Sustainable biochar to mitigate global climate change: Supplementary information

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1 Supplementary Note

The term “biochar” first appeared in the peer-reviewed scientific literature in 1999 where it was used to describe an activated carbon prepared from sorghum grain for use in a reverse-burn gasification (Chem Char) process for treating hazardous waste[43]. The lead author of this paper, Bapat, recalls that they invented the term to differentiate the sorghum-based material from coal-derived activated carbon used in earlier implementations of the process; they had intended to develop similar materials from other grains[44]. For several years thereafter biochar was used primarily in the bioenergy literature to describe a charcoal prepared from various crop residues for use as fuel. The technological concept of using biochar on a large scale as a climate-change mitigation approach stems from two papers published independently in 1993 (refs 45,46), well before the term itself was invented. Ref. 45 clearly described the large reservoir of carbon stored in soils, the historical use of charcoal by indigenous peoples in the Amazon region as a soil amendment with multiple benefits, and its potential as a climate-change mitigation strategy. Ref. 46, on the other hand, focused on the climate-change mitigation aspects from large-scale industrial production and burial of charcoal in landfills; he did not consider its use as a soil amendment *per se*. To our knowledge, the matching of the term “biochar” with the climate-change mitigation concept did not occur until March of 2005 in a presentation by Lehmann entitled “Bio-char sequestration in soil: A new frontier”[47] which was followed by two publications in 2006 that used the term with its current meaning[48,49]. Lehmann relates that the term stemmed from a discussion he had with Peter Read while working on the revisions to one of these publications and preparing the 2005 presentation.

2 Supplementary Methods

2.1 Biomass Feedstock Availability

We consider three scenarios for global biomass availability: scenario ‘Alpha’, which involves the use of biomass available with little change to current practices; scenario ‘Beta’, which assumes some legislation or incentives to promote both sustainable land-use practices and reduced contamination of biomass waste streams; and a ‘Maximum Sustainable Technical Potential’ (MSTP) scenario representing the maximum biochar production technically achievable within our sustainability constraints if humanity were to strive to do their utmost to mitigate climate change. In assessing the biomass availability within these scenarios, no account has been taken of the impact of climate change on biomass availability. Rather, this analysis is premised upon the assumption that biochar production will form part of a suite of mitigation measures that succeed in controlling climate change to within reasonable bounds. If other strong mitigation measures are not also put in place, reductions in global crop yields may reduce the availability of feedstock from agricultural residues. Other feedstocks such as timber from tree mortalities may however increase. The net impact on biomass availability under various warming scenarios is beyond the scope of this study.

Nor have we included the effects of population change in this assessment for the reason that its main effect on biomass availability is likely to be the clearance of land to increase food production for a growing population, resulting in an increase in production of agricultural residues and thus greater biomass availability. However, for the reasons given in section 2.1.6, land clearance to provide biomass feedstock is not a viable climate mitigation strategy. We have therefore chosen to investigate what the potential for biochar production is assuming that no land clearance is used to provide feedstock. Since, in practice, it will be very difficult to distinguish land-clearance driven by population growth from other drivers, the best way to maintain clarity that this assessment includes only the potential for biochar without land clearance is to omit any potential increases in feedstock arising from population growth. While this may lead to a low (conservative) estimate of the availability of agricultural residues, it does provide transparency about how the sustainability principles are applied.

The availability of sustainable biomass feedstocks, whose utilisation does not entail an increase in human appropriation of global net primary productivity, is broken down by category of feedstock.

2.1.1 Agricultural and forestry residues

World production of crop residues was 3.8 Pg dry matter in 2001[50], 75% of which was from cereals and 8% from sugar cane. Of the remaining 17% of crop residues, most are produced in too small quantities per hectare to remove them without unacceptable risk of soil erosion and loss of soil organic matter[50], or have too high a water content to be well-suited to pyrolysis. Resources such as nut and

peanut shells, sunflower residues and orchard prunings may be of local importance, but together amount to only approximately 1.5% of total residue production and were therefore omitted from this analysis on the basis that they are not of high global significance.

2.1.2 Cereals excluding rice

The fraction of crop residues that may be removed without incurring unacceptable deterioration or loss of soil is highly variable depending on soil, climate, terrain and management regime. In practice, therefore, acceptable levels of extraction need to be assessed on a site-by-site basis[51]. Estimates of the fraction of above-ground cereal crop residues that may be removed without causing undue harm include 25% (ref. 52), 27% (ref. 53), <= 25% in erosion-prone soils (ref. 54), 30% (ref. 51), 25-50% (ref. 55), and 40% with mulch-till and 70% with no-till (ref. 56). Those studies that support higher rates of extraction (refs 55,56) are based on modeling, whereas studies based on field measurements (refs 51-54) tend to support lower levels of extraction. It is also important to note that all of these studies are based on North America; extrapolation to the global situation must therefore be done with some caution. Accordingly, we have assumed maximum 25%, 35% and 45% cereal residue extraction rates in the Alpha, Beta and MSTP scenarios, respectively. Achieving the higher values of residue extraction will require global adoption of best practices for soil conservation, including use of cover crops which can significantly reduce soil erosion[57-61]. Where harvested straw is utilised as animal bedding, it is assumed that it will still subsequently become available for pyrolysis once mixed with manure. The most important use of harvested straw that is not compatible with subsequent pyrolysis at end of life is its use as animal fodder – a practice that is widespread in the developing world but little employed in the developed world (other than cereals grown for silage which are not included in crop residue production). Where the fraction of cereal straw already used for fodder exceeds the maximum extraction rates determined by soil conservation considerations, we have assumed that no further extraction to provide pyrolysis feedstock will be undertaken. The total straw and stover from non-rice cereals that will thus be available for pyrolysis under each scenario is given in Table S1. Once maximum extraction rates and use of straw as fodder are accounted for, the total amounts of straw and stover available for pyrolysis amount to 8, 14 and 20% of total residues generated in the three scenarios, respectively.

2.1.3 Rice

Globally, 890 Tg yr^{-1} of paddy rice straw are produced[50], of which 230 Tg yr^{-1} are used for animal feed[62]. No rice straw is required to be left on the paddy fields for erosion control[63]. In addition to this, 86 Tg yr^{-1} rice hulls are produced and being generally regarded as a waste product[48] may be available as feedstock. Thus, the maximum biomass from rice that is potentially available is 746 Tg yr^{-1}. In the MSTP scenario, we assume that straw used for purposes such as animal bedding and thatch will become available at end of life and that extraction losses will be 10%, to yield 671 Tg DM yr^{-1} available for pyrolysis. In the Alpha and Beta scenarios, we reduce the available amount to 70% and 80%, respectively, of the maximum potentially available.

2.1.4 Sugar Cane

Cane production generates 314 Tg yr^{-1} residues[50]. Of this, approximately 50% is bagasse and 50% field trash[64]. We have assumed that all bagasse may be utilised, since bagasse that is currently utilised to provide heat and power to sugar mills could instead be used for cogeneration of biochar and energy, thus using a greater fraction of bagasse to provide the mill's energy requirements while also generating less waste bagasse. Until recently, field residues were commonly burnt. However, sugar cane production is rapidly adopting the practice of green harvesting with trash left as a mulch or tilled in, with around 50% of this green-harvested field trash being recoverable in 50% of fields with currently available equipment[64]. We have accordingly assumed 25% field residue utilisation in the Alpha scenario rising to 50% and 75% in the Beta and MSTP scenarios respectively, on the basis that improved technology and soil conservation practices are likely to permit extraction rates greater than the present 50% up to a maximum agronomically desireable extraction rate of 75% (ref. 64). The amounts of sugar cane biomass potentially available for the three scenarios thus total 196, 239, and 275 Tg DM yr^{-1} for the Alpha, Beta, and MSTP scenarios, respectively.

2.1.5 Manure

Annual production of cattle and pig manures are in the order of 470 Tg C and 34 Tg C respectively[65]. Assuming mean carbon contents of 29.9% and 37.6% (fraction of dry matter), this gives 1570 and 90 Tg DM yr^{-1}, respectively. We approximate annual poultry manure production as 134 Tg, based on 121 Tg from chickens[62] divided by 0.9 – the fraction of poultry that is chickens (FAOSTAT 2007[66]). Since turkeys and geese produce more manure per capita than chickens, this is probably a slightly conservative estimate.

Estimates of the fraction of cattle manure that might be available for bioenergy feedstock range from 12.5-25%[67]. Accordingly, for cattle manure we have used availability factors of 12.5%, 18.75% and 25% to estimate total amounts available of 196, 294, and 392 Tg DM yr^{-1} for the Alpha, Beta and MSTP scenarios, respectively. For pigs and poultry, as they are kept mainly in housing or confined areas, we have assumed higher availability factors of 50, 70, and 90% for the Alpha, Beta and MSTP scenarios, respectively. Total amounts of pig manure available are thus 45, 63, and 81 Tg DM yr^{-1} for the Alpha, Beta and MSTP scenarios, respectively. Total amounts of poultry manure available are 67, 94, and 121 Tg DM yr^{-1} for the Alpha, Beta and MSTP scenarios, respectively.

2.1.6 Biomass crops

The potential for growing biomass crops to provide biochar feedstock is limited by land availability constraints including considerations of conservation of habitat, ecosystem services and ecosystem-stored carbon. Conversion of currently unmanaged land or grassland releases as CO_2 carbon that was stored in plants and soil. If the purpose of the land conversion is to grow feedstock for biochar or bioenergy, this CO_2 emission represents a 'carbon debt' that may be repaid over time by subsequent reduced or avoided emissions. The time taken until subsequent reductions in CO_2 emissions exceed the initial carbon debt is referred to as the carbon payback time. Figure S2 shows the variation of carbon payback time against carbon debt, assuming that post-conversion 15 Mg ha^{-1} yr^{-1} of woody biomass are pyrolysed to biochar with biochar C yields of 0% (bioenergy production only), 25% and 50%.

To ensure that biochar production provides net environmental benefits, criteria for feedstock provision and conversion technologies that are broadly similar to strict guidelines being developed for sustainable biofuels[68] will need to be implemented. Although no internationally recognised framework for assessing and certifying sustainable biofuel production has yet been adopted, guidelines under development do include considerations of carbon storage. In particular it has been recommended for the UK Renewable Transport Fuel Obligation (RTFO) that carbon-payback times must be less than 10 years, based on the IPCC guideline that a land-use change (LUC) should be considered to last for 20 years and the suggestion that any biofuel project should yield net greenhouse gas reductions for at least half of that time[69]. For equity with existing biofuel standards on carbon storage therefore, we have adopted this 10 year maximum carbon payback criterion.

In order to maintain a carbon payback time of less than ten years, the LUC carbon debt must be less than 80 Mg CO_2e ha^{-1} (22 Mg C ha^{-1}). This rules out clearance of forest or wooded savanna, or conversion of grassland to annual crops[70] (in which instances even loss of below-ground biomass will exceed the threshold) as a strategy to mitigate climate change. Carbon storage criteria are, thus, broadly convergent with habitat conservation considerations. We have, therefore, not included biomass whose provision requires the clearance of previously unmanaged lands or conversion of agricultural land to biomass-crop production (as this may induce land clearance by displacing agriculture[71]) in this analysis.

While this precludes 'LUC for biochar' that incurs significant carbon debt as a means of climate change mitigation, it does leave open the question of whether original vegetation removed during LUC that is occurring anyway (i.e., as part of business as usual (BAU) rather than for the explicit purpose of feedstock provision) should be considered as a sustainably available feedstock within our scenarios. To remain within our stated objective of making conservative assumptions where possible, we have not included original vegetation from BAU LUC for two principle reasons. Firstly, the large discrepancy between biomass quantities initially generated during the first year of land-clearance (~250 Mg ha^{-1} from rainforest) and subsequently from agricultural residues (<5 Mg ha^{-1} yr^{-1}) mean that pyrolysis facilities sized for ongoing operation will not have the capacity to process the large quantity of initial material, whereas no mobile or temporary pyrolysis facility capable of processing this quantity of material in often remote areas with poor infrastructure without producing unacceptable gaseous or soot emissions (see 2.4.1.2) currently exists. The second reason we have not included BAU LUC residues in our analysis is that if such residues were to be admissible under any protocol that delivered financial incentives for biochar production, then LUC that was promoted by the availability of such incentives would, in practice, be impractical to distinguish from BAU LUC. Under such circumstance, biochar might become a driver of increased land-clearance beyond BAU, whereby it would exacerbate rather than mitigate climate change.

Although clearance of previously unmanaged land leads to unacceptably high carbon payback times, planting perennial biomass crops on degraded, abandoned cropland incurs little carbon debt and can lead to "immediate and sustained GHG benefits"[70]. A maximum of 0.6 Pg C yr^{-1} biomass may be produced on abandoned cropland that has not subsequently become urban, pasture or forest[72]. We assume that half of this might be provided by herbaceous crops with a C content of 47%, and half from woody crops with 49%C. In the Alpha scenario, we assume that 50% of this maximum potential might

be implemented, rising to 75% and 100% in the Beta and MSTP scenarios. The total amounts available from herbaceous biomass crops thus are 319, 479, and 638 Tg DM yr^{-1} for the Alpha, Beta and MSTP scenarios, respectively. For woody biomass crops, the comparable amounts are 306, 459, and 612 Tg DM yr^{-1}.

We note that, despite concerns over the sustainability of biomass plantations, establishment of biomass crops on degraded land is not fundamentally contrary to the principles of sustainability as there exist land-management options such as long-rotation coppice and use of native flora such as prairie-grass mixes that can enhance biodiversity and simultaneously provide biomass (see for example ref. 73). Furthermore, careful choice of plants can reverse land degradation and rebuild soil fertility.

2.1.7 Harvested wood

Annually, 1.9 Pg of timber is extracted[74], of which 1.04 Pg is firewood (derived from 2007 FAO ForeStat data[66] using mean hard and soft wood densities of 0.42 and 0.57 Mg m^{-3} from ref. 74). Using firewood for biochar production would result in a reduction in the useful energy available from the wood and thus lead to greater fuel poverty and/or increased extraction, either of which are unacceptable consequences. Aside from losses due to fire, fungi and termites, most of the remaining industrial wood eventually ends up in various waste streams. However, the wide range of chemicals used in production, preservation and colouration of both wood and paper products means that only a small fraction of this carbon may currently be suitable for production of biochar as a soil amendment. Hazardous waste wood categories include wood panels, chip and fibre boards, and sawnwood treated with preservatives, paints or varnish. Paper products may contain heavy metals in inks and Cl from bleaching. In addition to Cl from bleaching, pulping waste may also contain high concentrations of Na from the Kraft process which would contribute to sodicity or salinization if added to soils. We recognize, however, that biochar from some paper mill residues may have value in some soils, but adopt a conservative approach and do not include them in our calculation.

In the Alpha scenario therefore, we have limited available feedstock from timber to 50.5 Tg of wood milling residues[66]. For the Beta and MSTP scenarios, however, we assume that legislation will be enacted to replace Cu, Cr and As based wood preservatives with treatments such as acetylation, potassium silicate, tung oil or linseed oil that will not render the wood unsuitable for adding to soil, thus making 40-80% of the 198 Tg yr^{-1} sawnwood available for pyrolysis at end of life in the Beta and MSTP scenarios respectively. Total amounts available are 51, 130, and 209 Tg DM yr^{-1} for the Alpha, Beta and MSTP scenarios, respectively.

2.1.8 Forestry residues

The total amount of forest residue is taken to be the difference between reported forest fellings and forest removals, giving a figure of 0.65 Pg biomass yr^{-1} (ref. 74). As with agricultural residues, forest residues provide a number of important services including providing nutrients, regulating water flows, reducing soil erosion, creating habitat and provide food for fungi and detritivores. Extraction of felling losses can have adverse impacts on these functions. The fraction of felling losses that may be removed without unacceptably impeding one or more of these services is site dependent, and may vary from 0% to 75%[75]. Removal of nutrients may be minimised by seasoning residues prior to extraction until foliage has been shed[76], and by topping crown-wood prior to extraction leaving the finer brash on site[77].

In the EU, it has been assessed that, on average, 59% of potentially recoverable forest residues (which, in turn, are 75% of total residues) may be recovered if environmental constraints are taken into consideration[75]. In the absence of similarly detailed assessments on a global scale, we have assumed that the European figure is applicable globally, allowing 44% of total felling residues to be recovered sustainably. Thus, we have assumed that, in each of our scenarios, 286 Tg DM yr^{-1} of forestry residues will be available.

