0.09	Srinivasarao et al., 2012a
		Soya beans	0.15	Srinivasarao et al., 2012c
Western India	Entisols	Pearl millet	0.17	Srinivasarao et al., 2011

Pérez et al., 2009), while anaerobic decomposition requires ready access to water and equipment for the digestion process (Smith et al., 2015; Bansal et al., 2017). Therefore, in labour or water limited farming systems, composting and anaerobic digestion may not be feasible either (Smith et al., 2019). In-situ priming with inoculants has potential to circumvent N immobilisation using fungi that are adapted to use organic matter with a higher C to N ratio than bacteria to facilitate initial decomposition and release of the N contained in the straw itself (Gaiind and Nain, 2007; Kumar et al., 2019). To be accessible to resource poor farmers, these inoculants would need to be prepared directly on the farm, such as is done in natural farming systems (Smith et al., 2020); these methods have high potential to provide a feasible solution to soil degradation in India, but further work is needed to demonstrate the repeatability and efficacy of such approaches.

Instead of reducing the C to N ratio of the crop residues, pyrolysis converts the C in the crop residues into organic matter with a highly stabilised, aromatic structure (biochar) (Atkinson et al., 2010; Bruun et al., 2011; Nkoh et al., 2021) (Table 3). Because this structure is recalcitrant to decomposition (Yang et al., 2007), pyrolysing the crop residues before incorporation can reduce microbial demand for N (Nelissen et al., 2015). Biochars produced at higher temperatures (up to

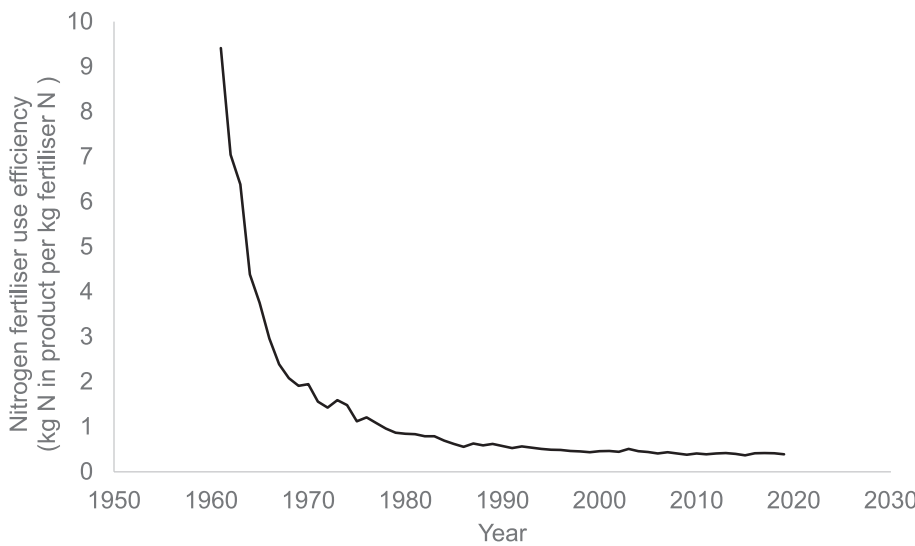


**Fig. 2.** Association of average yield of paddy rice with total nitrogen fertiliser use for agriculture in India; data source (FAO, 2022).

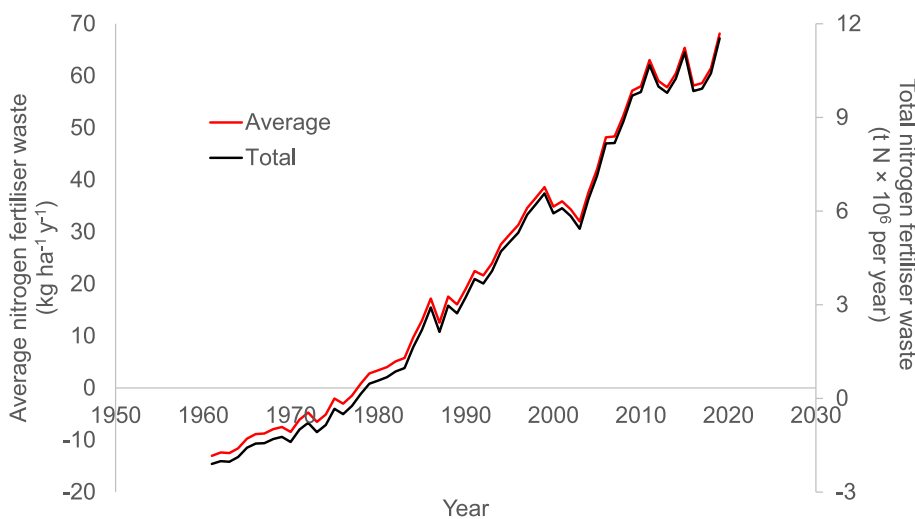
**Table 2**

Impact on average yields of changes in soil organic C across India with cultivation between 1960 and 2000. Note: SE = standard error; <sup>a</sup> source = Lal (2004); <sup>b</sup> assumes bulk density 1.3 g cm<sup>-3</sup> and topsoil depth 25 cm; <sup>c</sup> assumes minimum change in yield of 0.01 t ha<sup>-1</sup> per 1 t ha<sup>-1</sup> soil organic carbon after Srinivasarao et al. (2014); <sup>d</sup> assumes maximum change in yield of 0.17 t ha<sup>-1</sup> per 1 t ha<sup>-1</sup> soil organic carbon after Srinivasarao et al. (2014).

Region	<sup>a</sup> Soil organic carbon in cultivated soils (g kg <sup>-1</sup> )		<sup>a</sup> Soil organic carbon in undisturbed soils (g kg <sup>-1</sup> )		Reduction in soil organic carbon with cultivation (g kg <sup>-1</sup> )		<sup>b</sup> Reduction in soil organic carbon with cultivation (t ha <sup>-1</sup> )		<sup>c</sup> Minimum change in yield (t ha <sup>-1</sup> )		<sup>d</sup> Maximum change in yield (t ha <sup>-1</sup> )	
	Mean	SE	Mean	SE	Mean	SE	Mean	SE	Mean	SE	Mean	SE
Indo-Gangetic Plains	4.2	0.9	10.4	3.6	6.2	4.5	20	15	0.2	0.1	3.4	2.5
Northwest Himalayas	24.3	8.7	34.5	11.6	10.2	20.3	33	66	0.3	0.7	5.6	11.2
Northeast India	23.2	10.4	38.3	23.3	15.1	33.7	49	110	0.5	1.1	8.3	18.6
Southeast India	29.6	30.1	43.7	23.4	14.1	53.5	46	174	0.5	1.7	7.8	29.6
West Coast	13.2	8.1	18.6	2.1	5.4	10.2	18	33	0.2	0.3	3.0	5.6
Deccan Plateau	7.7	4.1	17.9	7.6	10.2	11.7	33	38	0.3	0.4	5.6	6.5
Average									<b>0.3</b>	<b>0.7</b>	<b>5.6</b>	<b>12.3</b>



**Fig. 3.** Change nitrogen fertiliser use efficiency for agriculture in India; data sources are crop yields and area of crops (FAO, 2022), and typical percentage nitrogen content (Pathak and Fagodiya, 2022) for 12 major crops (chick peas, groundnuts with shells, jute, maize, millet, pigeon peas, paddy rice, seed cotton, sorghum, soybeans, sugar cane and wheat) covering 74% (1961) to 86% (2019) or the total arable area. Total N fertiliser use (FAO, 2022) adjusted according to the percentage area of land included in the crops considered. Nitrogen fertiliser use efficiency defined as nitrogen in harvested product / nitrogen fertiliser applied.



