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Competing uses for China's straw: the economic and carbon abatement potential of biochar

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Abstract

China is under pressure to improve its agricultural productivity to keep up with the demands of a growing population with increasingly resource-intensive diets. This productivity improvement must occur against a backdrop of carbon intensity reduction targets, and a highly fragmented, nutrient-inefficient farming system. Moreover, the Chinese government increasingly recognizes the need to rationalize the management of the 800 million tonnes of agricultural crop straw that China produces each year, up to 40% of which is burned in-field as a waste. Biochar produced from these residues and applied to land could contribute to China's agricultural productivity, resource use efficiency and carbon reduction goals. However competing uses for China's straw residues are rapidly emerging, particularly from bioenergy generation. Therefore it is important to understand the relative economic viability and carbon abatement potential of directing agricultural residues to biochar rather than bioenergy. Using cost-benefit analysis (CBA) and life-cycle analysis (LCA), this paper therefore compares the economic viability and carbon abatement potential of biochar production via pyrolysis, with that of bioenergy production via briquetting and gasification. Straw reincorporation and in-field straw burning are used as baseline scenarios. We find that briquetting straw for heat energy is the most cost-effective carbon abatement technology, requiring a subsidy of \$7 $MgCO_2e^{-1}$ abated. However China's current bioelectricity subsidy scheme makes gasification (NPV \$12.6 million) more financially attractive for investors than both briquetting (NPV \$7.34 million), and pyrolysis (\$−1.84 million). The direct carbon abatement potential of pyrolysis (1.06 $MgCO_2e$ per odt straw) is also lower than that of briquetting (1.35 $MgCO_2e$ per odt straw) and gasification (1.16 $MgCO_2e$ per odt straw). However indirect carbon abatement processes arising from biochar application could significantly improve the carbon abatement potential of the pyrolysis scenario. Likewise, increasing the agronomic value of biochar is essential for the pyrolysis scenario to compete as an economically viable, cost-effective mitigation technology.

Keywords: biochar, bioenergy, biomass, briquetting, China, gasification, pyrolysis

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Introduction

In the next two decades, China must increase gross agricultural productivity by an estimated 30–50% to keep pace with a growing population and their progressively resource intensive diets (Zhang *et al.*, 2011). Moreover, it must achieve this on arable land that is diminishing in size and fertility due to industrial-contamination of soils (Chen, 2007) and which suffers from low soil organic matter levels (Pan, 2008; Fan *et al.*, 2012). Additionally China needs to tackle the current widespread overuse of chemical fertilizers and pesticides, which is leading to significant eutrophication of water bodies (Zhang *et al.*, 2013a), alongside substantial air pollution and associated climate change from anthropogenic emissions of reactive nitrogen (Liu *et al.*, 2013a).

In principal, biochar is a technology that may be able to address many of these challenges. Biochar is the charred by-product of biomass pyrolysis, which is the heating of plant-derived material in the absence of oxygen (Sohi *et al.*, 2009). The pyrolysis process also produces combustible gases (predominantly H_2, CO, CH_4) that can be captured and used for energy (Brown, 2009).

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The biochar product has a porous latticed structure, formed from stable aromatic rings of carbon that are more resistant to decomposition than the biomass from which they were initially created. Evidence suggests that fractions of this initial biochar product may stay stable for hundreds (Haberstroh *et al*., 2006) or even thousands (Masiello, 1998; Lehmann *et al*., 2008) of years, inferring potential for biochar as a carbon sequestration and climate mitigation tool. Indeed, some studies even suggest that the conversion of available biomass to biochar could reduce annual net global emissions of carbon dioxide, methane and nitrous oxide by 12%, without endangering food security, habitat or soil conservation (Woolf *et al*., 2010).

In addition to this global warming mitigation potential, biochar also has positive agronomic impacts when applied to agricultural soils, specifically by increasing soil organic carbon (SOC) levels (Kimetu *et al*., 2008; Zimmerman *et al*., 2011); stimulating higher crop productivity or maintaining yields with lower input costs (Biederman & Harpole, 2013; Crane-Droesch *et al*., 2013; Liu *et al*., 2013b); improving fertilizer-use efficiency (Steiner *et al*., 2008; Chan & Xu, 2009; Van Zwieten *et al*., 2010); and/or remediating contaminated soils (Beesley *et al*., 2011; Bian *et al*., 2013; Houben *et al*., 2013). Moreover, China appears to have soils upon which biochar's impact on crop yields may be most significant, as demonstrated in a recent global meta-analysis of biochar studies (Crane-Droesch *et al*., 2013); research on the decline of SOC in China's soils, particularly on non-paddy land (Lal, 2002; Tang *et al*., 2006); and many China-based agronomic trials (Zhang *et al*., 2010; Bian *et al*., 2013; Lashari *et al*., 2013).