2.1.9 Agroforestry

The largest single contributory cause of human appropriation of net primary productivity (NPP), accounting for a reduction in global NPP of 6 Pg C yr^{-1}, is the loss of production caused by replacing natural ecosystems with less productive ones[78]. In particular, cropland is dominated by the cultivation of annual crops that waste resources through having insufficient leaf and root development for much of their life cycle to capture available light, nutrients and water. There may, therefore, be considerable scope to increase global net primary production through choice of crops, and in particular through greater use of perennial crops.

We consider in our scenarios just one of the many ways in which biomass feedstock availability may be increased by planting alternative crops. We consider here the potential impact of replacing grass pasture in the tropics with a silvopastoralism system in which the main source of fodder for the livestock is foliage from fast-growing leguminous shrubs or trees that simultaneously produce wood for biochar. Since the wood producing species are also the source of food crop, this allows for dense planting as in the 'fodder bank' style of agroforestry, in contrast to alley cropping agroforestry systems where wood production occupies only a small fraction of the available area. A number of different management options are suited to such production of multi-purpose trees, including allowing the stock to browse directly (on a rotational basis), or coppicing with cut-and-carry (and drying for storage) of the foliage. In addition to providing biochar feedstock, replacing livestock pasture managed by intentional burning with silvopastoral systems will also have the added benefit of reducing the CO_2, N_2O and CH_4 emissions associated with fire[2], although this effect has not been included in our model.

A great many species of trees and shrubs are suited to co-production of fodder and biomass, the preferred choice locally depending on factors such as adaptation to soil and climate, invasiveness, pest/disease resistance, nutritive value, palatability, ease of management, and preference to native over exotic species where possible. A few examples of tree legumes that can provide high yields of high quality forage and human food together with timber[79], include *Leucaena* spp.[80], *Moringa oleifera*[81], *Acacia* spp.[82] and *Caragana* spp.[83], the relative yields of leafy and woody material being controllable by varying the frequency of cutting. We have assumed no net increase in N_2O emissions resulting from nitrification or denitrification of biologically-fixed N under legume crops, for two reasons. Firstly, there may be no net increase in N_2O emissions once avoided emissions due to fire and N additions in displaced production systems have been accounted for. Secondly, biologically-fixed N is efficiently used by legume crops to produce protein making it initially less available for nitrification and subsequent denitrification[84]. Where necessary, any surplus reactive-N remaining in downstream products (e.g. manure) may then be reduced to N_2 by pyrolysis for biochar production, as described in Section 2.4.1.2.

Under ideal conditions, *Leucaena* hybrids can give annual yields in excess of 50 Mg ha^{-1} total DM biomass of which half is forage[85]. However, we assume much more modest average yields of 15 Mg DM ha^{-1} yr^{-1} (50% food and fodder and 50% woody biomass) to allow for less than ideal conditions, and for selection of species based on the full range of desirable attributes, not just maximum yield.

To produce a spatially explicit estimate of the amount of land currently under pasture that would be suitable for silvopastoralism, we selected from a global 5' of arc resolution map of existing pasture[86] only those regions that are in agroecological zones with no serious climate, soil or terrain constraints to rainfed crop production (i.e., zones having constraints no greater than 'moderate' on plate 28 of the joint IIASA / FAO global agroecological zone map[87] 2000). This yields a total of 170 Mha of land at latitudes less than 25° (Fig. S12). Thus, we estimate that the maximum potential production of woody biomass feedstock from such agroforestry systems in the tropics to be 1280 Tg DM yr^{-1} of which 10, 55 and 100% are assumed to be implemented in our three scenarios respectively.

2.1.10 Green Waste

Urban organic waste generated has been estimated[67] to amount to 1-3 EJ yr^{-1}. Accordingly, we use values of 1, 2, and 3 EJ yr^{-1} in the three scenarios respectively. Assuming an energy content of 17.5 GJ Mg^{-1} DM (see section 5), this corresponds to 57-172 Tg DM yr^{-1}. We base our estimate of the useable fraction on a 2007 EPA study of US urban waste in which 20% of urban organic waste was yard trimmings and 20% was food waste[88]. In the Alpha scenario, we assume that 75% of yard trimmings are used as feedstock, whereas in the Beta and MSTP scenarios we assume that all yard trimmings and food waste will be used. Other categories of organic waste – paper, wood and miscellaneous organics – are excluded due to the high probability of contamination. This gives us an estimated 8.6, 46 and 69 Tg DM yr^{-1} in the three scenarios respectively.

2.1.11 Other excluded feedstocks

The feedstocks considered in these scenarios do not provide an exhaustive inventory of all potential resources. Together though, they do constitute the majority of the biomass that may be utilised without increasing human appropriation of NPP.

One potential resource that has been excluded from this study is substitution of slash-and-burn agriculture with slash-and-char both for reasons given in section 2.4.1.2 and because it may be impractical to differentiate legitimate use of this practice to displace shifting agriculture from its use as a means of land clearance.

Another potential resource that has been excluded is forest thinnings. We recognize that managed forests in developed countries are thinned during the normal growth cycle and may also be thinned for

fire control, and that these thinnings might be considered as suitable feedstock for biochar production. Implementation of thinning programs, however, is often controversial, even in the US, and has been subject to abuse. Extraction of thinnings is also often costly, difficult and incurs risks of soil disturbance and nutrient depletion. Estimation of the amount of feedstock potentially obtained by thinning is also difficult due to a lack of suitable data from most countries. For all these reasons, and to avoid even the appearance of promoting conversion of forest land to other purposes, then, we exclude forest thinnings from this assessment.

Another potential feedstock resource that we have not included is the use of invasive species, harvesting of which may provide a means of control. The Global Invasive Species Database (GISD) lists 377 plant species[89], many of which might be practically harvested as part of a management strategy. However, although data on the ranges of these species are known, there are no comprehensive global surveys of the quantity of biomass they represent. An assessment of the potential biomass that might be generated from invasive species will require land-based, local, regional surveys and it is not possible to perform on a global study of this nature. For this reason, and also because successful control of these species through harvesting would deplete this resource, we have not included this class of feedstock in our assessment.

2.2 Avoided biomass decomposition

The primary influence of biochar production on greenhouse gas fluxes is due to the avoidance of emissions that would have occurred had the biomass been left to decompose. Here we consider the avoided emissions of carbon dioxide, methane and nitrous oxide that would have occurred under present biomass management methods.

2.2.1 Carbon dioxide

The rate at which biomass will decay, if it is not pyrolysed, varies considerably depending both on the feedstock and on how it is used, stored or disposed. No avoided emissions can be considered to have occurred until after the time that biomass would have decomposed. In the case of ephemeral forms of biomass such as crop residues, this has little impact on the rate at which avoided emissions may accrue. However, pyrolysis of potentially durable forms of biomass, such as live standing timber or timber products that have not yet reached end-of-life disposal, will lead to an up-front increase in GHG) emissions that will not be recouped for some time. The criterion that the carbon-debt payback period for biochar production must be less than ten years precludes the use of such potentially durable feedstocks. We have therefore restricted ourselves to feedstocks with short life expectancy and have thus made the simplifying assumption that biomass “in the wild” would have decayed exponentially with a half-life of 1 yr for non-timber feedstocks, and 3 yr for timber, if not pyrolysed.

To calculate the rate at which biomass carbon would have contributed to atmospheric CO_2, account must also be taken of the contribution that biomass would have made to soil organic carbon (SOC) stocks had it not been pyrolysed. Large-scale diversion of biomass from soil may lead to a depletion of non-BC SOC stocks.

The decomposition rate of organic matter in soil varies depending on biomass type, soil, and climate[90]. In a study of the potential for carbon sequestration in European soils through straw incorporation[91], the RothC model was used to predict the rate of SOC accumulation in a silty clay loam soil in north-west Europe over 100 years resulting from annual additions of cereal straw at a rate of 4.25 Mg DM ha^{-1}. The mean rate of carbon accumulation over the first twenty years was 0.46 Mg C ha^{-1} yr^{-1}, falling to 0.12 Mg C ha^{-1} yr^{-1} in the second fifty years. In the absence of similar long-term studies for the full range of biomass types, soils and climatic regions considered in this study, we have used this study as a basis for calculations. It is to be expected that this will lead to an overestimation of the amount of SOC that would accumulate in soils, since the rate of SOC mineralization tends to increase with decreasing latitude[90], and also because substituting biochar for more labile forms of SOC will reduce the susceptibility of soil carbon to climate change induced losses[92]. This will, therefore, lead to a correspondingly conservative estimate of the avoided emissions achievable through biochar production. Thus, although a more accurate assessment of loss of SOC from reduced biomass inputs to soil may be possible, this methodology suits our requirement that we should endeavour to make a conservative estimate of the potential for biochar that can provide a low-risk basis for decision making.

Loss of SOC as a consequence of reduced biomass inputs to soil was thus modelled by fitting a two-pool exponential decay function of the form

$$\Delta SOC = A \exp(-k_1 t) + B \exp(-k_2 t) \quad (1)$$

to the data in ref 91 using the non-linear minimization function, nlm(), in the R statistical programming

language[93], yielding the coefficients

A=7.800, k_1=-0.02669, B=130.0, k_2=-0.0005246

with a correlation coefficient of r^2 = 0.99999

Loss of detritus inputs to soils by diverting biomass to biochar will be offset somewhat by increased crop yields as a result of biochar additions to soils. We have assumed in this study that, where biochar-induced enhanced yields occur (calculated as described in 2.6.3 and 2.6.4 below), increases in above-ground biomass production will become available as a feedstock for biochar production, and will therefore not contribute to SOC stocks. Previous studies have not shown biochar amendment to increase below-ground net primary production as much as above-ground biomass. In a pot trial of cowpea in biochar amended feralsols[94], although above-ground yields increased with biochar additions, no increase in below-ground production was observed. In another study of common beans (*Phaseolus vulgaris*) in an oxisol[95], increases in both above- and below-ground production were observed, but the absolute increase in below-ground production was only approximately 20% as much as for above-ground biomass (at biochar additions of 30 g kg^{-1}). Taking the average of these two studies, we have assumed that the increase in below-ground biomass will be, on average, only 10% of above-ground enhanced yield, and that this enhanced below-ground production will add to SOC stocks the increase in SOC being calculated according to equation (1) above.

2.2.2 Methane

When biomass decomposes under anaerobic conditions, methane is evolved. Diverting biomass from these methanogenic processes into biochar can therefore lead to a reduction in CH_4 emissions[96,97]. We consider here avoided emissions from paddy rice cultivation, manure management, and landfill waste. All other biomass feedstocks in our model produce negligible CH_4 emissions under current conditions.

2.2.2.1 Rice

In order to arrive at a global emission factor for rice residues that are pyrolysed, we begin by deriving from published studies a mean emission factor of methane per mass of rice straw that is incorporated into paddy fields (see Table S5). There is a factor of over 40 between the largest and smallest values that may be accounted for by regional variations in climate and cultivation practices.

Not all straw that is pyrolysed would otherwise have been added to fields. Approximately 26% (0.23 Pg) of the 0.89 Pg yr^{-1} rice straw generated is currently not recovered[62]. Of the straw that is recovered, we assume that none of it is subsequently returned to paddy fields. Thus, we estimate that the amount of rice straw currently being incorporated into paddy fields is 0.23 Pg DM yr^{-1}. Multiplying this by the mean CH_4 emission factor of 0.076 kg CH_4-C (kg straw)$^{-1}$ implies a net global production of methane from straw incorporated into paddy fields of 17.5 Tg CH_4-C yr^{-1} (402 Tg CO_2e yr^{-1}), or approximately 40% of the total global methane emissions from paddy rice production[98] of 41 +/- 13 Tg CH_4-C yr^{-1} (940 +/- 300 Tg CO_2e yr^{-1}).

In order to derive an emission factor for all rice straw that is pyrolysed, we must multiply the mean emission factor of 0.076 by the fraction of pyrolysed straw that would have been left in field. To define this fraction we assume that once maximum production has been reached, all currently unharvested straw (0.23 Pg) will be used, plus a fraction of the remainder. Therefore (at time of maximum production when the annual quantity of rice straw pyrolysed is R_{max}), the fraction of pyrolysed straw that would otherwise have been left in the field is equal to 0.23 Pg / R_{max} and the emission factor for pyrolysed straw will be 0.076 * 0.23 Pg / R_{max}. We assume that this emission factor remains constant over time, i.e., that the fraction of pyrolysed straw derived from residues that are currently unharvested remains constant.

2.2.2.2 Manure

Estimates of global CH_4 emissions from livestock manure are 10.4 (ref. 99) and 25.5 Tg CH_4-C yr^{-1} (ref.65), most of this deriving from anaerobic storage systems such as lagoons or manure heaps. We base our calculations on the more conservative of these, 10.4 Tg CH_4-C yr^{-1}, as total CH_4 emissions from ruminants are 68 +/- 14 Tg CH_4-C yr^{-1} (ref. 100), the bulk of which arises from enteric fermentation. Of this total CH_4 flux, 4.5, 4.0, and 1.0 Tg CH_4-C yr^{-1} originate from cattle, pig and poultry livestock manures, respectively[99]. We derive a global mean emission factor of methane from livestock manure by dividing total CH_4 emissions from manure by global manure production. Total manure production is taken to be 470 Tg C yr^{-1} for cattle[65], 34 Tg C yr^{-1} for pigs[65], and 46 Tg C yr^{-1} for poultry manure (section 2.1.5). Assuming carbon contents of 29.9%, 37.6% and 34.7% for cattle, pig and poultry manures respectively (see Table S3), this yields mean

emission factors of 0.0029, 0.044, and 0.0025 kg CH_4-C kg^{-1} DM (0.0874, 1.35 and 0.0759 kg CO_2e kg^{-1} DM) for cattle, pig and poultry manures, respectively. Given that in the Alpha and Beta scenarios, manure for pyrolysis will be sourced principally from housed livestock, not from manure deposited directly on land (from which CH_4 emissions are negligible[65]), we assume that the emission factor for cattle manure will be the global mean divided by the fraction of manure that is derived from housed livestock. Of the 100.6 Tg (wet) cattle manure produced annually, 65.5 Tg (65%) is from housed cattle[65], giving a CH_4 emission factor for housed cattle manure of 0.0045 kg CH_4-C kg^{-1} DM (0.138 kg CO_2e kg^{-1} DM). In the MSTP scenario, we assume that avoided methane emissions accrue only from manure that is currently produced from housed livestock. Any additional manure procurement is not considered to contribute to avoided methane emissions.

2.2.2.3 Landfill

Globally, 23±11 Tg CH_4 yr^{-1} are emitted from landfill refuse[100]. Diversion of waste from landfill to pyrolysis can avoid some of these emissions. In order to calculate the avoided methane by diverting green waste from landfill to pyrolysis, we use a CH_4 emission factor for landfill green waste of 0.038 ± 0.015 kg CH_4-C kg^{-1} DM, derived from the mean of seven experimental incubation and field studies[101,102]. As most of the methane is evolved during the first year[101], we assign all the avoided methane emissions to the same year in which the biomass was pyrolysed. We assume that 50% of green waste pyrolysed has been diverted from landfill, the rest being diverted from disposal methods that generate negligible methane. Although anaerobic or partially anaerobic composting of green waste also leads to methane emissions, the increasing use of fully aerobic in-vessel composting entails negligible methane production[103].

2.2.3 Nitrous oxide

Global N_2O emissions from manure management and other biomass are 205 and 187 Tg CO_2e yr^{-1}, respectively[104]. While some of these emissions may be avoided by diverting biomass into pyrolysis, this must be balanced against emissions due to nitrogen compounds formed during pyrolysis.

During pyrolysis, feedstock nitrogen is partitioned between the biochar and gaseous products. Of the nitrogen that remains in the biochar, its biological availability – and thus the rate at which it may contribute to soil N_2O emissions – depends upon the compounds in which it occurs. Despite the expectation that nitrogen forms relatively stable heterocyclic aromatic compounds[105] in a study of ^{15}N labelled chicken manure biochar, it was found that the same amount of the nitrogen in the biochar was plant-available as in uncharred poultry litter[106]. Thus, if this result is generally true of biochars produced under a range of conditions, N retained in the biochar will still contribute to soil reactive nitrogen concentrations and may contribute to N_2O emissions with a similar emission factor to N in the original biomass. For the purpose of this study we take the conservative assumption that this is generally the case, and do not attribute any avoided N_2O emissions to the N fraction retained in the biochar.