**Fig. 4.** Nitrogen fertiliser waste from agriculture in India; data sources use are crop yields and area of crops (FAO, 2022), and typical percentage nitrogen content (Pathak and Fagodiya, 2022) of 12 major crops (chick peas, groundnuts with shells, jute, maize, millet, pigeon peas, paddy rice, seed cotton, sorghum, soybeans, sugar cane and wheat) covering 74% (1961) to 86% (2019) or the total arable area. Total N fertiliser use (FAO, 2022) adjusted according to the percentage area of land included in the crops considered. Nitrogen fertiliser waste calculated as fertiliser nitrogen applied – nitrogen in harvested product.

~700 °C) tend to be more porous and highly stabilised (Mukherjee and Lal, 2014), whereas low temperature biochars (up to ~400 °C) retain some labile organic matter that can result in short term N immobilisation (Mukherjee and Zimmerman, 2013; Nelissen et al., 2015). However, lower temperature biochars also tend to have more highly charged structures (Novak et al., 2009b), so can improve the immediate retention of available N by increasing the nutrient and water holding capacity

of the soil (Novak et al., 2009a). This is especially the case in highly weathered and acidic soils that are deficient in exchange sites (Jeffery et al., 2011; Crane-Droesch et al., 2013). Benefits are also observed in soils with a low C content (Crane-Droesch et al., 2013); for example, Kanthle et al. (2016) measured a significant reduction in leaching of nitrate applied to columns of low C soils following addition of biochar. Higher temperature biochars tend to be alkali, and so would not be

**Table 3**  
Impact of pyrolysis on carbon in crop residues. Sources: <sup>a</sup> Bruun et al. (2011), <sup>b</sup> Yang et al., 2007.

Temperature of pyrolysis (°C)	Heating rate	Carbon retained (%) <sup>a</sup>	Recalcitrant carbon (%) <sup>b</sup>
475	Low	20	~100%
	High		(88–97)%
575	Low	50	~100%
	High		~100%

recommended in alkali soils (Mukherjee and Lal, 2014).

Pyrolysis occurs by thermal decomposition of organic materials under low oxygen conditions (Mohan et al., 2006) and can emit less PM<sub>2.5</sub> than conventional burning; for example rocket-type pyrolysis cookstoves reduce emissions of PM<sub>2.5</sub> by 46% compared to traditional burning methods (MacCarty et al., 2010). Pyrolysis of crop residues could be done in large scale industrial systems (Panwar et al., 2019) with regional processing companies collecting crop residues, converting them into biochar and selling the biochar back to farmers (Müller et al., 2019). Alternatively, it could be done at the small-scale, allowing the energy provided by pyrolysis stoves to be directly used within the household for cooking (Müller et al., 2019). If there is no time or labour available to collect crop residues from the field, simple top-lit updraft gasifiers could be constructed out of metal barrels for direct pyrolysis within the field (Steiner et al., 2018). An indigenous practice of North East India, known as *Thang bun*, achieves within-field pyrolysis by burning mounds of plant biomass covered with soil to improve the fertility and sustainability of slash-and-burn systems (Hombegowda et al., 2021). Similarly, Zhou et al. (2018) described the development in China of in-situ “burn and soil cover” techniques that could be used to produce biochar in the field with significantly reduced cost, time, labour and particulate emissions.

While stabilisation of the organic matter reduces the tendency for N to be immobilised following residue incorporation, a key disadvantage of thermal pyrolysis compared to microbial treatment is that it drives off a higher proportion of the N and other nutrients contained in the crop residues themselves (Hu et al., 2017). The N content of biochar is usually higher following pyrolysis conducted at low than at high temperatures (Biederman and Harpole, 2013), but this is in part due to N being concentrated in the biochar by the associated higher loss of C at low temperatures (Bruun et al., 2011). If the losses of C are accounted for, typically between ~55% and ~100% of the N in the original feedstock is lost during pyrolysis, with a higher proportion of the N being lost at low than at high temperatures (Table 4). Nitrogen is lost as ammonia (NH<sub>3</sub>) and organic nitrogen (HCN) due to thermal decomposition of ammonium (NH<sub>4</sub><sup>+</sup>) and protein-type nitrogen compounds (Hu et al., 2017). This can contribute to atmospheric pollution because NH<sub>3</sub> and HCN are the main precursors of NO<sub>x</sub> which results in acid rain and photochemical smog (Hu et al., 2017).

The aim of this paper is to quantify the impact of incorporating biochar on the net N waste from farming systems in India, accounting for

**Table 4**  
Estimated nitrogen losses during pyrolysis. Notes: <sup>a</sup> Source: Jassal et al. (2015) (heating rate = 1–3 h); <sup>b</sup> Assumes retention given at low heating rate at 475 °C for 500 °C and at 575 °C for 600 °C by Bruun et al. (2011).

	C content (%) <sup>a</sup>		N content (%) <sup>a</sup>	C:N ratio		C retained (%) <sup>b</sup>	N lost (%)	
	Min	Max		Min	Max		Min	Max
Poultry litter								
	37		2.7	13.7				
After pyrolysis 500 °C	50	56	3.99	12.5	14.0	20	78.1	80.5
After pyrolysis 600 °C	50	56	3.27	15.3	17.1	50	55.2	60.0
Spruce pine fir								
Before pyrolysis	47		6.1	7.7				
After pyrolysis 500 °C	70	90	0.4	175.0	225.0	20	99.1	99.3
After pyrolysis 600 °C	70	90	0.54	129.6	166.7	50	97.0	97.7

losses of N occurring during pyrolysis and the potential savings in N due to increased capture of reactive N from other parts of the farming system and from industry and energy sectors. We will quantify the potential national production of biochar from the crop residues that would currently be burnt in the fields. The potential of this biochar to capture N losses occurring in typical farming systems and deliver N to the growing crop will be estimated, using both the sorption potential of the biochar and the availability of wasted N for sorption. The potential impact on net N waste across India will be quantified by balancing this against the N losses occurring during pyrolysis. Potential reduction in N losses from soils will not be included in this balance because N loss from soil is highly dependent on soil type, cropping and weather conditions. Therefore, the balance will provide a lower estimate for the potential benefits of incorporating N enriched biochar. Finally, the impacts on net N waste of biochar incorporation will be compared to the alternative methods that could be used to increase soil organic matter by incorporation of crop residues.

## 2. Materials and methods

### 2.1. Production of biochar from the crop residues that are currently burnt

Data on the amount of crop residues burnt across India were obtained from a recent analysis provided by Sahu et al. (2021). They considered eight major crop types that are widely grown throughout India; rice, wheat, maize, mustard, groundnut, sugarcane, cotton and coarse cereals (sorghum, pearl millet and barley), covering ~60% of total agricultural area in India (FAO, 2022). They obtained cropping data from statewide crop statistics and production data for 2018 provided by the Government of India (2018, 2020), augmented by district level crop activity data compiled from Indiastat (2018). This produced a detailed seasonal inventory on a high resolution grid (10 km × 10 km) for the year 2018.