The remainder of biomass N is evolved in the volatile fraction where it may be present as a range of compounds including N_2, NH_3, HCN, NCO, N_2O, NO and NO_2. Of these, N_2 is the most abundant, accounting for around 80% of N in the gaseous fraction[107-109]. We therefore calculate the amount of biomass N converted to N_2 as 0.8 times the difference between mass of N in feedstock and in biochar (using N concentrations as given in section 5).

Of the other gaseous N compounds, NH_3 is the most abundant. Given increasingly stringent regulation, pyrolysis of high-N feedstocks can only be considered a viable option if measures are taken to minimise emission of environmentally deleterious N compounds including NO_x and NO_x precursors such as NH_3 and HCN[107]. NH_3 may be converted to ammonium and adsorbed onto the biochar, where it will act as a fertiliser. While this usefully recaptures some of the biomass N, it means that nitrogen evolved as NH_3 will be returned to the soil in a biologically available form where a portion of it will be converted to N_2O with an emission factor similar to that for N in the original feedstock. Thus, N evolved as NH_3 (or any gas other than N_2) can not justifiably be said to contribute to a net reduction in atmospheric N_2O emissions. N_2O and NO_x yields are generally small, surface reactions on biochar aiding in their reduction to N_2[110], with further reductions to acceptable levels being achievable, if necessary, through flue-gas scrubbing[111]. Production of HCN is negligible below around 500°C[107], and may be suppressed by the presence of CO_2[112].

The emission factors we use for calculating avoided N_2O emissions are the IPCC-recommended values[113,114]. For direct emissions, these are 1% (kg N_2O–N kg^{-1} N) for N applied to cropland, and 2% for cattle and pig (but not poultry) manure deposited at pasture, range or paddock. It has been estimated that, globally, 50% of manure N is applied to cropland[115]; thus the mean emission factor we have used for direct emissions from cattle and pig manure is 1.5%. An indirect emission factor (to account for N_2O emissions from leached and volatized N) of 0.325%, derived from IPCC default

values[116], is also added to the direct emissions to calculate total avoided N_2O.

It has been suggested that the IPCC emission factors may seriously underestimate N_2O emissions from N applied to soils[113], with the actual rate being 3-5%, due to indirect processes that occur in groundwaters and surface waters after reactive N has left the soil column. Although we have used IPCC recommended values for the main results, we have therefore also included a sensitivity analysis for N_2O emission factors in the range 1-5%.

Avoided N_2O emissions are thus calculated as the fraction of biomass N converted to N_2 multiplied by the N_2O emission factor, according to the equation:

$$0.8 * (M_f * N_f – M_{bc} * N_{bc}) * E_{N2O}, \quad (2)$$

where, M_f = mass of feedstock, N_f = N content of feedstock, M_{bc} = mass of biochar, N_{bc} = N content of biochar, E_{N2O} = emission factor of biomass-N to N_2O.

2.3 Transport

The worst-case scenario for transport costs will occur when pyrolysis plant size is large, requiring biomass to be transported to a central facility from a large area. To provide a conservative estimate, we base our transport costs on this worst case, whilst recognising that, in practice, a more distributed network of smaller plants will be more amenable to both lower transport costs and possibly also greater utilisation of process heat close to areas of heat demand. We do not, therefore, propose that a network of large, highly centralised plants as assumed for this calculation will be the most likely or economic form of infrastructure, but make this assumption only to remain true to our aim of making conservative assumptions where possible. To calculate the mean transportation distance, we calculate how many 1-GW (feedstock higher heating value (HHV)) capacity pyrolysis plants would be required to convert the biomass. We estimate that these are distributed evenly across an area of 6.6 Gha, calculated by summing the area of all grid cells on a 5' of arc raster map of the globe that contain some cropland (Fig. S7). We then calculate the mean separation (S) between plants as the square root of the area served by each plant. The mean transport distance (D) for both biomass and biochar is then calculated as D=S/2, a value that accounts for indirect routing of roads by assuming that the distance travelled between two points (x,y) and (0,0) is given by x+y as it would be if roads were set out on an orthogonal grid.

Transport energy per kilometre is then calculated according to the UK average road-freight consumption of 2.2 MJ Mg^{-1} km^{-1} (ref. 117). Although this is a somewhat crude estimate, given that according to these calculations transport energy contributes less than 2% of the overall GHG budget, a more sophisticated analysis is unlikely to affect the overall results greatly. This is supported by analysis of the greenhouse gas balance from charcoal production, in which transport emissions were found to be a negligible component[3].

2.4 Pyrolysis

2.4.1 Products of pyrolysis

2.4.1.1 Carbon yield

We define C yield as the mass of C in the solid biochar residue divided by the mass of C in the initial dry biomass feedstock. We have assumed a C yield of 49% (the mean of values obtained by slow pyrolysis at atmospheric pressure in Ref. 118) to generate Figures 2 and 3 in the main text. For the sensitivity and Monte-Carlo analyses, we have investigated C yields up to 62% as demonstrated by a pressurized, flash-pyrolysis process[118]. C yields close to 100% have been demonstrated using a high pressure, catalysed hydrothermal pyrolysis process, but we have not included this process in our analysis because the low stability of the resultant char (mean residence time of 4-29 yr) means that it does not provide a stable form of carbon sequestration[119].

2.4.1.2 Gas emissions

Gaseous products of pyrolysis include CO_2, CO, CH_4, H_2, N_2O, NOx and smaller quantities of volatile organic compounds (VOCs). In order to calculate the GHG budget of pyrolysis, we consider emissions of CO_2, CH_4, and N_2O. CO has low global warming potential and a short lifetime in the atmosphere before oxidising to CO_2. For simplicity, therefore, CO emissions are simply treated as CO_2. Yields of VOCs contribute a negligible amount to the overall GHG budget. We thus assume that all feedstock-C is converted to either biochar-C, CH_4, or CO_2, the yield of CO_2–C being calculated as (1 – biochar-C – CH_4-C). N_2O emissions will also be negligible for low N feedstocks, and may be controlled or eliminated in high-N feedstocks (see 2.2.3).

Depending on the conversion technology utilised, the pyrolysis gas may or may not be fully combusted before venting to atmosphere. Failure to combust the pyrolysis gases (as may occur,

for example, in traditional charcoal kilns or smouldering slash piles) will lead to the squandering of a potential energy source combined with increased greenhouse-gas emissions.

In a study of emissions from 8 different charcoal kilns in Brazil and Kenya[120], methane emission factors varied from 24-47 g CH_4-C kg^{-1} charcoal, or approximately 6-11 g CH_4-C kg^{-1} dry feedstock, the lower figures being from earth-mound kilns and the higher value from a brick and metal kiln. Average emissions from all charcoal making are approximately 8 g CH_4-C kg^{-1} dry feedstock[121]. This model calculates that yields of just 1 g CH_4-C kg^{-1} dry feedstock are enough to cause the carbon-payback time to exceed 10 years. Thus pyrolysis in traditional kilns will not satisfy our carbon-payback criteria. The suggested practice of slash-and-char agriculture[122] through such low-tech conversion methods would not, therefore, be a viable greenhouse-gas reduction strategy unless it replaces slash-and-burn with an increase in CH_4 emissions of no more than 1 g CH_4-C kg^{-1} dry feedstock compared to those from burning. CH_4 emissions from biomass burning are typically in the order of 5 +/- 2 g CH_4-C kg^{-1} dry feedstock[121]. Air-quality considerations also require that pyrolysis plants be implemented in such a way that emissions of soot, oxides of sulphur or nitrogen, carbon monoxide and other pollutants are negligible. Therefore, we will not include biochar production through low-tech conversion.

2.4.2 Energy production and fossil fuel offsets

If we assume that the gaseous and volatile products of pyrolysis (referred to hereafter simply as 'pyrolysis gas' for brevity) are completely combusted at close to stoichiometric aeration, then the net energy extracted from the process (i.e., the difference between the feedstock enthalpy and the combined enthalpies of the biochar produced and the exhaust gases) is dictated by the yield of biochar and the temperature at which the exhaust is vented. If we also assume that sufficient energy is recovered from the exhaust gas stream to lower its temperature to just above the water condensation temperature, then the maximum energy (per unit mass of feedstock) available from the process without condensing the exhaust stream is given by

$$E_{max} = m_{dm}.LHV_{dm} – m_c.HHV_c – m_w.(\Delta H_{vap} + (T_{vap} - T_a).c_w), \quad (3)$$

where,
m_{dm} =mass of dry matter feedstock
LHV_{dm} = lower heating value (LHV) of feedstock dry matter.
m_c = mass of biochar
HHV_c = biochar higher heating value (the higher heating value being used here since none of the enthalpy remaining in the biochar is available for energy)
m_w = mass of water
ΔH_{vap} = specific heat of vaporization of water, taken as 2.26 kJ g^{-1}
T_{vap} = 100°C
T_a = ambient temp, taken as 20°C
c_w = specific heat capacity of water, taken as 4.18 J g^{-1} K^{-1}

In practice, however, inefficiencies arise from heat losses and parasitic loads. The energy recovered from pyrolysis will therefore be given by

$$E=\eta_p E_{max}, \quad (4)$$

where η_p, the pyrolysis energy efficiency, is defined as the fraction of theoretically recoverable energy that is recovered in practice. We base our estimate of η_p on the operating efficiency of a currently commercially available slow pyrolysis plant from Best Energies which, it is reported, typically obtains an energy yield of 38% of the feedstock energy when the conditions are optimised for maximum biochar yield[96]. If we assume that this energy yield of 38% is achieved under favourable conditions using a low water-content feedstock such as wheat straw, then (using values of HHV_c, LHV_{dm}, C% and m_w, given for wheat straw in sections 5 and 6 and a pyrolysis C-yield of 48%), η_p is given by

$$\eta_p = E/E_{max} = [0.38\ LHV_{dm}] / [LHV_{dm} – m_c.HHV_c – m_w.(\Delta H_{vap} + (T_{vap} - T_a).c_w)], \quad (5)$$

in which m_c is determined by the pyrolysis C-yield and the biochar C% and m_w is determined by the biomass water content. This yields an estimated value of η_p of 75%. While we recognise that higher efficiencies are achievable, and might become the norm over the course of a century, this provides a conservative estimate of the energy production that can be obtained.

It is evident from equation (3) that increasing water content of biomass carries a penalty in terms of reduced energy efficiency. For example, high-ash feedstocks such as manure, will require more

energy to pyrolyse than may be recovered if the water content is greater than approximately 58% (although the net avoided greenhouse emissions may still be positive when sequestered carbon and other feedbacks are considered).

The energy recovered from pyrolysis may be used to offset fossil fuel consumption. The fossil fuel emissions offset by the co-production of energy with biochar then depend on two factors – what fuel is being offset, and ratio of the efficiencies with which the fossil fuel and pyrolysis gas are utilised[123] – according to the equation

$$A_{ffo} = E\, C_{ff}\, \eta_b / \eta_f \qquad (6)$$

where
A_{ffo} = avoided CO_2-C emissions,
C_{ff} = fossil fuel C penalty (i.e., CO_2-C emissions per unit energy of fossil-fuel heating value),
η_b = thermal efficiency by which pyrolysis gas is converted to useful energy,
η_f = thermal efficiency by which fossil fuel is converted to useful energy

We use IPCC default values for C_f of 0.026 Mg C GJ^{-1} for coal, 0.019 Mg C GJ^{-1} for oil and 0.015 Mg C GJ^{-1} for natural gas[123].

To estimate the C-intensity of the energy mix that may be offset by biomass energy (with or without simultaneous biochar production) we begin by considering the current mix of world energy generation using IEA figures[124]. Current total primary energy production is 27% coal, 33% oil, 21% gas and 19% non-fossil, yielding a mean C-intensity of 0.0165 Mg C GJ^{-1}. In our analysis, however, we consider how climate mitigation from biochar might be maximised. The sensitivity results given in the main paper show that mitigation is maximised by optimising for maximum yield of recalcitrant char. We therefore base our analysis on the assumption that slow rather than fast pyrolysis with its higher char yield will be the preferred biochar production process. This will tend to favour the production of syngas for static applications over liquid fuels for transport. Therefore, of more relevance than the overall world energy mix is the energy mix for static applications. Current world electricity production is 41% coal, 6% oil, 21% gas and 31% non-fossil, yielding a combined C-intensity of 0.015 Mg C GJ^{-1}. If we also consider heat demand, the current average C-intensity becomes 0.0154 Mg C GJ^{-1}. We adjust our baseline from these current statistics somewhat, though, to take account of two considerations. First, that rising prices and the prospect of depletion are driving a trend away from use of oil in electricity production. Second, it is likely that, to some extent, policy mechanisms will be used to promote the offsetting of higher C-intensity fuels more than lower C-intensity ones. We therefore use a baseline C-intensity for offsets of 0.0175 Mg C GJ^{-1} calculated from a mix of 50% coal, 30% gas and 20% of pyrolysis plants that will not generate any fossil fuel offsets, either because no energy is recovered, because the energy source being offset is renewable or nuclear, or because the energy recovered represents an increase in overall energy consumption (the 'rebound effect').

For (pyrolysis gas → electricity) offsetting (coal → electricity), the ratio η_b/η_f is assumed to be 32%:40% = 0.8. For applications (including CHP) where heat is recovered, the expected efficiency will be in the order of 85% for both pyrolysis gas and fossil fuels, yielding a ratio of 1.

In order to calculate the maximum energy available when optimising for energy rather than char production, we use a η_b/η_f ratio of 0.92, C-yield = zero (i.e., negligible char is produced when optimising for energy production), and all other assumptions identical to the biochar scenarios.

2.5 Biochar properties

2.5.1 Decay kinetics

Biochar is not a homogeneous substance, but comprises both condensed or residual aliphatic compounds and black carbon (BC) which is itself a continuum ranging from slightly charred biomass that is readily decomposed to highly recalcitrant condensed aromatics[125]. That biochar consists of a mixture of compounds with different decay kinetics in soil is evident from incubation studies in which decay rates are initially rapid but slow down over time[126,127], and from the long mean residence times of BC in some soils[92]. For simplicity, we have accordingly modelled biochar as a two-phase material with a labile and a recalcitrant component, each following an exponential decay curve as expressed by the equation

$$M(t) = M_0\, [L \exp(-\ln(2)/t_{½L}\, t) + R \exp(\ln(2)/t_{½R}\, t)],$$

where

M(t) = mass of biochar carbon at time t

M_0 = initial mass of biochar carbon

L = labile fraction of biochar,

R = recalcitrant fraction of biochar,

$t_{½L}$ = labile half-life

$t_{½R}$ = recalcitrant half-life

The labile fraction in this model subsumes both highly labile material with $t_{½}$ in the order of months-years, and moderately labile material with $t_{½}$ in the decadal range. While this approximation may be inaccurate for describing biochar decay kinetics over short timescales, for our purposes in modeling the net response over the course of a century, the distinction between highly labile and somewhat labile becomes substantially irrelevant, as both largely decay over the course of 100 years. We have used a ranges of $t_{½L}$=1-25 years and L=5-30% for the sensitivity and Monte-Carlo analyses, and values of $t_{½L}$=20 years and L=15% to generate Figures 2 and 3 in the main paper. While data on decay kinetics over the decadal timescale are scarce, these values are in accord with incubation studies that show 4.5% C loss from perennial ryegrass (*Lolium perenne*) biochar after 3 years[126], and 7-10% C loss in high-ash biochars and 3-5% loss for wood biochar after 2 years[127].

The half-life of complete (both labile and recalcitrant) BC in soil represents a lower bound on the half-life of the recalcitrant fraction alone. Achieving reliable estimates of biochar half-life in soils is subject to considerable methodological difficulties - estimates derived from studies of historic BC in soils are confounded by the difficulty of distinguishing between mineralization and other loss processes such as leaching and erosion[128]. Incubation studies on the other hand are poor indicators of behaviour in soils *in situ*[129,130]. Published estimated values for the half-life of BC in soils are highly variable (see Table S6). In part, this variability may be attributed to differences in environment such as mean annual temperature[131]. Importantly though, this variation may reflect differences in the degree of carbonisation of the initial BC stock as BC includes materials on a spectrum from slightly to highly charred with a corresponding variation in recalcitrance[125,132]. We note that all of the studies in Table S6 that suggest BC half-lives of less than 400 years (refs 129,133,134) consider BC formed under uncontrolled conditions during wildfires, which is therefore likely to have contained considerable quantities of lightly charred (and thus labile) material. This is supported by the fact that in ref 134, initial mass loss over the first 30 years was rapid after which BC concentration stabilised, implying both a high labile content of the biochar and a long half-life of its recalcitrant fraction. Thus, the recalcitrant half-life of 300 years that we assume for Figures 2 and 3 in the main paper seems highly conservative and the range of 50-1000 years that we use for sensitivity and Monte-Carlo analyses covers only the lower end of plausible values.

While BC losses from the top soil horizons due to horizontal or vertical transport will not affect the carbon-sequestration benefits of biochar, they will affect its impact on soil fertility.

2.5.2 Carbon fraction

The C fraction of biochar is highly dependent on the ash (and, to a lesser extent, H and O) contents of the feedstock[135]. Assumed values for C and ash content of biochars are given in Table S4. We have based application rates to land (see below) on mass of carbon rather than mass of biochar. Thus high ash biochars (for example from manure) will have a higher application rate to land to achieve the equivalent additions of C.

2.6 Biochar soil application

2.6.1 Application rates

The available evidence suggests that application rates of at least 50 Mg C ha^{-1} can be made to the top 15 cm of soils with positive or neutral effects on biomass yield (see Section 2.6.4 below). In some instances, initial N-immobilization may occur, caused by metabolism of highly labile C present in some biochars, resulting in an associated yield decrease. Where such decreases in yield have been observed, a year or two of weathering may be needed before this is overcome [e.g., see yield data of Deenik et al.[136]]. We assume here, then, that single application rates of 50 Mg C ha^{-1} can be safely made on all soil types. Given that 1.53 Gha of cultivated land are available, the minimum global soil storage capacity for biochar can thus be calculated at 76.5 Pg. This capacity is sufficient to account for at least a century of biochar production at sustainable production rates. Further soil capacity may be achieved by application to extensive pasture (range) land. However, much range land may be remote from sources of biochar feedstock and access for machinery is often also constrained by terrain. While intensively managed grasslands could readily have biochar added during cultivation

prior to sowing a new ley, such seeded grasslands are already included in the 1.53 Gha of cropland. Surface application of up to 10 Mg C ha^{-1} biochar to range grasslands may be practical in some instances, but will be limited in regions prone to wind erosion and would be wholly inappropriate on grasslands managed by, or at high risk of, fire. We therefore estimate that, at most, 20% of the theoretical capacity from applying 50 Mg C ha^{-1} biochar to 2.5 Gha of global extensive pasture land might be practically achievable, giving a total additional storage capacity of 25 Pg C in pasture soils.

If soil capacity is not to be the limiting factor to biochar's potential for climate-change mitigation, alternative sites for biochar storage will need to be found. These might include deeper incorporation into soils (a depth of 30 cm effectively doubling the soil storage capacity), disused mines and quarries, aggregate in construction materials, or biochar-filled ditches along field boundaries and riparian zones to reduce agricultural leaching. However, we do not explicitly model other potential storage options for biochar than in soils other than to calculate when the maximum capacity of soils for biochar has been reached as outlined above. The model assumes that no further increases in GHG feedbacks due to soil N_2O or CH_4 fluxes or enhanced yields accrue beyond this point. We do not assume, however, that production ceases at this time, as other storage options may be available. The decomposition rate of biochar produced after this time is assumed to be the same as for biochar added to soil. Future research will need to establish whether and where more biochar may be sustainably stored in soils, and the suitability and availability of alternative sinks.

2.6.2 Tillage

Incorporation of biochar into soil requires tillage. We calculate CO_2 emissions from tillage using a value of 1.5 hr ha^{-1} tractor time at a diesel fuel consumption rate of 15.4 dm^3 hr^{-1}, giving 19.6 kg diesel ha^{-1} (assuming the density of diesel is 0.85 kg dm^{-3}) based on mean figures for rotary tillage from the EcoInvent life-cycle analysis database[137]. We then calculate CO_2 emissions using an energy density of 45 GJ kg^{-1} diesel, and a C-penalty of 0.019 Mg CO_2-C GJ^{-1}. We assume incorporation of biochar into soil is a one-off operation with all the biochar being added to the soil in a single batch.

2.6.3 Fertility and enhanced net primary productivity

The prime focus of this paper is an assessment of the potential of biochar for climate change mitigation rather than of its potential for agronomy. Nonetheless, an assessment of the potential for biochar to increase agricultural yields is a necessary part of our calculation. Increases in crop yield as a result of biochar additions will create a feedback by altering the NPP on croplands and increasing the mass of crop residues available as biochar feedstock. In order to model this response, we have used published data from field trials to estimate the expected response to biochar additions of different crops on soils of varying fertility, and combined this with an analysis of the global geographical distribution of these crop-soil combinations.

Soil fertility classifications were taken from a joint FAO / IIASA assessment of global agro-ecological zones[138]. Extracting from this just the layer of data regarding soil fertility constraints to crop production provides a 5-minute of arc resolution raster of soils categorised into seven classes of soil fertility constraints, namely; none, few, slight, moderate, severe, very severe, or unsuitable. This was combined with a 5-minute resolution map of global cropland distribution[86] to produce a global map of cropland, categorised by severity of soil-fertility constraints (Fig. 6 in the main paper). The global area of land under cultivation in each of the soil-fertility classes is given in Table S8.

Next, these map data were combined with a 5-minute resolution data on the geographical distribution of the harvested area and yield of 175 distinct crops of the world circa AD 2000[139]. Combining this with harvest indices or residue-to-crop ratios, we calculated the production of crop residues on soils of each fertility class. It should be noted that the assessment in ref 139 of the distribution of crop yields is based on a combination of a 5-minute resolution map of cropland distribution with lower resolution data on aggregated yields from geographic regions, the lower resolution aggregated yields being distributed equally over all cropland within the region. As the geographic regions from which the aggregated crop yield data are taken will in general contain soils of various qualities, the assumption that yields were homogeneously distributed over the cropland in the region will likely lead to an overestimation of yields on poor soils and a corresponding underestimation of yields on more fertile soils. Despite this shortcoming, no better high-resolution data on global crop distribution are currently available. Our analysis based on it will, therefore, likely lead to an overestimation of the potential to increase crop yields on poor soils, and thus runs counter to our general philosophy of making conservative assumptions where possible. The calculated values of the potential for increased crop residue yields to provide feedstock for further biochar production should therefore be interpreted as an upper bound on the expected size of this feedback.

2.6.4 Estimation of biomass-yield responses to biochar soil applications.

We approached this in two stages. First, we assumed that the actual yield response would progressively decrease from a maximum level seen only in soils having maximum soil fertility constraints (i.e., ranked as unsuitable). Thus, a soil having no soil-fertility constraints would show no yield response to amendment with biochar, whereas one having a moderate soil-fertility constraint would show 50% of the maximum response and one having a very severe soil fertility constraint would show 90% of the maximum response. These assumed yield responses, which are basically fractions of the potential yield response seen in a soil rated as unsuitable for cultivation, are summarized for different levels of soil-fertility constraints in Table S9.

We then calibrated these fractional responses by estimating the relative yield response to biochar application rates based on the very limited literature on this topic, which dominantly involves field and greenhouse studies with soils classified as having very severe to unsuitable soil-fertility constraints. This response was assumed to be equivalent to the maximum biomass yield response.

We converted all biochar application rate data to units of biochar-C ha^{-1}, estimating this value for one study where a biochar C analysis was not provided[140]. Relative biomass yields (RBY) were calculated as the ratio of biomass yield for a plot receiving biochar to the yield from an otherwise identical control plot that did not receive biochar, and then normalised per Mg C ha^{-1} of biochar applied. Most of these comparisons involved light fertiliser applications to both types of plots. To avoid fertility responses to non-carbonaceous constituents of biochar such as phosphorus and lime from poultry-litter feedstocks, only studies using biochar prepared from wood were included in the analysis. Poultry-litter biochars generally show excellent crop-yield responses (e.g. refs141,142), but only account for about 2% of the total biochar applications considered in this work.

In addition to these selection criteria, we did not use data from some studies where the results seemed extreme, and therefore unlikely to be generally replicated [Iswaran et al.[143]; Topoliantz et al.[144]; Yamoto et al. (peanut, Site A)[145]; Steiner et al. (sorghum)[146]; Baronti et al. (irrigated maize)[140]] or where insufficient information was presented to estimate a response (Oguntunde et al.[147]). In general, these excluded data showed very strong positive responses to biochar amendments. However, a single datapoint for irrigated maize (Baronti et al.[140]) was also excluded because it showed a 20% decrease in yield and was thus inconsistent with the remaining yield data for cereals collected under dryland or greenhouse conditions. We note that yield decreases at low biochar levels are observed sometimes during the first year after incorporation due to nitrogen immobilisation while the most labile C in the biochar is metabolised[148,136] (unless sufficient N is added to compensate for this effect), and hypothesize that this may be the cause of the decreased yield observed by Baronti et al.[140]. The dataset used to develop the relationship is shown in Table S10.

In general, we found that field and greenhouse data agreed well, but that leguminous species and cereals responded differently to biochar application, with cereals showing roughly a three-fold higher response. We also found instances where application rates greater than 50 Mg C ha^{-1} resulted in yield decreases. We thus developed relationships for leguminous species and cereals separately that are valid only for biochar application rates below 50 Mg C ha^{-1}. These are simply the slope of a line fit to the relative biochar yield data plotted in terms of the biochar application rate. The values obtained for each crop type and study type, together with the values used in our model, are given in Table S11, while plots of the data and fitted lines for each crop type are given in Fig.S9. As implemented in the model, one RBY value (0.022 ha Mg^{-1} C) is used for cereal crops including maize, wheat, and rice, and the legume RBY value (0.0066 ha Mg^{-1} C) is used for all other crops.

2.6.5 Impact on existing stocks of SOC

We identify three distinct ways in which biochar can affect the quantity of SOC. Firstly, as discussed in section 2.2.1, there will be reduced inputs of detritus to soil as a result of diverting biomass into biochar production rather than adding it directly to soil. Secondly, where biochar additions to soil result in greater plant biomass yield, the unharvested fraction of this enhanced yield (i.e., increased below-ground biomass) will increase the rate of input of detritus to soil and thus increase SOC stocks. We calculate this by assuming that increased above-ground yields as calculated above are matched by an equal increase in below-ground biomass, and that this biomass will add to SOC pools according to the equation 1 in section 2.2.1. Thirdly, as we discuss here, the presence of biochar in soils may alter the rates of humification, stabilisation and respiration of soil organic matter (SOM).

Greatly enhanced rates of mass loss from boreal forest (*Pinus sylvestris*) litter mixed with charcoal (50:50 mixture) have been observed, giving rise to concern that biochar additions to soils might lead to significant losses of native (non-biochar) soil organic carbon and a corresponding reduction in the net carbon sequestered[149]. Enhanced soil-respiration rates in the presence of BC have also been observed in incubation experiments[126,150,119], all with the caveat that C loss could not be attributed to

biochar or SOC. Where attribution was possible[126], both increased and decreased mineralization of SOC was observed in a 2-year incubation study. We should be wary, however, of extrapolating from short term laboratory incubations or litter-bag studies to long-term kinetics in soils in their native environment where other processes such as humification and mineral stabilization of SOC and development of complex soil microbial communities (including mycorrhiza that obtain their carbon from roots rather than soil) will also affect SOC stocks and may be influenced by BC. Indeed, in *terra preta* soils, where high BC concentrations have existed in soil for long periods, higher concentrations of BC are found to occur with higher concentrations of non-BC SOC relative to adjacent soils[151]. Too little is known, though, about the formation processes of *terra preta* to assume that the same increases in humified and mineral-stabilized SOC will be generated by additions of biochar to other soils globally.

In a study of BC-rich soils from historical charcoal blast-furnace sites[131], total SOC in both the BC-rich soils and adjacent soils was measured. The labile fraction of the soils was also determined by incubation. No statistically significant effect of BC concentration on labile-SOC concentration was found, with a 5% mean increase of labile-SOC stocks in BC-rich soils compared to adjacent soils (Fig. S11).

As these soils cover a climatic sequence with mean annual temperature ranging from 4 to 17 °C, and charcoal was deposited in all of these soils approximately 130 years ago, this study provides the best currently available data to estimate the effect of biochar additions to global soils over the course of the century timescale considered in this paper. It suggests that no significant effect may be expected. However, we note that the world's soils contain an estimated 1500 Pg organic C to 1 m depth[152]. Since only a small percentage change in this carbon pool would create a significant feedback, further research to determine the size of this effect with greater confidence is a high priority.

2.6.6 Soil N_2O emissions

Biochar additions have been observed to reduce soil nitrous oxide emissions by 50 – 80% at an application rate of 20 Mg ha^{-1} in an acid savanna oxisol[153], and by 70-80% at an application rate of 5 g kg^{-1} (equivalent to approximately 20 Mg ha^{-1} if incorporated to a depth of 0.3 m) in a slightly acidic to neutral upland mollisol developed on glacial till[150]. Although further research is required to determine the range of soil conditions under which reduced N_2O emissions will occur[48], the striking reduction noted in those studies that have measured this effect and the absence of studies that did not find a reduction in N_2O, mean that it is important at least to estimate how important this feedback might be. We begin by calculating a global mean emission rate of N_2O from cropland. Globally, 120 Tg N yr^{-1} is used in agriculture[154], of which 79% is applied to cropland[115], yielding a mean application rate of 62 kg N ha^{-1} yr^{-1} to the 1.53 Gha of global cropland. Direct soil emissions of N_2O-N, while highly variable, are on average 1% of applied N[116,155] plus a further 1 kg N_2O-N ha^{-1} yr^{-1} that is not related to quantity of applied fertilizer[155]. This gives us an estimated global mean of 1.6 kg N_2O-N ha^{-1} yr^{-1} from cropland (equivalent to 2.5 kg N_2O ha^{-1} yr^{-1}). We have then calculated annual avoided soil N_2O emissions (E_N) to be this mean emission rate multiplied by the area of land that has been amended by biochar (A_b) and a reduction factor R_N.

$E_N = R_N$ (2.5 kg N_2O ha^{-1} yr^{-1}) A_b

For the sensitivity and Monte-Carlo analyses we investigate values for R_N in the range 0-80%, and for the main results, given that these emission reductions have not been demonstrated over a wide range of conditions, we assume a modest value of R_N=25%.

2.6.7 Soil CH_4 flux

There is some reported evidence of reduced CH_4 emissions from soil after biochar additions[153]; however the evidence for this effect is weak, particularly given that dry soils are generally net sinks rather than sources of CH_4. A study of biochar amendments to paddy rice systems found no reduction in CH_4 emissions as a result of biochar additions[156]. Accordingly, we have assumed no effect of biochar on CH_4 emissions from soils. Enhanced CH_4 sinks of 150 mg CH_4-C m^{-2} yr^{-1} in dry soils have also been reported[157], and we have accordingly used the range 0-150 mg CH_4-C m^{-2} yr^{-1} in both the sensitivity and Monte-Carlo analyses, with a median value of 75 mg CH_4-C m^{-2} yr^{-1} used to generate Figures 2 and 3 in the main paper.

2.6.8 Fertiliser application

Biochar can improve fertiliser-use efficiency, thus lowering fertiliser input requirements and avoiding CO_2 emissions associated with fertiliser manufacture. Several studies have shown biochar-induced reductions > 50% in ammonium- and nitrate-N leaching[158]. Leaching from terra preta soils is also significantly lower than from adjacent soils suggesting that the reduced leaching effect persists over time[94]. Given that gaseous losses of N have not been found to increase with biochar application (and may, rather, decrease), reduced leaching losses of N imply an increase in N use efficiency[159] once N

immobilized by initial adsorption to biochar has been accounted for.