The parameters provided by Sahu et al. (2021) (Table 5) were used with cropping areas and production data from FAO (2022) to provide updated estimates for the dry mass of burnt crop residues,  $M_{burnt}$  (t y<sup>-1</sup>) using the formula

$$M_{burnt} = \sum_i \left( A_i \times M_{yld,i} \times r_i \times P_{DM,i} / 100 \times p_{burnt,i} \right) \quad (1)$$

where  $A_i$  is the area of crop  $i$  in India (ha),  $M_{yld,i}$  is the average yield (t ha<sup>-1</sup>),  $r_i$  is the ratio of residues to harvestable product,  $P_{DM,i}$  is the percentage dry matter in the crop residue (%), and  $p_{burnt,i}$  is the proportion of the particular crop residues burnt in the field. Calculations were first done for 2018 to check correspondence with the original calculations provided by Sahu et al. (2021), and then repeated for 2019 and 2020 to provide the estimates of residue burning using the most recently available yield data.

The potential dry matter production of biochar across India,  $M_{biochar}$  (t y<sup>-1</sup>), was then calculated as



**Table 5**  
Parameters and data used to calculate crop residues burnt in India.

Crop	Residue to crop product ratio <sup>a</sup>	Dry matter fraction <sup>a</sup>	Fraction burnt <sup>a</sup>	Typical carbon content (kg per 100 kg dry matter) <sup>b</sup>	Area (× 10 <sup>6</sup> ha) <sup>c</sup>			Yield (t ha <sup>-1</sup> ) <sup>c</sup>			Total yield (× 10 <sup>6</sup> t y <sup>-1</sup> )		
					2018	2019	2020	2018	2019	2020	2018	2019	2020
Barley	1.7	0.85	0.25	47	0.66	0.58	0.62	2.7	2.8	2.8	1.78	1.63	1.72
Groundnuts, with shell	2	0.8	0.1	50	4.89	4.89	4.89	1.9	1.4	1.6	9.25	6.95	7.97
Maize	1.5	0.88	0.25	47	9.38	9.38	9.38	3.1	3.1	3.1	28.75	28.80	28.68
Millet	1.7	0.85	0.25	47	9.22	9.22	9.22	1.3	1.2	1.3	11.63	11.17	11.86
Rice, paddy	1.5	0.86	0.25	41	44.16	44.16	44.16	4.0	4.1	4.0	174.72	179.17	174.96
Seed cotton	3	0.8	0.1	50	12.35	12.35	12.35	1.2	1.2	1.4	14.66	15.25	17.02
Sorghum	1.7	0.85	0.25	47	5.02	5.02	5.02	1.0	0.8	0.9	4.80	4.27	4.36
Sugar cane	0.4	0.88	0.25	50	4.74	4.74	4.74	80.2	80.1	77.3	379.90	379.46	366.40
Wheat	1.75	0.8	0.25	49	29.65	29.65	29.65	3.4	3.5	3.4	99.87	104.77	101.73
Typical carbon content weighted according to total yield across India (kg per 100 kg dry matter)											47.55	47.51	47.51

Source: <sup>a</sup> Sahu et al. (2021); <sup>b</sup> Venkatramanan et al. (2021); <sup>c</sup> FAO (2022).

$$M_{\text{biochar}} = (M_{\text{burnt}} \times P_{C,\text{crop}} \times P_{C,\text{ret}}) / (100 \times P_{C,\text{biochar}}) \quad (2)$$

where  $P_{C,\text{crop}}$  is the C content of the crop residues previously burnt (assumed to be 47.5 kg per 100 kg dry matter, Table 5),  $P_{C,\text{ret}}$  is the percentage of C in the feedstock that is retained in biochar on pyrolysis (assumed to be 20 to 50% after Bruun et al., 2011), and  $P_{C,\text{biochar}}$  is the C content of biochar (assumed to be 50 to 90% after Jassal et al., 2015).

### 2.2. Sources of nitrogen to enrich biochar

Only wasted N that can easily be captured using biochar was considered as a potential source of N to enrich the biochar produced. This assumption was used so that the sorption of this N on biochar would represent a real saving in N, not just an alternative way to deliver synthetic fertiliser N to the crop.

The sources of wasted N for sorption on biochar,  $N_{\text{waste}}$  (t y<sup>-1</sup>), were estimated from N that could readily be captured from both industrial processes,  $N_{\text{ind}}$  (t y<sup>-1</sup>), and losses on the farm,  $N_{\text{farm}}$  (t y<sup>-1</sup>),

$$N_{\text{waste}} = N_{\text{ind}} + N_{\text{farm}} \quad (3)$$

For industrial scale enrichment of biochar, major losses of N occur as NO<sub>x</sub> from combustion sources associated with transport, and the energy and industry sectors (Sharma et al., 2017). Emissions from transport

would not be easy to capture on biochar, so only the emissions from energy and industry were included. Adsorption of NO emissions in energy and industry contexts has been demonstrated, for example, on a fixed-bed reactor containing biochar (Anthonysamy et al., 2022). The N available for biochar enrichment from industry was calculated as

$$N_{\text{ind}} = P_{\text{e\&i,NO}_x} / 100 \times N_{\text{NO}_x} \quad (4)$$

where  $P_{\text{e\&i,NO}_x}$  is the percentage of the NO<sub>x</sub> gases produced in India that originate from the energy and industry sectors, and  $N_{\text{NO}_x}$  is the annual emissions of N in NO<sub>x</sub> gases in India (t y<sup>-1</sup>);  $N_{\text{NO}_x}$  was estimated by Sutton et al. (2017) using data from EDGAR (2016), to be a total of 3.2 × 10<sup>6</sup> t y<sup>-1</sup> in 2015. Gurjar et al. (2017) estimated that ~32% of the NO<sub>x</sub> produced in India originated from the transport sector in 2015, so by difference,  $P_{\text{e\&i,NO}_x}$  was assumed to be ~68%. The energy and industry sectors also produce lower emissions of nitrous oxide, nitric acid and adipic acid that could potentially be captured on biochar but these have not been included here as data on uptake by biochar is limited (Sharma et al., 2017).

At farm scale, losses of N occur in many different ways (Fig. 5). In cropping systems, there has been a focus on reducing losses of N by leaching of nitrate (as well as NH<sub>4</sub><sup>+</sup> from some soils with low cation exchange capacities), denitrification of nitrous oxide and nitrogen gas, and volatilisation of NH<sub>4</sub><sup>+</sup>. However, while these losses could be reduced as a result of application of the biochar to the soil, they are not readily

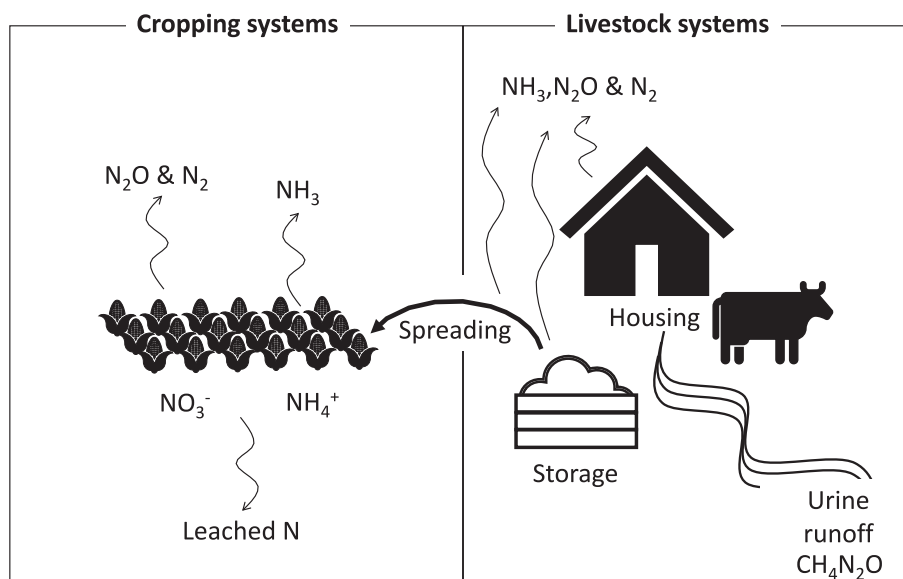


Fig. 5. Schematic representation of the key losses of nitrogen from the whole farm system.