Current global N-fertiliser production is 120 Tg N yr^{-1} (ref. 154). We assume that CO_2 emissions due to N-fertiliser production, transport and application are 0.858 Mg C Mg^{-1} N (ref. 160), although values as large as 1.23 Mg C Mg^{-1} N have been estimated[84]. Energy requirements for phosphorus (P) and potassium (K) fertilisers are an order of magnitude lower[160], and are therefore omitted from this calculation. Thus, global fertiliser production is currently responsible for emission of approximately 103 Tg CO_2-C yr^{-1}. To estimate how much this might be reduced by biochar production, we begin by assuming that a 50% increase in N use efficiency will reduce global fertiliser production by 50% times the fraction of cropland that has been amended with biochar. After also taking into account reactive N converted to N_2 during pyrolysis (calculated as described in section 2.2.3 above), we estimate that 360 Tg CO_2-C equivalent cumulative emissions from fertiliser production over 100 years would be avoided in the Alpha scenario – approximately 0.5% of the total avoided emissions from biochar.

Some N will initially be immobilized following biochar application due to both microbial decomposition of the biochar labile fraction and to adsorption of N onto biochar. Although there may be numerous opportunities to supply some of this N by using biochar to remove N from effluent sources such as sewage, slurry and aquaculture, we make the conservative assumption that this immobilised N will have to be replaced by mineral fertiliser. Despite its agronomic importance, we do not include N immobilized by microbial decomposition of biochar in the overall greenhouse gas balance as it is a temporary effect and will over time become plant-available again as the microbes die. Furthermore, microbial N immobilization is a problem only when using biochar with a significant volatile or labile fraction[136,161], and can thus be minimised or eliminated by ensuring that only biochars that have a low content of volatiles are used. The quantity of N immobilized by adsorption is linearly related to the soil cation exchange capacity (CEC). Thus, a doubling of CEC will require a doubling of soil N in order to maintain the same concentration of N in solution. This allows us to make a 1st order estimate of adsorption immobilized N. In highly infertile soils, BC concentrations of 15 g kg^{-1} (equivalent to approximately 58 Mg ha^{-1} if biochar is incorporated to a depth of 0.3 m) increase soil CEC by a factor of two[162]. In fertile soils with a high native CEC, little or no increase in CEC as a result of biochar amendment is likely. Interpolating linearly between soil fertility classes, we thus estimate the biochar induced increase in CEC in soils of different fertility constraints to be as in Table S13.

The mean relative increase in CEC, weighted by land area under cultivation in each fertility classification (as given in Table S8), is thus 1.5. Thus, approximately 50% increase in N applications will be required during the first year of biochar application to maintain dissolved N concentrations. Over the course of a century then, this will require an additional half a year's equivalent of fertiliser production, leading to a total GHG emission of 52 Tg CO_2-C. This is approximately 14% of the avoided emissions due to increased N-use efficiency, and represents just 0.07% of total avoided GHG emissions due to biochar production in the Alpha scenario.

2.7 Rate of adoption

Although this model is designed to calculate the effect on GHG emissions from biochar, not its economics, we nevertheless must include an estimate of costs in order to make realistic assumptions about how rapidly production capacity could potentially be implemented. Published estimates of capital cost for biomass pyrolysis with power generation are given in Table S15.

We use a sigmoidal Gompertz function (Fig. S3) of the form $B_a = B_{max} \exp(-k_1 \exp(-k_2.(t-t_0))$ to model the rate at which annual biomass utilisation (B_a) approaches its maximum value (B_{max}). The point of inflection at which maximum slope is reached occurs when the 2nd derivative is zero at $t= (\ln(k_1)/k_2) – t_0$. The annual increase in biomass carbonisation rate is given by the 1st derivative $dB_a/dt = B.k_1.k_2 \exp(-k_1^2 – \ln(k_1))$. The parameters $k_1 = 11$, $k_2 = 0.25$ and $t_0 = 4$ were set such that the maximum rate of capital expenditure is reached after 15 years and, in the Alpha scenario, is 0.1% of 2007 global GDP, assuming a capital intensity of US$395 Mg^{-1} yr (dry feedstock), based on the highest estimate for an intermediate sized plant in Table S15 (German Energy Agency figures), and on a mean life expectancy of the plant of 20 years.

The expenditure rate on biochar capital of 0.1% GDP was chosen to be compatible with the estimate that total expenditure on climate mitigation needs to be in the order of 2% of global GDP[163] together with the assessment, based on the results of this model that, in the Alpha scenario, biochar might provide around 5% of the 15 Pg CO_2-C_e yr^{-1} abatement required (see Table S14) to stabilise atmospheric GHG concentrations. We did not include operating costs or feedstock costs in this analysis, since it has been estimated that together they are approximately equal to the value of the energy produced[164]. The estimate that maximum capital expenditure is reached after 15 years allows for a lead-time of approximately 5 years during which little plant capacity is commissioned to provide time for further research that is a prerequisite to large-scale deployment. Longer development times

than this are not expected to be necessary, since plant is already commercially available and early deployment is already under way particularly in regions with poor soils. At these rates, maximum production could be approached by mid-century.

2.8 Aspects not considered in the model

We have attempted to include in this model the most important effects of biochar on the net GHG balance. Greenhouse gases are not the only manner, though, in which biochar can impact on the climate system. Biochar production also has potential to affect both atmospheric aerosol loading and albedo.

Combustion of biomass is a major contributor to atmospheric soot emissions[165]. Pyrolysis of biomass in well-designed, modern plant does not entail significant atmospheric emissions of soot. Thus, replacing biomass combustion in developing regions with pyrolysis is likely to both reduce tropospheric BC concentrations and improve indoor air quality where biomass is used as a cooking fuel. Tropospheric BC is the second most important contributor to anthropogenic global warming after carbon dioxide[166] and, in the case of the Asian Brown Cloud, responsible for reduced monsoon rainfall and agricultural yields[167,168]. Poor indoor air quality as a result of biomass burning for cooking is the primary cause of acute respiratory disease in less developed countries where it is the chief cause of child mortality[169]. Thus, replacement of traditional cooking fires with pyrolysis stoves may also have important health benefits.

Although pyrolysis itself should not produce significant particulates, we should be aware of the possibility that mismanagement of biochar during storage, transport and field application could lead to significant amounts of black carbon dust becoming airborne. If the material is finely divided, then some of these particles might be small enough to remain airborne for significant periods and to travel large distances. It may therefore be necessary to pelletize biochar in order to minimise dust.

Soil darkening as a result of biochar addition may also reduce the albedo of cropland during bare fallow periods and while crop cover is sparse.

An evaluation of the size of these effects is, however, beyond the scope of this study and will require further research.

2.9 Monte-Carlo and sensitivity analyses

The ranges of the variables explored by sensitivity analyses are given in Table S7. For the Monte-Carlo anaylsis, random values were generated for all variables (with the exception of N_2O emission factor) with probabilities following a Gaussian distribution with mean equal to the estimated value and standard deviation set such that the range is a 95% confidence interval. That is, the standard deviation, σ (determined numerically), is such that $\Phi_{\mu,\sigma^2}(X_H) - \Phi_{\mu,\sigma^2}(X_L) = 0.95$, where, $\Phi_{\mu,\sigma^2}(X)$ is the Gaussian cumulative distribution function $\Phi[(X – \mu)/\sigma]$, μ is the mean, X_L is the lower limit of the range and X_H is the upper limit of the range. Random values for the N_2O emission factor were generated on a flat probability distribution within the given range as there was no rationale to suggest that values suggested by Ref. 78 are more or less likely than the IPCC default values.

3 Supplementary Figures

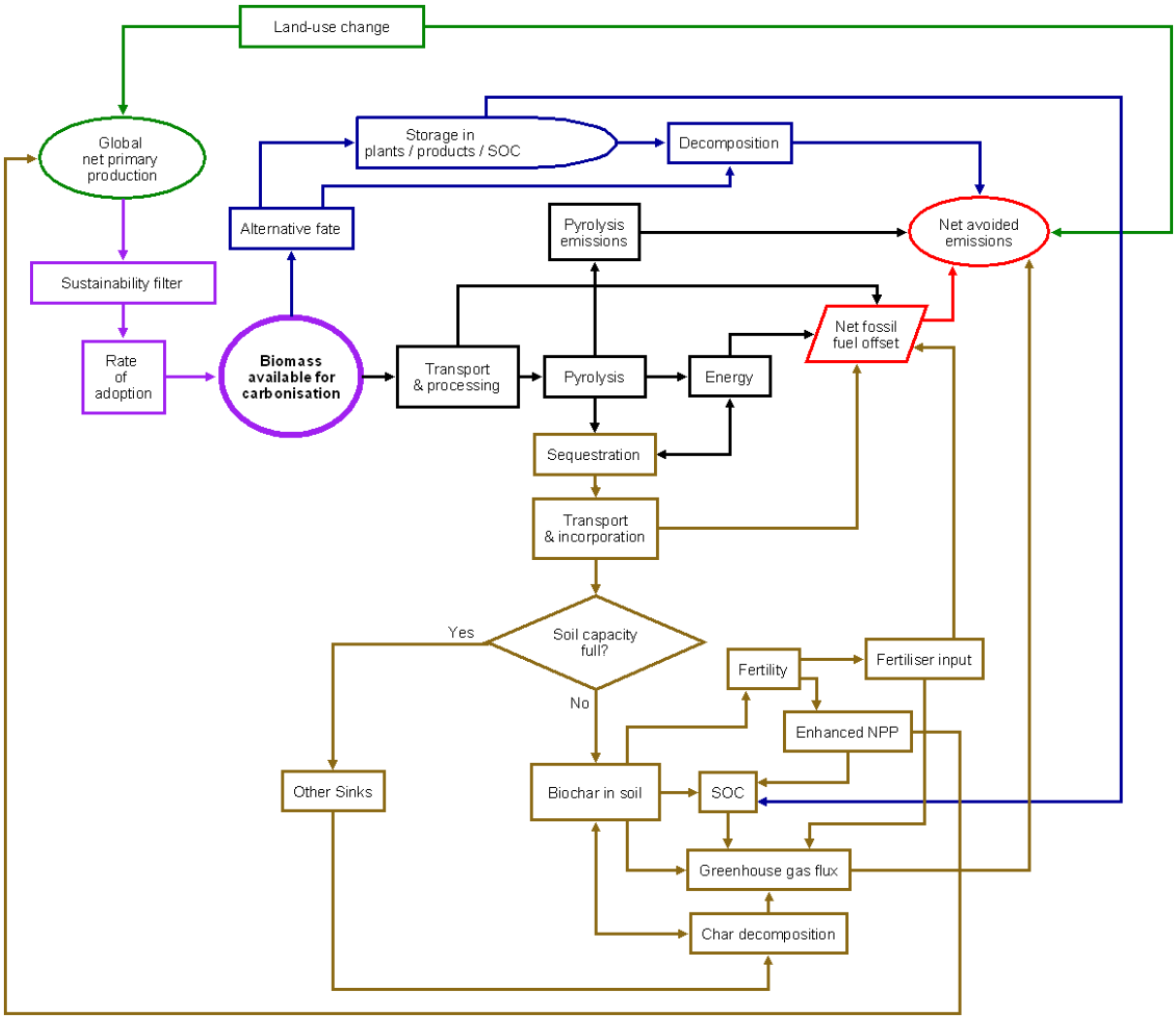

Figure S1: Schematic of biochar greenhouse gas assessment model, BGRAM 1.0

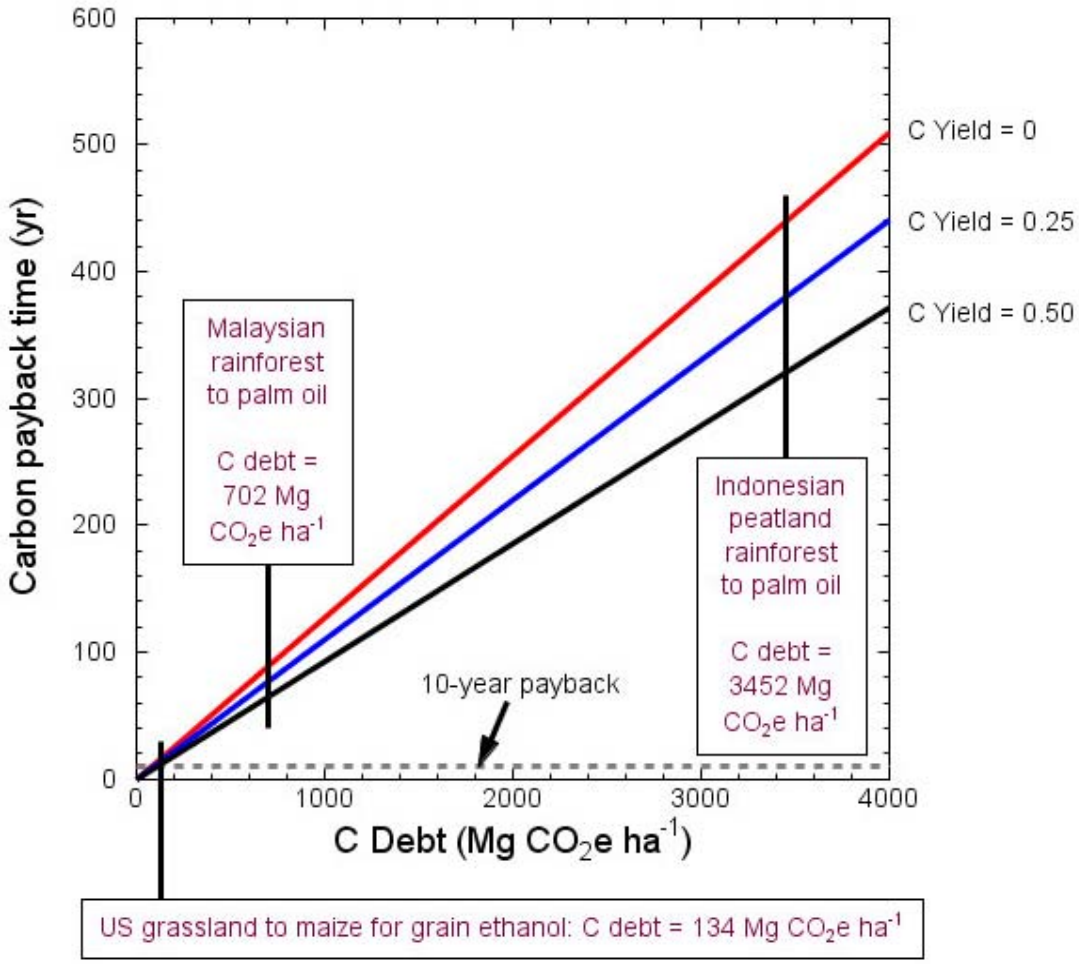

Figure S2: Payback time for the carbon debt incurred by cropping for biochar in dedicated biomass plantations. Estimates assume annual pyrolysis of 15 Mg ha^{-1} of woody biomass after conversion of the land to dedicated biomass production. No increase in this biomass production rate due to biochar amendment is assumed. These estimates do not include any contribution from changes to soil N_2O emissions that may arise from increased N-fertiliser use on the biomass crops relative to the prior land use, nor do they include any mitigation of this potential increase in soil N_2O emission due to biochar additions to the soil. Example values of carbon debt are from ref 70.