captured on the biochar before application, so are not included as potential sources of N for biochar enrichment. By contrast, the large losses of N from livestock systems, which occur during housing, manure storage and spreading (Reidy et al., 2009), have high potential to be captured on biochar, so are the focus of the estimates of N available for sorption on the farm (Jassal et al., 2015; Smith et al., 2019; Keskinen et al., 2021). Losses of N from animal systems depend on the urine to faeces composition of the manure, management of the manure and the environmental conditions (Bussink and Oenema, 1998). Following enzymatic hydrolysis to  $\text{NH}_4^+$ , urea is the main component of cattle urine responsible for losses by  $\text{NH}_3$  volatilisation (Whitehead et al., 1989); in poultry, the main N source is uric acid (Valli et al., 1991). By contrast, very little N is contained in cattle faeces as  $\text{NH}_4^+$  or urea (Ettala and Kreula, 1979), or in a form that is rapidly released by decomposition (Van Faassen and Van Dijk, 1987). Therefore, losses of N from the faeces are relatively low, resulting in the proportion of urine to faeces being a key determinant of N loss in cattle manure.

In China, a detailed analysis of nutrient flows through the “manure management chain” of feed intake, excretion, housing, storage, treatment and application, suggested that 78% of the excreted N was lost and 39% of these losses occurred as  $\text{NH}_3$  from housing and during storage (~30% of excreted N lost from housing and storage), while 30–73% were due to easily avoidable direct discharge (23–57% of excreted N lost by direct discharge) (Bai et al., 2019). In the European Union, an average of 65% of N excreted in animal housing was estimated to be recycled back to the land (Oenema et al., 2007), leaving a remarkably similar level of N loss from housing and during storage (35% compared to ~30% in China). Traditional dairy production systems in China were found to mainly lose N as  $\text{NH}_3$ , whereas losses from large industrial dairy feedlots with no land for disposal of manure were mainly by discharge of manure into water bodies (Bai et al., 2019). Runoff of urine from animal housing has also been observed to be a major loss process in villages in Northern India; for example, Ditzler et al. (2018) observed that in Nainital District, Uttarakhand, animals were kept in uncovered yards without bedding on bare soil or stones, with few farmers using gutters to collect urine. Therefore, there is high potential to use biochar in housing to reduce  $\text{NH}_3$  volatilisation, to capture discharged urine, or as an additive to manure storage heaps to reduce losses during storage and spreading of the manure.

The amount of N available to enrich biochar at the farm scale ( $N_{\text{farm}}$ ) was therefore estimated from the N excreted by livestock across India multiplied by the proportion that is typically lost from the farm,

$$N_{\text{farm}} = \left( \frac{365}{1000} \right) \times \sum_{\text{ani}} \left( \frac{x_{\text{ani}}}{1000} \times m_{\text{ani}} \times n_{\text{ani}} \times p_{\text{loss,ani}} \right) \quad (5)$$

where for a specific animal type,  $\text{ani}$ ,  $x_{\text{ani}}$  is the N excretion rate (kg per 1000 kg animal per day),  $m_{\text{ani}}$  is the typical mass (kg per animal),  $n_{\text{ani}}$  is the number of head across India and  $p_{\text{loss, ani}}$  is the proportion of the excreted N that is lost. This was summed across five livestock categories (cows, buffalos, sheep, goats and pigs) with data for  $x_{\text{ani}}$  and  $m_{\text{ani}}$  obtained from IPCC (2019), for  $n_{\text{ani}}$  obtained from FAO (2022). In the absence of better data for India, the proportion of excreted N lost ( $p_{\text{loss, ani}}$ ) was assumed to be 0.3 after Bai et al. (2019); if the proportion of urine to faeces in the manure is ~0.6 (Vaddella et al., 2010), this is equivalent to a loss of approximately 50% of the urine excreted.

### 2.3. Potential sorption of nitrogen on biochar

The actual potential for N sorption,  $N_{\text{sorb}}$  ( $\text{t y}^{-1}$ ), was calculated as the minimum of the potential sorption limited only by N available,  $N_{\text{sorb, NLim}}$  ( $\text{t y}^{-1}$ ), and potential N sorption limited only by the available biochar ( $N_{\text{sorb, BC}}$ ),

$$N_{\text{sorb}} = \min(N_{\text{sorb, NLim}}, N_{\text{sorb, BC}}) \quad (6)$$

As discussed in the previous section, the N limited sorption ( $N_{\text{sorb, NLim}}$ )

was assumed to be equivalent to the amount of wasted N that can be readily captured from industrial and farm processes ( $N_{\text{waste}}$ , Eqs. (3–5)).

The potential of the biochar to sorb N ( $N_{\text{sorb, BC}}$ ) was calculated as

$$N_{\text{sorb, BC}} = P_{\text{sorb}} \times M_{\text{BC}}/100 \quad (7)$$

where  $M_{\text{BC}}$  is the potential production of biochar across India (Eq. (2)), and  $P_{\text{sorb}}$  is the rate of N sorption onto the biochar (g N per 100 g biochar).

Available evidence for sorption of N on biochars produced from crop residues ( $P_{\text{sorb}}$ ) is limited. Karim et al. (2022) reviewed nutrient enrichment of biochars and found most evidence for biochars produced from wood (Fig. 6). In general, biochar produced at higher temperatures absorbed less N than lower temperature biochars from the same feedstock, although this was not always the case (e.g. Takaya et al., 2016). Kizito et al. (2015) observed similar sorption of N on rice husk biochar as on wood biochar. Jassal et al. (2015) investigated sorption and release of N on biochars produced from poultry litter with an initial C:N ratio of 14 and softwood chips of spruce, pine and fir with a C:N ratio of 470. They measured similar N sorption of approximately 5 g N per 100 g biochar over this wide range of feedstock compositions and over different pyrolysis temperatures (400–600 °C), suggesting that this same N sorption rate could be used for biochars produced from crop residues. The rate of N sorption on the biochars was not related to the H:C ratio which, because the H:C ratio is indicative of the number of functional groups on the biochar, suggests sorption was mostly by physical processes. This was also suggested by the observed sorption of  $\text{NH}_4^+$ -N that greatly exceeded the cation exchange capacity of the biochar. Despite this, the immediate release of N when extracted with 1 M KCl was only 0.02–0.04 g N per 100 g biochar, suggesting that N is held deep within the pores of the biochar, and that the enriched biochar could provide an effective slow release fertiliser as the sorbed N diffuses out of the biochar structure; note the efficacy of the biochar as a slow release fertiliser needs to be confirmed by direct crop trials. The average rate of N sorption observed in the papers reviewed was 6 g per 100 g biochar with a standard error of 1.1 g per 100 g biochar (Fig. 6). Anthonysamy et al. (2022) investigated adsorption of nitric oxide (NO) on biochar derived from rubber seed shells; NO constituting almost 90% of the  $\text{NO}_x$  gases produced in the energy and industry sectors (Chen et al., 2018). Under the most suitable operating conditions, they achieved NO sorption of 8.167 g per 100 g biochar, which is within the 95% confidence interval of the average rate of sorption observed for  $\text{NH}_4^+$ -N. Therefore, for simplicity for sorption of both  $\text{NH}_4^+$  and  $\text{NO}_x$ , a rate of  $P_{\text{sorb}} = 6 (\pm 1.1)$  g per 100 g biochar was used.