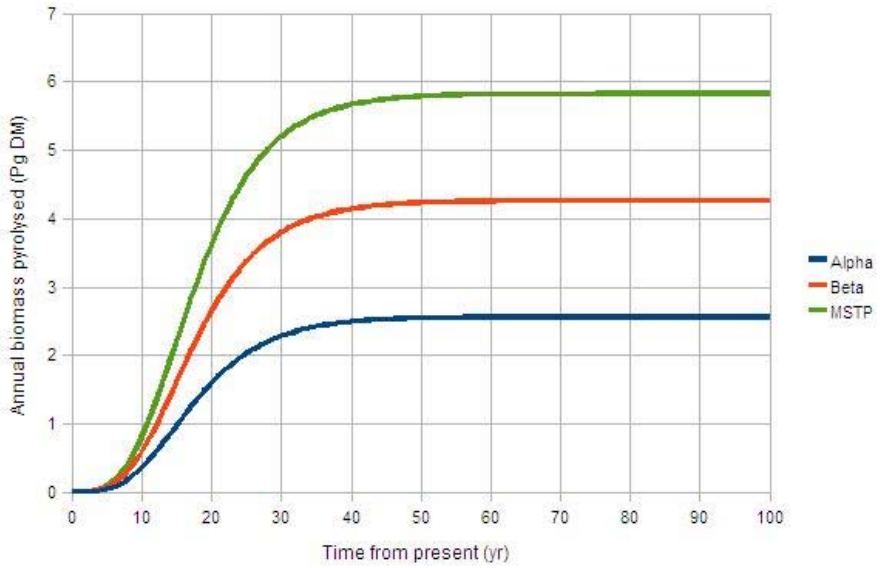

Figure S3: Annual global biomass pyrolysed as a function of time in each of the scenarios

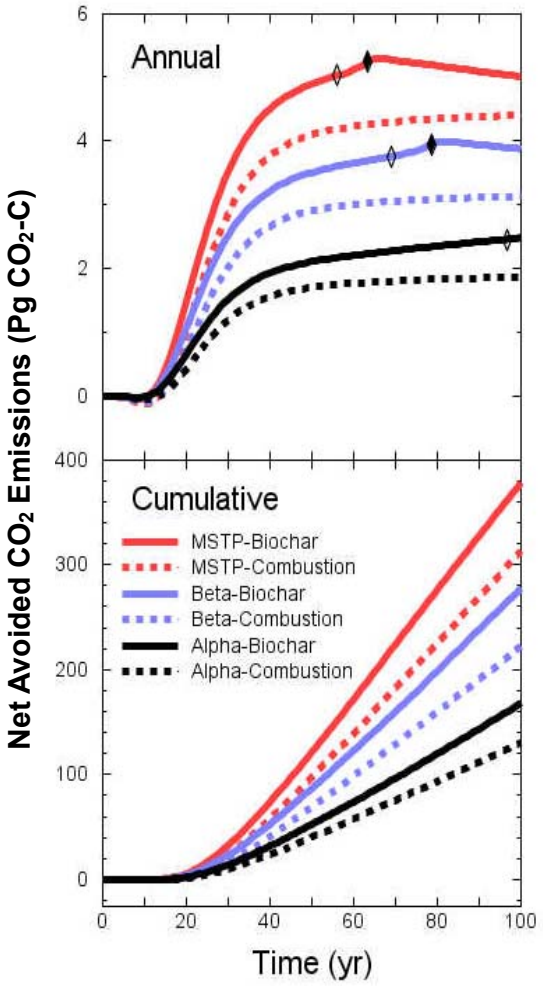

Figure S4: Net avoided CO_2 emissions relative to current use of biomass that are attributable to sustainable biochar production (solid lines) or biomass combustion (dashed lines) over 100 yr for the three model scenarios[a]. (top) annual avoided emissions; (bottom) cumulative avoided emissions. Diamonds indicate transition period when biochar capacity of top 15 cm of soil fills up and alternative disposal options will be needed.

[aa]All data used to produce plots in Figs S4-S6, Fig. S8 and Figs 2 & 3 in the main paper are available as an Excel spreadsheet in the Supplementary Dataset online.

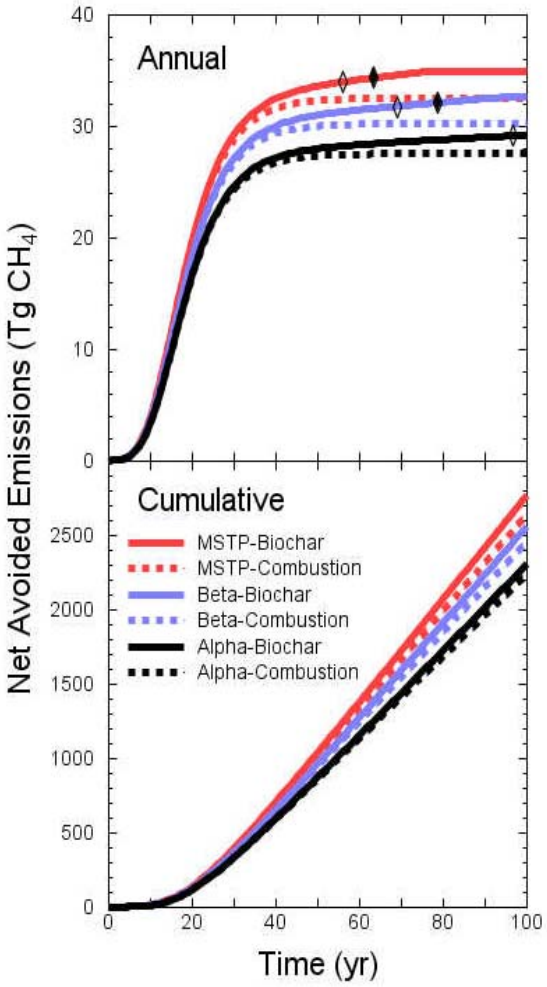

Figure S5: Net avoided CH_4 emissions relative to current use of biomass that are attributable to sustainable biochar production (solid lines) or biomass combustion (dashed lines) over 100 yr for the three model scenarios. (top) annual avoided emissions; (bottom) cumulative avoided emissions. Diamonds indicate transition period when biochar capacity of top 15 cm of soil fills up and alternative disposal options will be needed.

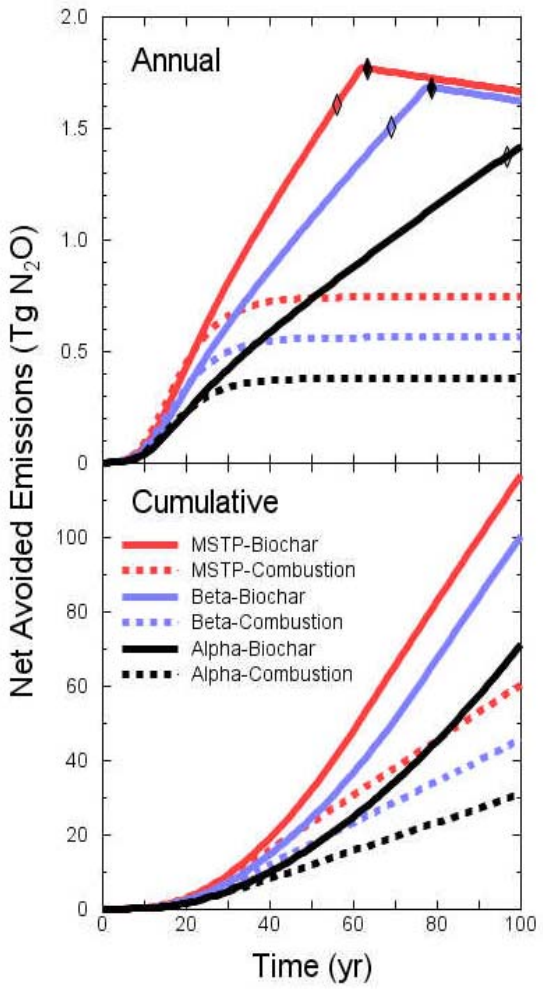

Figure S6: Net avoided N_2O emissions relative to current use of biomass that are attributable to sustainable biochar production (solid lines) or biomass combustion (dashed lines) over 100 yr for the three model scenarios. (top) annual avoided emissions; (bottom) cumulative avoided emissions. Diamonds indicate transition period when biochar capacity of top 15 cm of soil fills up and alternative disposal options will be needed.

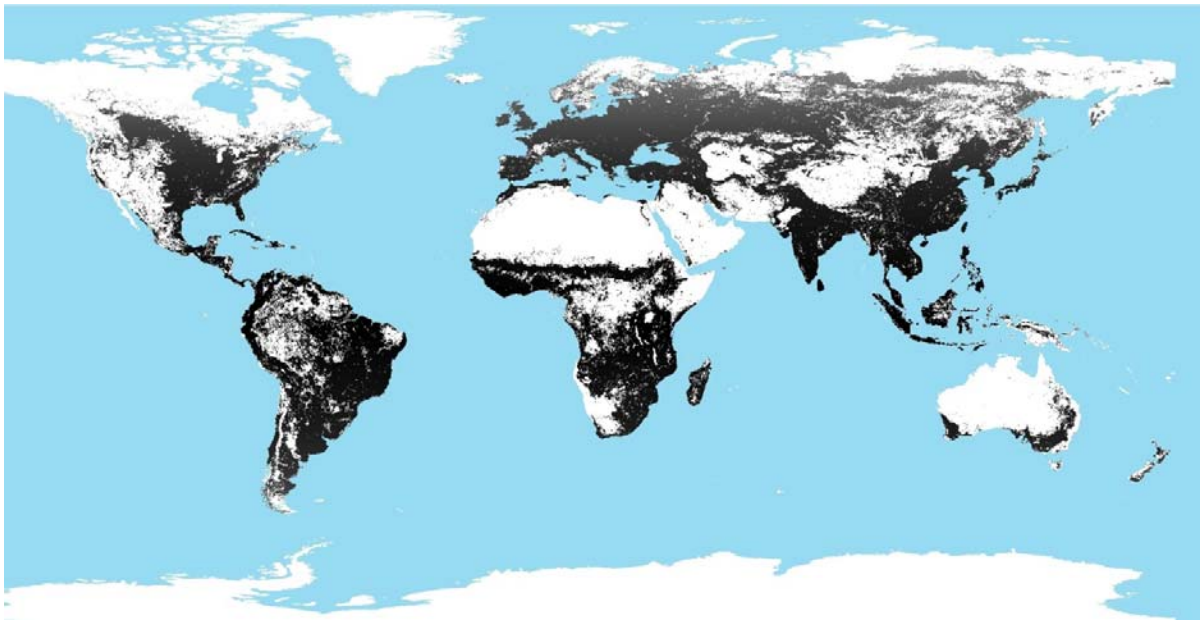

Figure S7: Land area over which biochar production is assumed (for the purpose of calculating transport distances) to be dispersed (5' of arc resolution). Every grid cell in which some feedstock from existing agricultural cropland is available has been coloured black. Note that this does not imply that all of this land area will be used for biochar production or soil incorporation as, in general, only a fraction of any individual black gridcell will contain cropland.

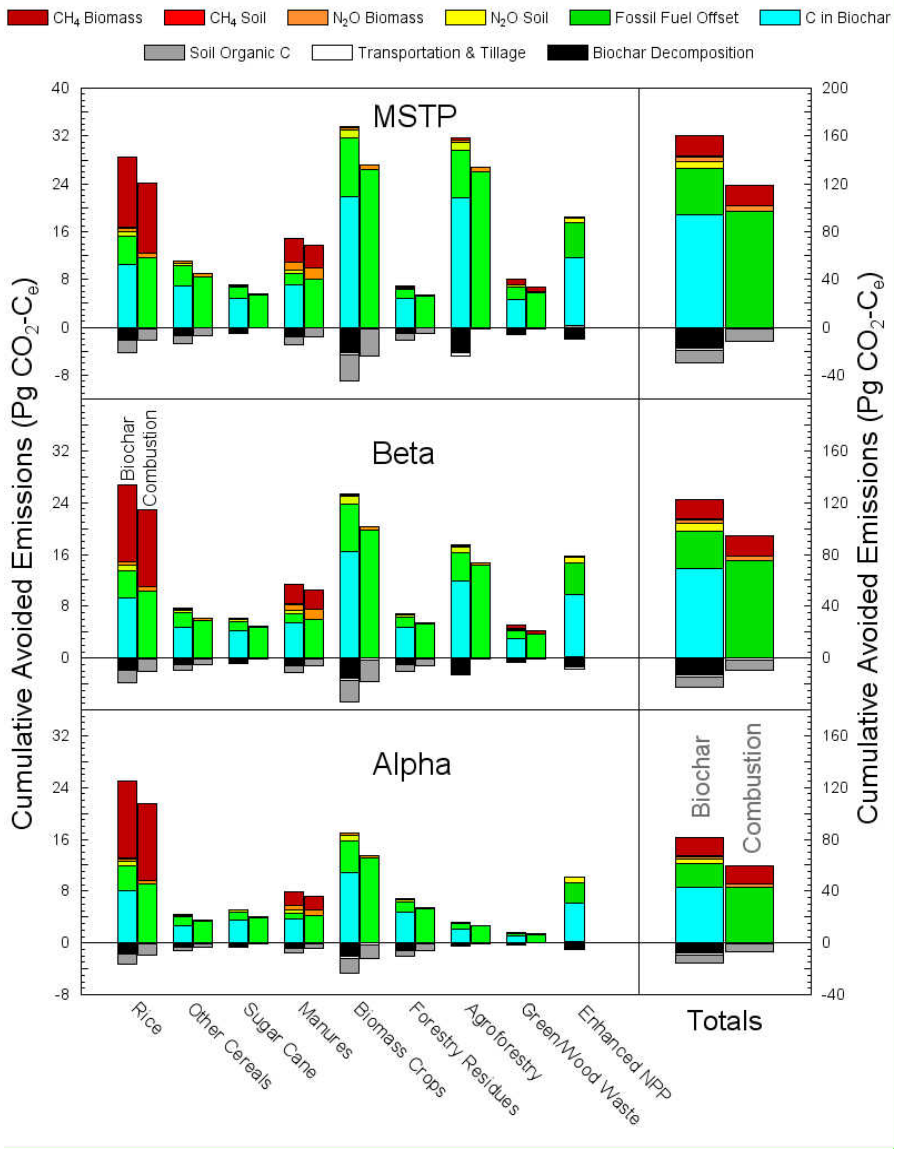

Figure S8: Breakdown of cumulative avoided GHG emissions (Pg CO_2-C_e) from sustainable biochar production and biomass combustion for the three model scenarios over 100 years by feedstock and factor[b]. Left side of figure displays results for each of eight feedstock types and the additional biomass residues attributed to NPP increases from biochar amendments; right side displays total results for each scenario. For each pair of columns, results for biochar are given in the left column and for biomass combustion in the right column. For each column, the total emission-avoiding and emission-generating contributions are given respectively by the height of the columns above and below the zero line. The net avoided emissions are calculated as the difference between these two values. Within each column, the portion of its contribution due to each of six emission-avoiding mechanisms and three emission-generating mechanisms is shown by a different color. These mechanisms (from top to bottom within each column) are 1) avoided CH_4 from biomass decay (CH_4 Biomass), 2) increased CH_4 oxidation by soil biochar (CH_4 Soil), 3) avoided N_2O from biomass decay (N_2O Biomass), 4) avoided N_2O due to soil biochar (N_2O Soil), 5) fossil fuel offsets due to pyrolysis energy production (Fossil Fuel Offset), 6) avoided CO_2 emissions due to C

[b]All data used to produce plots in Figs S4-S6, Fig. S8 and Figs 2 & 3 in the main paper are available as an Excel spreadsheet in the Supplementary Dataset online.

stored as biochar (C in Biochar), 7) decreased C stored as soil organic matter due to diversion of biomass to biochar (Soil Organic C), 8) CO_2 emissions from transportation and tillage activities (Transportation and Tillage), and 9) CO_2 emissions from decomposition of biochar in soil (Biochar Decomposition).