### 2.4. Losses of nitrogen during pyrolysis

The losses of N during pyrolysis of crop residues was estimated from the proportion of the N in the crop residues that is lost on pyrolysis ( $P_{\text{Nloss, py}}$ ), the N content of the residues of major crop residues burnt across India in 2020,  $P_{\text{N, res, crop}}$  (g per 100 g crop residues), and the dry matter content of the crop residues burnt,  $M_{\text{burnt, crop}}$  ( $\text{t y}^{-1}$ ),

$$N_{\text{loss, BC}} = P_{\text{Nloss, py}} \times \sum_{\text{crop}} \left( M_{\text{burnt, crop}} \times P_{\text{N, res, crop}}/100 \right) \quad (8)$$

The proportion of the N in the crop residues that is lost on pyrolysis ( $P_{\text{Nloss, py}}$ ) was assumed to be 0.95–1.0 after Bruun et al. (2011). The dry matter contents for the residues of the major crops in India ( $M_{\text{burnt, crop}}$ ) were calculated using the approach laid out in Eq. (1) and data presented in Table 5. The minimum and maximum N contents assumed for the crop residues burnt ( $P_{\text{N, res, i}}$ ) were obtained from a review of the literature; the minimum and maximum values and data sources are presented in Table 6.

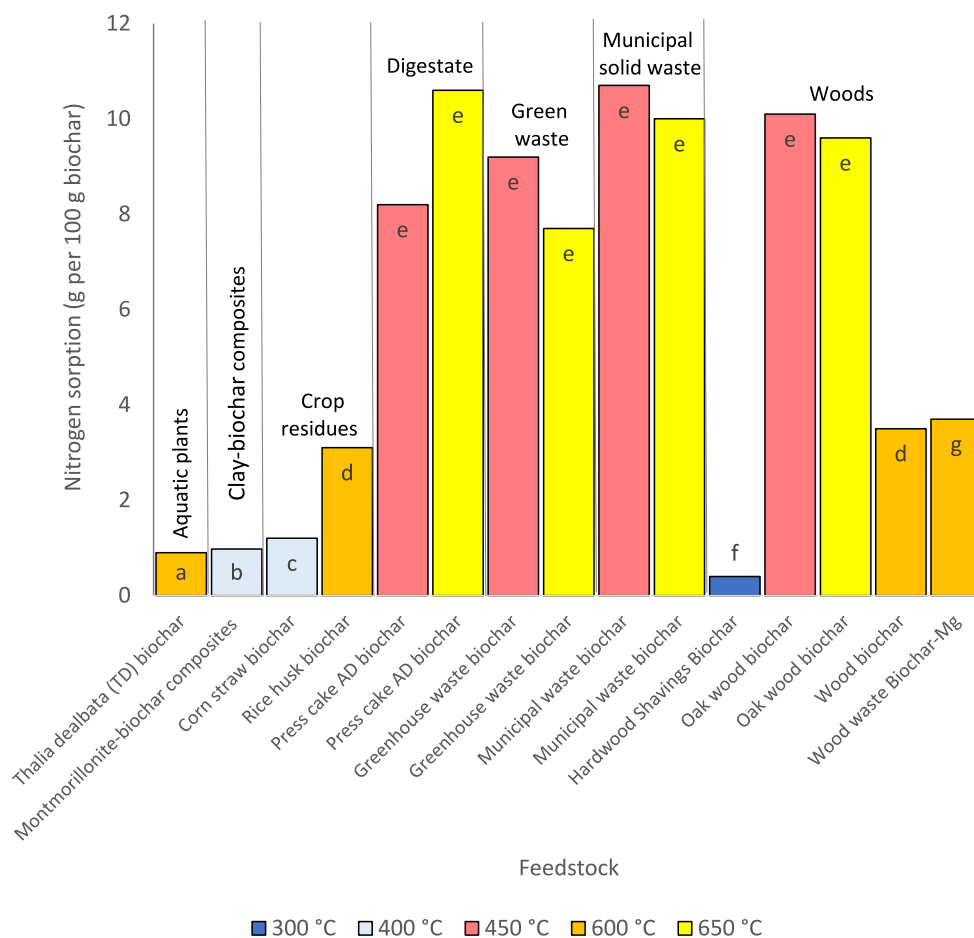


Fig. 6. Nitrogen sorption on biochars derived from different feedstocks and at different pyrolysis temperatures. Derived from data reviewed by Karim et al. (2022). Sources: <sup>a</sup> Zeng et al. (2013); <sup>b</sup> Chen et al. (2017); <sup>c</sup> Gai et al. (2014); <sup>d</sup> Kizito et al. (2015); <sup>e</sup> Takaya et al.; <sup>f</sup> Sarkhot et al. (2013); <sup>g</sup> Xu et al. (2018).

Table 6

Nitrogen (N) contents of crop residues burnt across India. Notes: <sup>a</sup> Nitrogen and sulphur application trials on barley in Rajasthan (Parashar et al., 2020); <sup>b</sup> N response trials for multi-cut barley in Rajasthan (Meena et al., 2011); <sup>c</sup> Experiments on different treatments of N, P, K and S in groundnuts – haulms (Vala et al., 2018); <sup>d</sup> Impact of integrated nutrient management on maize/wheat rotations in Rajasthan (Priyanka et al., 2019); <sup>e</sup> Elemental analysis of maize straw from India (Sandhu et al., 2019); <sup>f</sup> Trials with millet in Jodhpur, India (Aggarwal et al., 1997); <sup>g</sup> Field experiment with finger millet in Bangalore (Prashanth et al., 2020); <sup>h</sup> Values reviewed from a range of rice trials in tropics (Yang and Zhang, 2010); <sup>i</sup> Review of rice straw (Van Hung et al., 2020); <sup>j</sup> Nitrogen and irrigation trials on seed cotton in Rajasthan (Kumar et al., 2022); <sup>k</sup> N response trials on sorghum in Pantancheru, India (Bollam et al., 2021); <sup>l</sup> Values for sugar cane in Brazil (Fortes et al., 2013); <sup>m</sup> Review of wheat trials across South Asia (Blümmel et al., 2019).