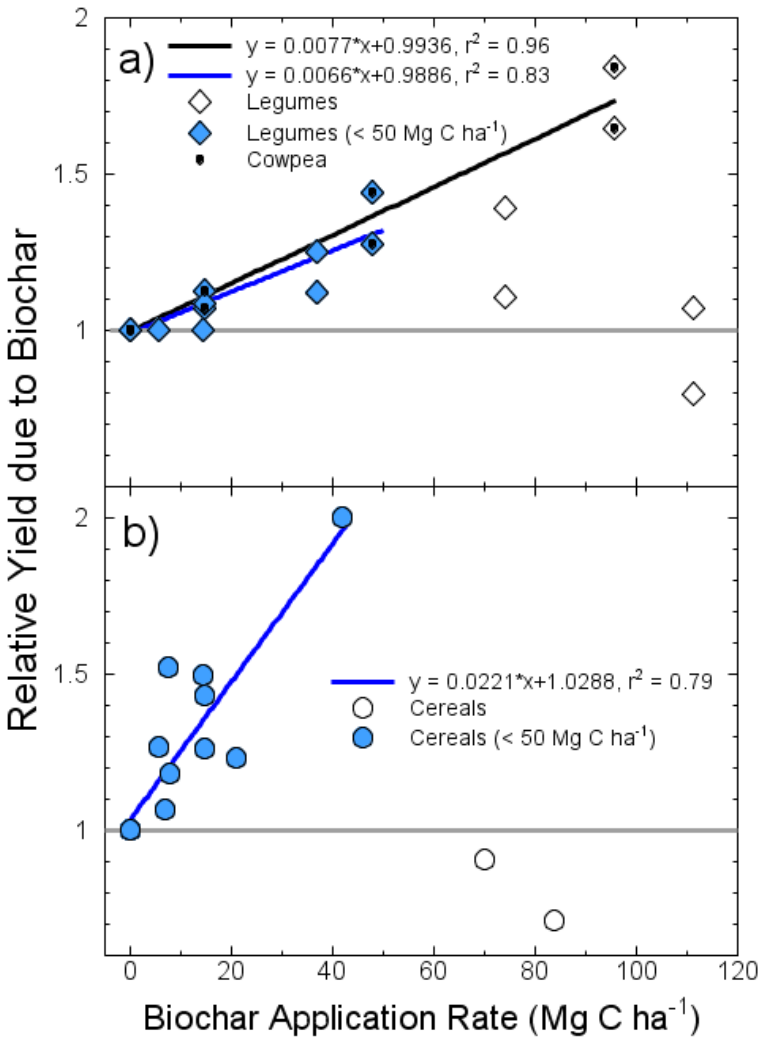

Figure S9: Relative biomass yields for biochar-amended soils. Results are calculated from data presented by field and greenhouse studies of (**a**) leguminous and (**b**) cereal crops and summarized in Table S10. Relative yield of 1 (shown by horizontal grey line) implies no yield response to biochar application. Soils are assumed to have maximum fertility constraints (i.e., to be "unsuitable" for cultivation) and yield responses thus are maximal values. Slopes of regression lines correspond to Relative Biochar Yield (RBY) factor used in model and listed in Table S11.

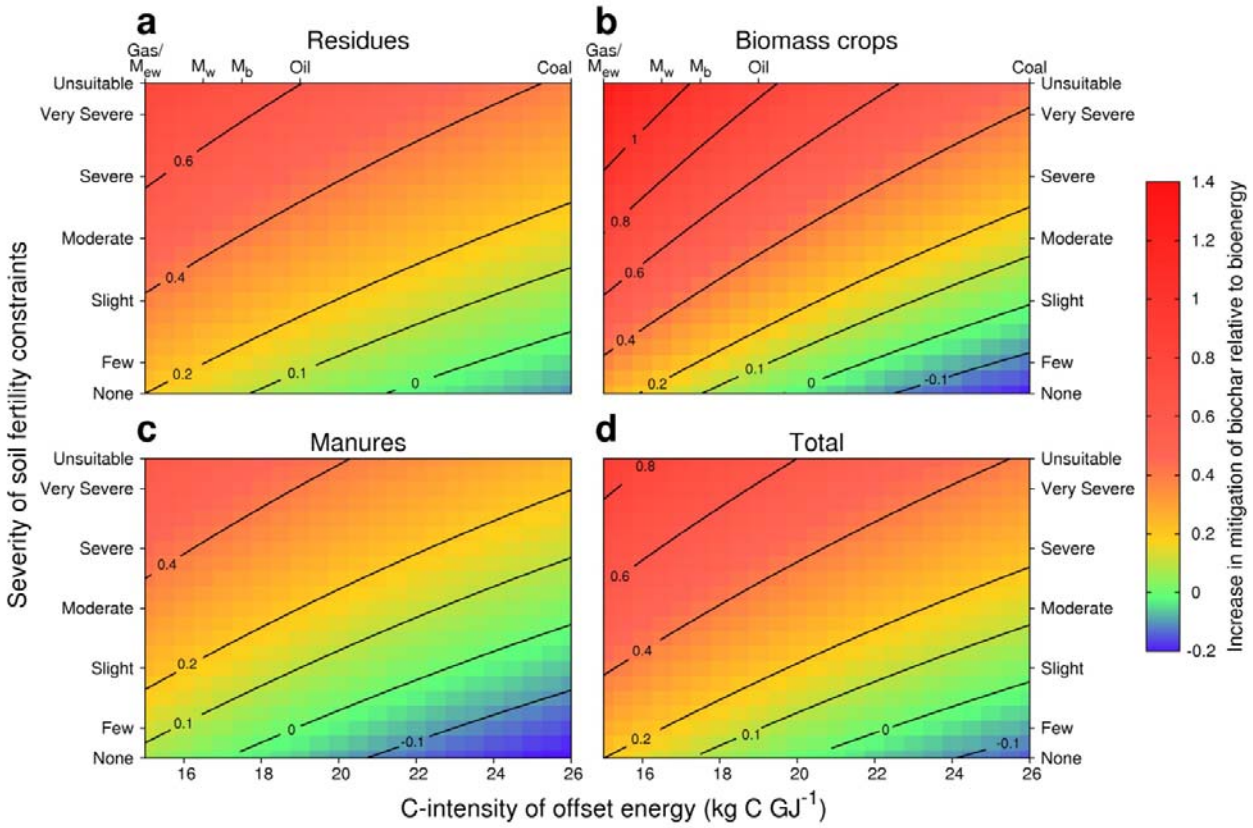

Figure S10: Cumulative mitigation potential of biochar relative to bioenergy as a function of both soil fertility and C-intensity of the type of energy being offset (in the Alpha scenario). Points M_{ew}, M_w and M_b on the x-axis refer to the current world electricity mix, the current world primary energy mix, and the baseline energy mix assumed in our scenarios, respectively. The relative mitigation is calculated as cumulative avoided emissions for biochar minus those for bioenergy, expressed as a fraction of the avoided emissions for bioenergy (e.g., a value of 0.1 indicates that the cumulative mitigation impact of biochar is 10% greater than that of bioenergy, a value of -0.1 indicates that it is 10% lower, and a value of zero indicates that they have the same mitigation impact). The soil-fertility classifications marked on the vertical axis correspond to the soil categories mapped in Fig. 6. Panel a (Residues) includes agricultural and forestry residues, together with green waste. Panel b (Biomass crops) includes both dedicated biomass crops and agroforestry. Panel c (Manures) includes cattle, pig and poultry manures. Panel d (Total) includes all feedstocks in the scenario.

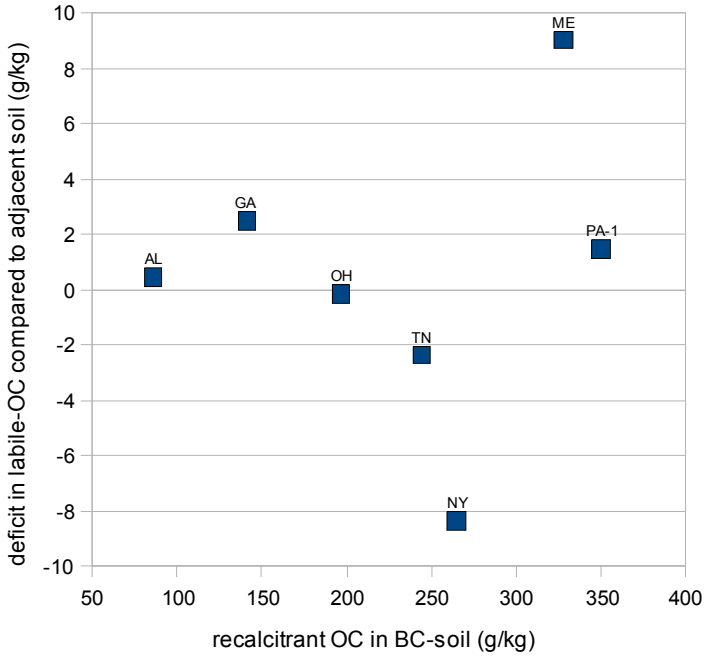

Figure S11: Effect of soil black carbon concentration on labile soil organic carbon pool. X-axis gives the concentration of recalcitrant organic carbon (OC) in black-carbon amended soil. Y-axis shows the difference in labile OC concentration between these soils and similar adjacent soils that have not been amended with black carbon (derived from Cheng et al.[132])

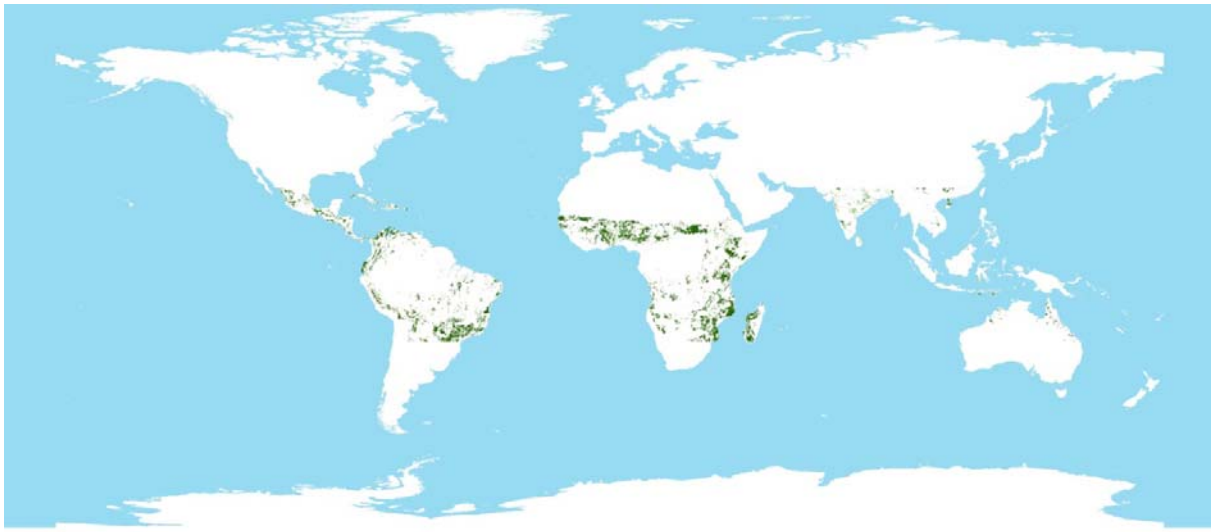

Figure S12: Land currently under pasture that is suitable for tropical silvopasture (5' of arc resolution). Although currently used as pasture, this land occurs in agroecological zones that have no serious constraints to rain-fed crop production.

4 Supplementary Tables

Table S1: Global quantities of non-rice straw and stover available as feedstock for pyrolysis

Region	Non-rice straw & stover (Tg DM yr^{-1})				
	Total generated[a]	Fodder[a]	Available for pyrolysis		
			Alpha	Beta	MSTP
East Asia	539.8	116.9	18.0	72.0	126.0
East Europe	146.2	14.8	21.8	36.4	51.0
Latin America & Caribbean	279.0	120.4	0.0	0.0	5.1
North & West Africa	122.4	52.8	0.0	0.0	2.2
North America & Oceania	388.4	11.4	85.7	124.5	163.3
South & Central Asia	269.0	195.8	0.0	0.0	0.0
Sub-Saharan Africa	218.0	141.1	0.0	0.0	0.0
Western Europe	183.6	5.8	40.1	58.5	76.8
Total	2146.4	659.1	165.6	291.4	424.5

[a] Derived from Ref 62

Table S2: Summary of biomass availability in scenarios (C content derived from values given in section 5)

Feedstock	Biomass availability in scenario (Pg yr^{-1})					
	Alpha		Beta		MSTP	
	DM	C	DM	C	DM	C
Cereals excluding rice	0.17	0.07	0.29	0.13	0.42	0.18
Rice	0.52	0.22	0.60	0.25	0.67	0.28
Sugar cane	0.20	0.09	0.24	0.11	0.27	0.13
Manure	0.31	0.10	0.45	0.14	0.59	0.19
Biomass crops	0.63	0.30	0.94	0.60	1.25	0.60
Harvested wood	0.05	0.03	0.13	0.07	0.21	0.11
Forestry residues	0.29	0.14	0.29	0.14	0.29	0.14
Agroforestry	0.13	0.06	0.70	0.34	1.28	0.62
Green waste	0.009	0.004	0.05	0.02	0.07	0.03
TOTAL	2.3	1.0	3.7	1.6	5.1	2.3

Table S3: Biomass properties used for modeling. Composition and energy content of biomass feedstocks used in BGRAM (and, where multiple sources are available, their standard deviations, "s.d."), expressed on a dry ash-free (DAF) and dry-mass (DM) basis. Except where indicated, data are derived from phyllis database at http://www.ecn.nl/phyllis/. HHV = highest heating value. LHV = lowest heating value.

Biomass	**Water**	**Ash**	**HHV**	**LHV**	**C**	**N**	**HHV**	**LHV**	**C**	**N**	**Fraction of category**
	%	%	GJ Mg^{-1} DAF	GJ Mg^{-1} DAF	% DAF	% DAF	GJ Mg^{-1} DM	GJ Mg^{-1} DM	% DM	% DM	%
	[% s.d.]	[% s.d.]	[% s.d.]	[% s.d.]	[% s.d.]	[% s.d.]	[s.d.]	[s.d.]	[s.d.]	[s.d.]	
Rice Hulls	9.6	17.8	18.67	17.34	48.7	0.83	15.3	14.3	40.0	0.7	11.53
	[16]	[23.4]	[6]	[7]	[3]	[78]	[1.2]	[1.2]	[2.4]	[0.5]	
Rice Straw	28.4	15.3	18.83	17.52	48.8	1.39	15.9	14.8	41.3	1.2	88.47
	[123]	[39]	[3]	[3]	[7]	[90]	[1.2]	[1.1]	[4.1]	[1.1]	
Rice hulls + straw	26.23	15.59	18.81	17.49	48.79	1.33	15.88	14.77	41.18	1.12	
Wheat Straw	12.1	7.4	19.56	18.18	48.9	0.88	18.1	16.8	45.3	0.8	45.10
	[97]	[59]	[5]	[5]	[3]	[86]	[1.2]	[1.2]	[2.5]	[0.7]	
Maize Stover	19.1	7.9	19.08	17.84	44.4	1.4	17.6	16.4	40.9	1.3	31.39
	[158]	[67]	[3]	[4]	[29]	[98]	[1.1]	[1.2]	[12.1]	[1.3]	
Maize cobs	24.7	6.7	17.46	16.15	46.8	1.55	16.3	15.1	43.7	1.4	6.12
	[142]	[88]	[7]	[9]	[5]	[119]	[1.5]	[1.7]	[3.5]	[1.7]	
Barley Straw	17.9	6.3	18.68	17.48	47.5	1.16	17.5	16.4	44.5	1.1	10.93
	[146]	[78]	[7]	[8]	[5]	[116]	[1.5]	[1.6]	[3.2]	[1.3]	
Rye straw	24.7	5.7	17.55	16.33	47.9	1.1	16.6	15.4	45.2	1.0	1.80
	[142]	[89]	[7]	[9]	[5]	[129]	[1.5]	[1.6]	[3.3]	[1.3]	
Sorghum straw	6	6.1	17.88	16.6	45.2	0.61	16.8	15.6	42.4	0.6	4.48
	[0]	[19]	[12]	[12]	[11]	[117]	[2.0]	[1.9]	[4.7]	[0.7]	
cereals ex rice	15.64	7.29	19.04	17.74	46.94	1.11	17.64	16.44	43.52	1.02	
Sugar cane Bagasse	50[192]	5.8	19.34	18.04	49.3	0.56	18.22	16.99	46.4	0.5	50
	[10][192]	[77]	[4]	[4]	[3]	[102]	[1.13]	[1.05]	[2.6]	[0.5]	
Sugar cane trash	6.5	5.5			49.7	0.74			47.0	0.7	50
Sugar cane total	15.15	5.65	19.34	18.04	49.5	0.65	18.25	17.02	46.70	0.61	
Hardwood	16.5	2.4	19.71	18.38	49.8	0.41	19.2	17.9	48.6	0.4	29
	[90]	[121]	[5]	[5]	[4]	[107]	[1.1]	[1.0]	[2.4]	[0.4]	
Softwood	19.9	0.9	20.3	18.95	51.4	0.16	20.1	18.8	50.9	0.2	71
	[93]	[131]	[6]	[6]	[4]	[75]	[1.2]	[1.1]	[2.1]	[0.1]	
Wood	18.91	1.34	20.13	18.78	50.94	0.23	19.86	18.53	50.26	0.23	