Crop	Minimum N content (%)	Maximum N content (%)
Barley	0.62 a, b	0.77 b
Groundnuts, with shell	1.63 c	1.90 c
Maize	0.37 d	0.60 e
Millet	0.51 f	1.18 g
Rice, paddy	0.60 h	1.71 i
Seed cotton	0.48 j	1.33 j
Sorghum	0.32 k	0.96 k
Sugar cane	0.26 l	1.28 l
Wheat	0.30 d	0.72 m

2.5. Net impact on nitrogen waste

The potential net impact of biochar on N waste was determined for business as usual, as well as for a future improved residue management regime. The potential impact on the business as usual losses of N of producing biochar from the crop residues that are currently burnt and using them to capture wasted N from industrial and farm processes,  $\Delta N_{waste,BAU} (t y^{-1})$ , was obtained directly from the sorption potential on the biochar available ( $N_{sorb}$ ).

$$\Delta N_{waste,BAU} = -N_{sorb} \tag{9}$$

The N losses occurring during pyrolysis were assumed here to be equivalent to the N losses occurring during burning of crop residues in the fields, and so were not included in the equation. This underestimates the potential benefits of applying biochar as it does not account for any reduction in leaching and gaseous losses of N that may occur from the soil itself (e.g. Novak et al., 2009a; Jeffery et al., 2011; Crane-Droesch et al., 2013; Mukherjee and Lal, 2014; Kanthle et al., 2016). These potential benefits were neglected as they are highly soil, climate and management specific, and so would require a more spatially explicit data and process-based simulations to estimate any change in N losses from the soil.

The future improved best case residue management regime was assumed to be a system in which crop residues are no longer burnt in the field, and are instead (successfully) managed to retain N in the farming system. The changes in N under this regime,  $\Delta N_{waste,best} (t y^{-1})$  was estimated by reducing the beneficial impacts of biochar on wasted N by the amount of N lost during pyrolysis ( $N_{loss,BC}$ )



$$\Delta N_{\text{waste,best}} = N_{\text{loss,BC}} - N_{\text{sorb}} \tag{10}$$

This regime might involve recycling of crop residues by composting or anaerobic digestion (Manzoni et al., 2010; Smith et al., 2014a, 2014b; Sfez et al., 2017). This equation assumes similar N use efficiency for N sorbed on biochar as for N applied as decomposed crop residues, which is not likely to be the case as significant amounts of N can be lost during production and spreading of compost and bioslurry. It also neglects the potential for a reduction in wasted N by direct capture of N runoff, for instance by using straw as bedding to reduce runoff and fix urea N (Yadav and Virk, 1994), or by collecting urine and incorporating it in a compost heap. The potential for these different management options to reduce wasted N has been omitted as it is again highly site and management specific.

### 3. Results and discussion

#### 3.1. Potential production of biochar from the crop residues that are currently burnt

The dry matter content of crop residues burnt in 2018–2020 ( $M_{\text{burnt}}$ ) estimated using data from FAO (2022) are shown in Table 7. The estimate of total residues burnt in 2018 corresponded closely with the 2018 estimate of Sahu et al. (2021) ( $1.46 \times 10^8 \text{ t y}^{-1}$  compared to  $1.52 \times 10^8 \text{ t y}^{-1}$ ), giving confidence in the use the FAO data to estimate more recent crop residue burning in 2019 and 2020. The total crop residues burnt in 2019 and 2020 are  $1.48 \times 10^8 \text{ t y}^{-1}$  and  $1.46 \times 10^8 \text{ t y}^{-1}$ , respectively. This results in an estimated potential production of biochar from crop residues that were burnt in 2020 ( $M_{\text{biochar}}$ ) of between  $(1.5\text{--}6.9) \times 10^7 \text{ t y}^{-1}$ .

The estimate of total crop residues burnt provided by Sahu et al. (2021) for 2018 ( $1.52 \times 10^8 \text{ t y}^{-1}$ ) was higher than estimated in earlier studies. Ravindra et al. (2019) estimated that crop residues burnt in the fields in India increased from  $(9.0 \times 10^7) \text{ t y}^{-1}$  in 2003–04 to  $(1.77 \times 10^8) \text{ t y}^{-1}$  in 2016–17, and amounted to 24% of the crop residues produced. This is in agreement with the estimates of Jain et al. (2014) for 2008–09 and Venkatramanan et al. (2021) for 2017, but is lower than the value estimated by Venkataraman et al. (2006) for 2003 (Fig. 7). The estimate of Sahu et al. (2021) used similar methodology to the earlier studies, but was based on higher resolution data, so is expected to be more accurate. Using the lower estimate of crop residue burning for 2017 provided by Venkatramanan et al. (2021) ( $M_{\text{burnt}} = 1.16 \times 10^8 \text{ t y}^{-1}$ ) would give a lower limit for potential biochar production of  $M_{\text{biochar}} = (1.2\text{--}5.5) \times 10^7 \text{ t y}^{-1}$  (Table 7).

In recent years, the government of India has introduced measures to reduce burning of residues in the fields, mainly centred around use of

crop residues in energy production (Bhuvaneshwari et al., 2019). Early reports from the Energy and Resources Institute, a not-for-profit policy research institute in India, suggest that these have been successful in reducing crop residue burning from the projected levels of  $(1.16\text{--}1.46) \times 10^8 \text{ t y}^{-1}$  to  $(0.72\text{--}1.27) \times 10^8 \text{ t y}^{-1}$  (P. Das, A. Datta and R. Suresh, The Energy and Resources Institute, New Delhi, India, pers. comm.). This would give a range for biochar production of  $M_{\text{biochar}} = (0.8\text{--}6.0) \times 10^7 \text{ t y}^{-1}$  (Table 7). If the reduced burning of crop residues has been achieved by alternative uses for crop residues, then this lower range should be assumed as the crop residues are already committed to other uses.

#### 3.2. Sources of nitrogen to enrich biochar

The N that could be readily captured from industrial processes ( $N_{\text{ind}}$ ) was estimated to be  $2.18 \times 10^6 \text{ t y}^{-1}$ , while  $6.40 \times 10^6 \text{ t y}^{-1}$  was available from loss processes on the farm ( $N_{\text{farm}}$ ) (Table 8). This gives a total potential source of N to enrich biochar ( $N_{\text{waste}}$ ) of  $8.58 \times 10^6 \text{ t y}^{-1}$ , with 75% of the available N coming from the loss processes on the farm and only 25% from the industrial and energy sectors.

#### 3.3. Potential sorption of nitrogen on biochar

The potential sorption N on the biochars that could have been produced from the crop residues burnt in the fields in India in 2020 assuming no limitation in available N ( $N_{\text{sorb, BC}}$ ) ranges from  $(0.6 \text{ to } 5.8) \times 10^6 \text{ t y}^{-1}$  using the projections from the estimates of crop residue burning provided by Sahu et al. (2021) (Table 9). Assuming the reported reduction in burning due to government initiatives, the range is reduced to  $(0.3 \text{ to } 5.0) \times 10^6 \text{ t y}^{-1}$  (Table 9). The large range in values is due to the high uncertainty in the amount of N that can be sorbed on the biochars produced by different methods and crops. This highlights the need for further experimental work to better understand the factors controlling sorption of N on biochar. The N limited sorption ( $N_{\text{sorb, NLim}}$ ) is estimated to be  $1.49 \times 10^7 \text{ t y}^{-1}$ . Therefore, the potential for sorption of N on biochars ( $N_{\text{sorb}}$ ) is estimated from the availability of biochar ( $N_{\text{sorb, BC}}$ ) and ranges from  $(0.6 \text{ to } 5.8) \times 10^6 \text{ t y}^{-1}$  or  $(0.3 \text{ to } 5.0) \times 10^6 \text{ t y}^{-1}$ .