Miscanthus	28.9	3.6	19.77	18.52	49.6	0.52	19.1	17.9	47.8	0.5	50
	[56]	[38]	[3]	[3]	[2]	[52]	[0.6]	[0.6]	[1.2]	[0.3]	
Switchgrass	12.1	6	19.17	17.83	49.4	0.64	18.0	16.8	46.4	0.6	50
	[19]	[24]	[3]	[4]	[5]	[29]	[0.6]	[0.7]	[2.4]	[0.2]	
Herbaceous biomass crops	20.5	4.8	19.77	18.52	49.5	0.58	18.82	17.63	47.13	0.55	
Willow	10.9	1.9	19.83	18.48	49.8	0.61	19.4	18.1	48.9	0.6	
	[76]	[48]	[4]	[4]	[2]	[51]	[0.8]	[0.7]	[1.1]	[0.3]	
Cattle manure	38.1	40.6	21.36	19.85	50.4	3.36	12.7	11.8	29.9	2.0	
	[83]	[51]	[16]	[16]	[10]	[64]	[4.9]	[4.5]	[10.9]	[1.5]	
Pig manure	59.3	25.3	21.19	19.91	50.4	3.56	15.8	14.9	37.6	2.7	
	[70]	[56]	[1]	[0]	[9]	[20]	[3.0]	[2.8]	[7.9]	[0.7]	
Chicken manure	44.4	22.8	19.68	18.5	45	5.74	15.2	14.3	34.7	4.4	
	[58]	[30]	[8]	[10]	[18]	[22]	[1.8]	[1.9]	[7.0]	[1.1]	
forest residues	29.6	1.2	18.32	17	48.8	0.71	18.1	16.8	48.2	0.7	
Green waste	25.4	16.3	22.37	20.9	52.1	0.77	18.7	17.5	43.6	0.6	
	[61]	[73]	[20]	[20]	[13]	[55]	[4.6]	[4.3]	[8.4]	[0.4]	

Table S4: Biochar properties used for modeling. Carbon, nitrogen, and energy content of biochars prepared from different feedstock classes as used in BGRAM. (HHV = higher heating value)

		cereals ex. rice	rice	sugar cane	cattle manure	pig manure	poultry manure	biomass crops (herb)	biomass crops (wood)	wood waste	felling losses	Agro-forestry	green waste
Biochar HHV	GJ Mg^{-1}	21.4[a]	16.47[b]	26.5[c]	11.0[d]	11.0[d]	11.0[d]	19.1[e]	31.2[f]	31.2[f]	31.2[f]	33.2[g]	26.4[h]
Biochar C %	Mg C Mg^{-1} biochar	59.9[a]	56.0[i]	69.9[j]	0.25[k]	0.25[k]	0.25[k]	56.8[l]	75.0[f]	75.0[f]	75.0[f]	63.0[g]	67.5[h]
Biochar N %	Mg N Mg^{-1} biochar	0.67[a]	0.60[m]	0.51[n]	1.0[o]	1.4[o]	2.3[k]	1.0[l]	0.31[f]	0.31[f]	0.31[f]	0.8[g]	0.18[p]

[a] Mean of wheat straw char in Refs 193-196
[b] Mean of values derived from Refs 197,198 (calculated using Boie equation[199]).
[c] Ref 200 gives 18.9-23 MJ kg^{-1} (gasifier char); Ref 198 (calculated using Boie equation), 27.36 MJ kg^{-1} (fast pyrolysis char); Ref [201], 32.4 and 30.7 MJ kg^{-1} (fast pyrolysis char); Ref [202], 27.7 MJ kg^{-1} (slow pyrolysis); Ref [203] gives 26.2, 24.8, 27.3 for slow pyrolysis char; we use mean of slow pyrolysis chars = 26.5 MJ kg^{-1}
[d] No data found for cattle or pig manure chars, therefore we have assumed value given for chicken manure in Ref [204] for all manure chars.
[e] Mean of 20.1 MJ kg^{-1} from Ref 205 and 18.04 MJ kg^{-1} derived from Ref 206 using Milne equation (both sources for switchgrass char).
[f] Mean of 18 sources (all sources in wood char category excluding demolition wood) in Ref 207.
[g] Based on *Leucaena leucocephala* char in Ref 208.
[h] Value for yard trimmings estimated as mean of values for straw and wood.
[i] Ref 197
[j] Ref 198 gives 71.41% (fast pyrolysis); Ref 201 gives 85.6, 81.5% (vacuum pyrolysis) at 500-530˚C; Ref 202 gives 71.6% (slow pyrolysis 380˚C); Ref 203 gives 68.1%, 68.2%, 71.6% (slow pyrolylsis at 400 C); we use mean of slow pyrolysis chars = 69.88%
[k] Mean of Ref 106 and non-activated char from Ref 142
[l] Mean of Refs 205,206 for switchgrass
[m] Mean of Refs 197,198,209
[n] Ref [198] gives 1.77%, (fast pyrolysis); Ref 201 gives 1.3%, 0.8% (gasifier char); Ref 202 gives 0.53% (slow pyrolylsis); Ref 203 gives 0.5%, 0.5% and 0.5% (slow pyrolysis); we use mean of slow pyrolysis = 0.508%).
[o] Calculated by assuming the same ratio between biochar N% to biomass N% as for chicken manure
[p] Ref 210

Table S5: Methane emission factors for rice straw

Rice methane emissions kg CH_4-C kg^{-1} straw	*Source*
0.069	Jermsawatdipong *et al*., 1994[170]
0.010	Singh *et al*., 1996[171]
0.043	Lee *et al*., 1997[172]
0.019	Sass & Fisher, 1997[173]
0.042	Kumagai and Konno, 1998[174]
0.179	Corton *et al*., 2000[175]
0.126	Goto *et al*., 2000[176]
0.19	Naser *et al*., 2007[177]
0.004	Liou *et al*., 2003[178]
0.076	Mean

Table S6: Estimates of biochar decay half-life

Source	$T_{½}$ /yr	Range /yr	Notes
Hammes et al. (2008)[129]	210	130-380	Measurements of in situ concentrations of historic BC from biomass burning in the Russian Steppe
Kuzyakov et al. (2009)[126]	1400		Incubation of laboratory biochar formed from perennial ryegrass at an unspecified temperature in Haplic Luvisol and Loess soils
Bird et al. (1999)[133]		< 100	BC concentration in fire-protected soils relative to regularly burned soils in sandy savanna soils in Zimbabwe.
Cheng et al. (2008)[131]	930		BC soils from historic blast furnace sites. SOC decomposition measured during incubation, and extrapolated to *in situ* conditions with a mean annual temperature of 10°C.
Nguyen et al. (2008)[134]	6		Western Kenyan agricultural fields cleared from forest by fire at 2-100 years BP. Short residence time was attributed to a combination of labile component of BC and to initially rapid vertical transport. Following early losses, BC concentrations stabilised after approximately 30 years.
Liang et al. (2008)[179]		Several centuries to several millennia	Measurements of turnover rate of aged char in terra preta soil.
Preston and Schmidt (2006)[180]	6600		Coastal temperate rain forest on western Vancouver Island, British Columbia
Lehmann et al. (2008)[92]		900-1800	1877 soil samples from Australian National Soil Archive analysed for BC and SOC content, and fitted to models of BC formation and loss rates.

Table S7: Range, mean and standard distribution of variables used in sensitivity and Monte-Carlo analyses

Variable	Range	Estimated value	σ
Pyrolysis C yield (%)	40 – 60	49	5.2
Pyrolysis energy efficiency (%)	65 – 85	75	5.1
C intensity of fuel offsets (kg CO_2-C GJ^{-1})	15 – 26	17.5	1.52
Biochar labile fraction (%)	5 – 30	15	5.9
½ life of biochar labile fraction (yr)	1 – 25	20	3.0
½ life of biochar recalcitrant fraction (yr)	50 – 1000	300	152
Increase in CH_4 reoxidation rate as a result of biochar (mg CH_4 –C m^{-2} yr^{-1})	0 – 150	75	38.2
Relative biomass yield (RBY) modifier (i.e. the fraction of the RBY predicted in section 1.7.4 that is realised in practice)	0.5 – 1.5	1	0.255
% reduction in soil N_2O emissions due to biochar application	0 – 80	25	15
Biomass N_2O emission factor (kg N_2O-N kg^{-1} applied-N)	1% – 5%	1.05%[a]	Values populated from a flat probability distribution constrained within the stated range

[a] weighted mean of IPCC default values for different feedstocks as outlined in 1.3.3

Table S8: Cultivated land area classified by soil fertility constraints

Severity of soil fertility constraints	Global area of cropland (Gha)
None	0.31
Few	0.29
Slight	0.21
Moderate	0.32
Severe	0.18
Very severe	0.13
Unsuitable	0.09

Table S9: Assumed biomass yield responses to biochar amendments as fraction of the maximum potential response

Severity of soil fertility constraint	Biomass yield response as fraction of potential maximum
None	0
Few	0.1
Slight	0.3
Moderate	0.5
Severe	0.7
Very severe	0.9
Unsuitable	1.0

Table S10: Dataset used to develop function describing biomass yield response to biochar soil amendments

Reference	Greenhouse / Field	Crop	Biochar C level (Mg C ha^{-1})	NPK Level (kg ha^{-1})	Yield (Mg ha^{-1})	Relative Biomass Yield
Yamoto et al. 2006 (Ref. 145)	Field	Cowpea, site A	0	75-75-75	12	1.00
Ref. 145	Field	Cowpea, site A	14.7	75-75-75	13.5	1.13
Ref. 145	Field	Cowpea, site B	0	75-75-75	11.5	1.00
Ref. 145	Field	Cowpea, site B	14.7	75-75-75	12.3	1.07
Ref. 145	Field	Peanut, site B	0	75-75-75	6	1.00
Ref. 145	Field	Peanut, site B	14.7	75-75-75	6.5	1.08
Major 2009 (Ref 181)	Field	Soybean	0	16-10-104	--[ab]	1.00
Ref 181	Field	Soybean	5.8	16-10-104	--[ab]	1.00
Ref 181	Field	Soybean	14.6	16-10-104	--[ab]	1.00
Ref. 145	Field	Maize, site C	0	0	9	1.00
Ref. 145	Field	Maize, site C	14.7	0	11.3	1.26
Ref. 145	Field	Maize, site C	0	75-75-75	10.5	1.00
Ref. 145	Field	Maize, site C	14.7	75-75-75	15	1.43
Steiner et al. 2007 (Ref 146)	Field	Rice	0	30-35-50-2100(lime)	--[a]	1.00
Ref 146	Field	Rice	7.8	30-35-50-2100(lime)	--[a]	1.52
Baronti et al. 2008 (Ref 140)	Field	Durum Wheat	0	?	8.11	1.00
Ref 140	Field	Durum Wheat	8	?	9.54	1.18
Ref 181	Field	Maize	0	163-34-105	--[ab]	1.00
Ref 181	Field	Maize	5.8	163-34-105	--[ab]	1.26
Ref 181	Field	Maize	14.6	163-34-105	--[ab]	1.50
Rondon et al. 2007 (Ref. 95)	Greenhouse	Common Bean	0	20-20-0-300(lime)	4.4	1.00
Ref. 95	Greenhouse	Common Bean	37.1	20-20-0-300(lime)	5.5	1.25
Ref. 95	Greenhouse	Common Bean	74.2	20-20-0-300(lime)	6.1	1.39

Ref. 95	Greenhouse	Common Bean	111.2	20-20-0-300(lime)	4.7	1.07
Ref. 95	Greenhouse	Common Bean, no N fixation	0	20-20-0-300(lime)	3.4	1.00
Ref. 95	Greenhouse	Common Bean, no N fixation	37.1	20-20-0-300(lime)	3.8	1.12
Ref. 95	Greenhouse	Common Bean, no N fixation	74.2	20-20-0-300(lime)	3.75	1.10
Ref. 95	Greenhouse	Common Bean, no N fixation	111.2	20-20-0-300(lime)	2.7	0.79
Lehmann et al. 2003 (Ref 94)	Greenhouse	Cowpea	0	0	0.73	1.00
Ref 94	Greenhouse	Cowpea	47.9	0	1.05	1.44
Ref 94	Greenhouse	Cowpea	95.7	0	1.34	1.84
Ref 94	Greenhouse	Cowpea	0	0-59-0	0.7	1.00
Ref 94	Greenhouse	Cowpea	47.9	0-59-0	0.89	1.27
Ref 94	Greenhouse	Cowpea	95.7	0-59-0	1.15	1.64
Ref 140	Greenhouse	Perennial Ryegrass	0	?	0.31	1.00
Ref 140	Greenhouse	Perennial Ryegrass	7	?	0.33	1.06
Ref 140	Greenhouse	Perennial Ryegrass	21	?	0.38	1.23
Ref 140	Greenhouse	Perennial Ryegrass	42	?	0.62	2.00
Ref 140	Greenhouse	Perennial Ryegrass	70	?	0.28	0.90
Ref 140	Greenhouse	Perennial Ryegrass	84	?	0.22	0.71

[a]Based on relative yields given by authors

[b]Grain yields

Table S11: Relative biomass yields (RBYs) per Mg C ha^{-1} biochar application (derived from dataset given in Table S10)

Crop	Field	Greenhouse	All
	---------- (ha Mg^{-1} C) ----------		
Cereals	0.028	0.024	0.022
Legumes	0.0048	0.0066	0.0066
Cowpea	0.0066	0.0077	--

Table S12: Mass of crop residues (Pg DM) produced on soils of different fertility classes

Soil fertility constraints	Cereals	Sugar Cane	Oil Crops	Pulses
None	0.69	0.03	0.14	0.008
Few	0.68	0.05	0.08	0.009
Slight	0.38	0.04	0.06	0.006
Moderate	0.45	0.1	0.14	0.007
Severe	0.22	0.03	0.07	0.004
Very Severe	0.14	0.02	0.06	0.002
Unsuitable	0.10	0.03	0.04	0.001

Table S13: Increase in soil cation exchange capacity (CEC) as a function of soil fertility class

Level of soil fertility constraint	Relative biochar induced increase in soil CEC
None	1
Few	1.2
Slight	1.4
Moderate	1.6
Severe	1.8
Very Severe / Unsuitable	2

Table S14: Calculation of current anthropogenic GHG emissions. Global warming potentials (100-yr), anthropogenic emissions, and calculated emissions in Pg CO_2-C_e yr^{-1} for CO_2, CH_4, and N_2O.

	100 yr GWP[a]	Pg yr^{-1}	Pg CO_2-C_e yr^{-1}	Year	Source
CO_2 from Land-Use Changes	1		1.467	2005	Ref. 188
Fossil-Fuel CO_2 Emissions	1		8.23	2006	Ref. 189 Boden et al. (2009)
Anthropogenic CH_4 Emissions (as CH_4)	25	0.582	3.968	2000-2004	Ref. 190,Table 7.6 p. 542
Anthropogenic N_2O Emissions (as N)	298	0.00670	1.711	1990s	Ref. 190,Table 7.7 p. 546
Total for CO_2, CH_4, and N_2O Only			**15.38**		

[a] GWP= Global Warming Potential from Table 2.14 in Ref. 191

Table S15: Estimated capital costs of biomass pyrolysis with power generation

Source	Capital Intensity (US$ Mg^{-1} yr)	Notes
Bridgwater[182]	217-617	Capital intensity related to plant size. $217 relates to 200 kg hr^{-1} (dry feedstock), $617 to 4 Mg hr^{-1}. Intermediate plant of 1 Mg hr-1 has capital intensity of $370. All costs have been converted from yr 2000 € to 2007 US$ using a factor of 1.08.
McCarl et. al.[164]	270	
Islam & Ani[183]	159 – 233	
Roberts[184]	222	Refers to cost of Coaltec plant reported in article plus an additional $110 Mg^{-1} yr for power generation from Ref 164.
Crucible Carbon[185]	173 – 422	
German Energy Agency[186]	395	Based on 10^6 Mg yr^{-1} fast pyrolysis plant, including all ancillary plant, site, planning and contingency. (This figure excludes the Fischer-Tropsch gas to liquid conversion stage also costed in the report).
Joseph[187]	425	Biochar cooking stoves for use in the developing world.

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