#### 3.4. Losses of nitrogen during pyrolysis

The potential losses of N during pyrolysis of the crop residues that were burnt across India in 2020 ( $N_{\text{loss, BC}}$ ) were estimated to range from  $(0.60 \text{ to } 1.84) \times 10^6 \text{ t y}^{-1}$  (Table 10). Assuming the reported government initiatives have resulted in a proportional reduction in crop residues burnt, losses of N on pyrolysis would be reduced to  $(0.30 \text{ to } 1.60) \times 10^6 \text{ t y}^{-1}$ .

**Table 7**

Potential biochar production across India. Note: <sup>a</sup> Feedstock carbon retained in the biochar assumed to be 20 to 50% (Bruun et al., 2011); <sup>b</sup> Carbon content of biochar assumed to be 50–90% (Jassal et al., 2015).

Symbol	Sahu et al. (2021)	FAO (2022)			Venkatramanan et al. (2021)	Reported reduction in burning due to government initiatives	
	Year	2018	2018	2019	2020	2020	2020
Typical carbon content weighted according to total yield across India (kg per 100 kg dry matter)							
$P_{c,crop}$	47.55	47.55	47.51	47.51	47.51	47.51	47.51
$M_{burnt}$	1.52	1.46	1.48	1.46	1.16	0.72	1.27
$M_{burnt} \times P_c$							
$crop$							
	Carbon content of crop residues burnt ( $\times 10^8 \text{ t y}^{-1}$ )	0.72	0.69	0.70	0.69	0.55	0.60
	Carbon content of potential biochar production from burnt crop residues <sup>a</sup> ( $\times 10^7 \text{ t y}^{-1}$ )						
	Minimum	1.45	1.39	1.41	1.38	1.10	0.68
	Maximum	3.61	3.47	3.52	3.46	2.76	1.71
$M_{burnt}$	Potential biochar production <sup>b</sup> ( $\times 10^7 \text{ t y}^{-1}$ )						
	Minimum	1.6	1.5	1.6	1.5	1.2	0.8
	Maximum	7.2	6.9	7.0	6.9	5.5	3.4
							6.0

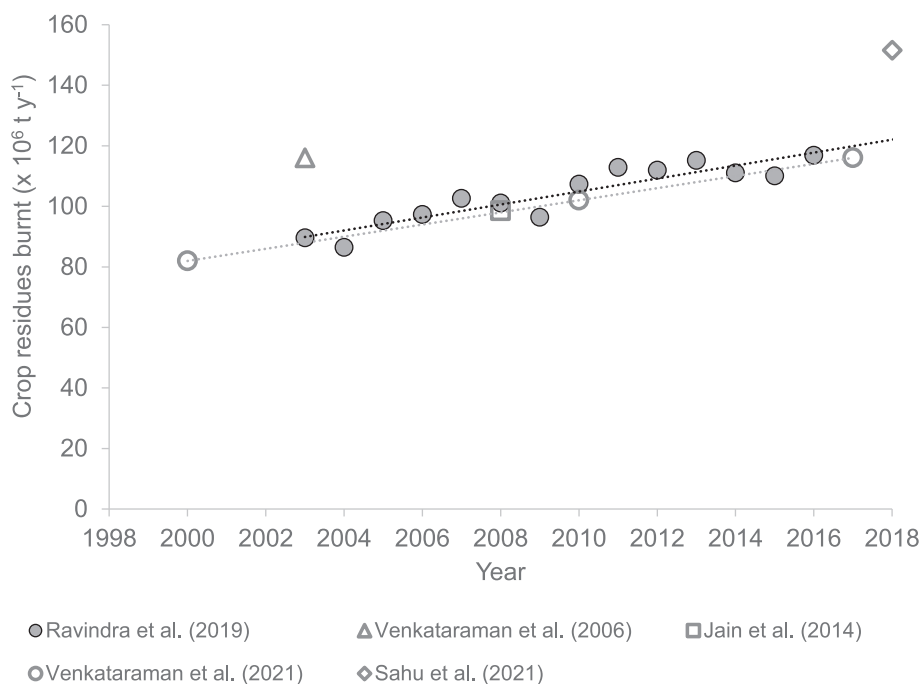


Fig. 7. Crop residues burnt in the fields in India.

Table 8

Estimation of total N wasted from excreted manure across India in 2020. Notes: <sup>a</sup> IPCC (2019); <sup>b</sup> FAO (2022); <sup>c</sup> Calculated according to Eq. (5) assuming the proportion of the excreted N that is lost ( $p_{loss, ani}$ ) is 0.3 (Bai et al., 2019).

Type of livestock	Nitrogen excretion rate, $x_{ani}$ (kg N per 1000 animal mass per day) <sup>a</sup>	Typical mass, $m_{ani}$ (kg per animal) <sup>a</sup>	Number of livestock in India in 2020 <sup>b</sup> , $n_{ani}$	Total N excretion across India ( $\times 10^6$ t $y^{-1}$ )	Total N excretion wasted across India, $N_{farm}$ ( $\times 10^6$ t $y^{-1}$ )
Cattle	0.65	285	194,482,355	13.15	3.95
Buffalos	0.57	321	109,719,011	7.33	2.20
Sheep	0.32	31	68,099,762	0.25	0.07
Goats	0.34	24	150,248,487	0.45	0.13
Pigs	0.595	86	8,852,111	0.16	0.05
Total				21.34	6.40

Table 9

Potential nitrogen (N) sorption on the biochar that could have been produced from the crop residues burnt in 2020. Note: <sup>a</sup> Maximum and minimum values given by the mean value  $\pm 95\%$  confidence interval assuming the mean rate of N sorption on biochar ( $P_{sorb}$ ) is 6 g per 100 g biochar with a standard error of 1.1 g per 100 g biochar.

	Potential biochar production across India ( $M_{BC}$ ) ( $\times 10^7$ t $y^{-1}$ )	Rate of N sorption on biochar ( $P_{sorb}$ ) <sup>a</sup> (g per 100 g biochar)	Potential N sorption ( $N_{sorb, BC}$ ) ( $\times 10^6$ t $y^{-1}$ )
Projections for 2020 from Sahu et al. (2021)			
Minimum	1.5	3.7	0.6
Maximum	6.9	8.3	5.8
Reported reduction in burning due to government initiatives			
Minimum	0.8	3.7	0.3
Maximum	6.0	8.3	5.0

Table 10

Potential nitrogen (N) loss during pyrolysis of the crop residues burnt across India in 2020.

Crop	Minimum N loss ( $\times 10^6$ t $y^{-1}$ )	Maximum N loss ( $\times 10^6$ t $y^{-1}$ )
Barley	0.00	0.00
Groundnuts, with shell	0.02	0.02
Maize	0.03	0.06
Millet	0.02	0.05
Rice, paddy	0.32	0.96
Seed cotton	0.02	0.05
Sorghum	0.00	0.02
Sugar cane	0.08	0.41
Wheat	0.10	0.26
Total	0.60	1.84

### 3.5. Net impact on nitrogen waste

The net impact on N waste of using the crop residues that were estimated to have been burnt in 2020 to produce biochar and to the sorb N lost by industrial and farming processes ( $\Delta N_{waste, BAU}$ ) was estimated to be a reduction in N waste of  $(0.6 \text{ to } 5.8) \times 10^6 \text{ t } y^{-1}$  (Table 9). This results in 7–67% of the wasted N being captured on the biochar and is equivalent to 3–31% of the fertiliser N used across India in 2020 (FAO, 2022). With reported reductions in crop residues burnt due to government initiatives, this would be reduced to  $(0.3 \text{ to } 5.0) \times 10^6 \text{ t } y^{-1}$  (Table 9), which is equivalent to lower rate of 3–59% of the wasted N being captured on the biochar and 1–27% of the fertiliser N use. The net impact on N waste compared to a potential future improved crop residue management regime ( $\Delta N_{waste, best}$ ) was estimated to range from  $-5.1 \times 10^6 \text{ t } y^{-1}$  (a 60% reduction in N waste at the higher estimate of biochar production) to  $1.3 \times 10^6 \text{ t } y^{-1}$  (a 15% increase in N wasted at the lower estimate of biochar production). With the reduced rate of residue burning reported due to government initiatives, this would be  $-4.4 \times 10^6 \text{ t } y^{-1}$  (a 52% reduction in N waste) to  $1.6 \times 10^6 \text{ t } y^{-1}$  (a 18% increase in N wasted).

Even without considering the positive impacts of biochar on nutrient retention in the soil, capturing up to 67% of the N lost would amount to

31% of the 2020 fertiliser use, which given an average price of urea-N between 2010 and 2019 of 625 \$ t<sup>-1</sup> (World Bank, 2022) is equivalent to an economic saving of up to 1.68 × 10<sup>9</sup> \$ y<sup>-1</sup> or 0.6% of current GDP from agriculture. Using the inflated 2022 fertiliser prices (1982 \$ t<sup>-1</sup>; World Bank, 2022), this would increase to 2.0% of the GDP from agriculture. Furthermore, capturing N before it is emitted to the environment could have profound impacts on air and water quality.

However, biochar is not the only option for reducing in-field burning of crop residues and retaining more C and N in the soil; other ways to do this include composting and anaerobic digestion. For the lowest level of N sorption on the biochar and the most effective retention of N by composting or anaerobic digestion, these methods could potentially capture up to 15% more wasted N than could be captured on biochar. This is due to avoiding the gaseous N losses that occur during pyrolysis, while retaining similar amounts of wasted N as the biochar. Losses of N during pyrolysis could be reduced in industrial processes by using biochar scrubbers to capture the emitted N, but at the household scale this would be difficult to achieve. The N retained during composting depends on management of the composting process; effective absorption of urine on straw and regular removal of bedding from animal housing, covering and layering of heaps to reduce volatilisation of NH<sub>3</sub>, and effective maturation of the compost so that it contains low levels of NH<sub>4</sub><sup>+</sup> and nitrate (Bai et al., 2013). Anaerobic digestion produces a bioslurry that is high in NH<sub>4</sub><sup>+</sup> (Smith et al., 2014a, 2014b), so post-processing is needed to reduce NH<sub>3</sub> losses by immobilising this N before applying it to crops, such as by mixing it into compost heaps or by combining it with biochar. Therefore, careful management is needed to achieve the N savings possible with both composting and anaerobic digestion. It is, therefore, likely that biochar will provide a more reliable way of capturing the wasted N.

Whether biochar production is the best option for an individual farmer also depends on the condition of the soil on their farm, the amount of work needed to produce biochar compared to the other recycling methods and the possible economic benefits. In a highly weathered soil with a very low cation exchange capacity and low organic matter content, annual use of biochar to deliver N to the crop will provide a more rapid increase in soil C and cation exchange capacity than other recycling methods because the C applied in biochar is more recalcitrant (Smith et al., 2014b) and highly charged (Novak et al., 2009a, 2009b). Therefore, on highly degraded soils, farmers applying biochar are likely to see more immediate improvements in crop production and economic benefits which might encourage them to provide the labour needed to collect crop residues from the fields and to produce N enriched biochar.

Other incentives to reduce burning of crop residues in the fields might include the sale of straw for centralised production of biochar using N losses from the energy and industry sectors to enrich the biochar. Industrial loss processes account for 25% of the N available for capture on biochar, so only 38% of the maximum N sorption potential could be achieved using industrial sources alone. One option would be to also provide centralised collection of urine for biochar enrichment, so providing farmers with a further source of income and incentivising reduced N waste.

Within household use of crop residues for household energy production and use of the biochar output to absorb urine in animal housing to produce an effective organic fertiliser directly on the farm could also encourage reduced in-field burning of crop residues. Pyrolysis cookstoves for household use have the advantage of being significantly cheaper than anaerobic digesters and not requiring additional water for the digestion process (Smith et al., 2019). However, while emissions of PM<sub>2.5</sub> from pyrolysis cookstoves are significantly reduced, they remain at just over 50% of emissions from traditional cooking methods (MacCarty et al., 2010) while emissions from biogas and LPG cookstoves are likely to show larger reductions of 66% to 99% (Semple et al., 2014). Therefore, conversion to pyrolysis cookstoves would only be recommended for households that are reliant on traditional cooking methods.

In addition, production of bio-oils during the pyrolysis (Kim, 2015; Leng et al., 2020; Sakhiya et al., 2020) has potential to result in environmental damage if alternative household uses or safe disposal methods are not found.

#### 4. Conclusions

Biochars produced from the crop residues that are estimated to have been burnt in the fields in 2020 could have been used to capture up to 67% of the N lost as NO<sub>x</sub> and NH<sub>3</sub> from the energy, industry and farming sectors. This is equivalent to 31% of the N that was applied as fertilisers. Therefore, biochars are an important tool in the reduction of N waste from farming systems, and could contribute directly to achieving the United Nations stated aim of reducing N waste by 50% by 2030 (UNEP, 2019). However, the benefits of N capture on biochars could be much lower (as low as 3% N capture and 1% of applied N fertilisers) if the rate of N capture is low, government initiatives provide alternative uses for crop residues, and alternative uses, such as composting or anaerobic digestion, are successful in capturing wasted N. Nevertheless, using biochar as a method to deliver wasted N to crops is likely to sequester more C than alternative uses because pyrolyzed C is highly recalcitrant. It is also likely to be a more reliable method of capturing N emissions and delivering N to crops because emissions of NH<sub>3</sub> during storage and spreading of compost or bioslurry can be high. Therefore, even if alternative uses of crop residues are favoured by farmers, it is recommended that N sorption on biochar should be part of the process, whether it is by direct capture of N from urine and industrial NO<sub>x</sub> emissions or by mixing of biochar with compost or the NH<sub>4</sub><sup>+</sup> rich bioslurry produced by anaerobic digestion. If produced centrally, purchases of wastes for biochar production have the advantage of also providing an economic incentive to crop residue and urine collection instead of disposal. If practiced at farm level, direct use could be made of the energy released during pyrolysis, but this should not be used to replace cleaner forms of household energy and further research is needed to find safe ways to use or dispose of any bio-oils produced. Further experimental trials are needed to fully characterise the recalcitrance and sorptive properties of biochar produced under different conditions and from different crop residues. This is essential to reduce the uncertainty in the estimated potential to reduce wasted N and sequester C. Evidence is also needed on the optimum conditions for N sorption from both industrial and agricultural sources and the release of N from the biochar when it is applied to the soil. These processes need to be fully characterised so that detailed recipes for production and enrichment of biochar and recommendations for rates of application to crops can be provided.

#### Declaration of Competing Interest

None.

#### Data availability

No data was used for the research described in the article.

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