Additionally, existing biochar systems analyses report strong economic and environmental preferences for the use of waste biomass materials as biochar feedstocks, rather than using wood or other virgin biomass (Roberts *et al*., 2010a; Shackley *et al*., 2011). China demonstrates significant potential in this regard, producing an annual 800 million tonnes of agricultural straw residues, of which an estimated 505 million tonnes are available after retaining sufficient straw to maintain soil quality (Jiang *et al*., 2012). Moreover, many studies report that high proportions of straw are burned in field. For example, Wu *et al*. (2001) report that 33% of crop straw was burned in Jiangsu province, compared to 32.4% for Guangdong province (Lin & Song, 2002), 40% for Fuzhou city (Yu, 2003), and 39.6% for Shanghai (Yao *et al*., 2001). This is a consequence of low mechanization rates (Tang *et al*., 2006) and farmer demographic characteristics, (Cao *et al*., 2006) with farmers of greater income tending to burn more straw because of reduced demand for straw as a household fuel, and a scarcity of on-farm labour for straw collection. This in-field straw burning emits high levels of particulate matter (PM), hydrocarbons and other pollutant gases to the atmosphere, resulting in significant local and regional air quality deterioration (Duan *et al*., 2004; Yan *et al*., 2006).

However, despite currently being plentiful, these straw residues are increasingly in demand as a result of China's bioelectricity subsidies. Recognizing the adverse environmental and health consequences of in-field straw burning, the Chinese government is providing financial incentives to promote the mechanized collection and conversion of straw to electrical energy that is fed into the national grid. The financial incentives offered are structured as a feed-in-tariff ($0.12 kWh^{-1} produced from agricultural and waste forestry biomass), subsidized loans, tax breaks and/or grants (Zhang *et al*., 2014). The feed-in-tariff rate is comparable to western bioenergy policies, (for example, UK energy companies can typically sell renewably-generated electricity for between $0.08 and 0.25 kWh^{-1}), however opinion is divided on whether these incentives are sufficient to create economically viable bioenergy projects (Lu & Zhang, 2010a; Zhang *et al*., 2013b, 2014).

In addition the extent to which these bioenergy subsidies might affect the economic viability of biochar projects is unknown. This therefore raises questions about how the agronomic results of biochar field trials translate into the development of biochar as a commercial product, and additionally whether commercial biochar projects can contribute to GHG emission reductions in China.

We therefore investigate and contrast the economics and carbon abatement potential of using China's straw resources for biochar production via pyrolysis with two bioenergy technologies: straw briquetting and straw gasification. These scenarios are compared against two reference cases (straw reincorporation and in-field straw burning) and are analyzed in terms of their relative profitability from a business perspective, and in terms of their environmental benefits from a global GHG balance perspective.

Materials and methods

Cost-benefit analysis (CBA) is used to compare the economic viability [net present value (NPV) per oven dry tonne (odt) straw], and life cycle analysis (LCA) is used to compare the environmental ($MgCO_2e$ per odt straw) outcomes associated with three straw utilization scenarios: straw briquetting and subsequent combustion for heat energy (S_{Briq}); straw gasification for electrical energy (S_{Gas}); and straw pyrolysis for biochar and electrical energy (S_{Pyr}). These are compared to two baselines of straw reincorporation (S_{Rein}) and straw burning (S_{Burn}). S_{Rein} assumes that all straw is incorporated into the field whereas S_{Burn} assumes that straw is burned in-field.

Technology scenario selection

Straw briquetting (S_{Briq}) was chosen as a comparison scenario based on observations of straw briquettes on sale in Chinese town markets and online. Briquetting has much lower capital and technological expertise requirements than gasification and pyrolysis, and is therefore likely to be perceived as lower risk by investors and as an accessible option for small businesses. However it does not qualify for government bioelectricity subsidies, as briquettes tend to be bought for local heat and cooking applications rather than burned for commercial electricity generation. In contrast, straw gasification (S_{Gas}) was chosen on the basis that gasification is identified as a priority bioenergy technology in Chinese national policy documents (Han *et al.*, 2008; Zhang *et al.*, 2014), has been implemented in many technological development projects across China (Kirkels & Verbong, 2011), and is reportedly a viable economic proposition for Chinese businesses (Lu & Zhang, 2010a). Although co-firing with coal has also been found to be an economic use of straw residues, (Lu & Zhang, 2010a), it was not included as an option because the Chinese government does not currently provide financial incentives for bioelectricity produced through co-firing. This is due to concerns over the accurate verification of biomass co-firing rates at existing coal-fired power stations (Gan & Yu, 2008; Dong, 2012).

The pyrolysis (S_{Pyr}) scenario investigates the use of slow pyrolysis technology to produce biochar and a relatively small amount of electricity. Slow pyrolysis always delivers less electricity than other bioenergy options, because a proportion of the feedstock is converted to biochar and not into heat or electrical energy (Brown, 2009).

Each of the S_{Briq}, S_{Gas} and S_{Pyr} technology scenarios is guided by interviews conducted in summer 2012 at the Sanli New Energy bioenergy-plant in Henan Province, China. Sanli New Energy has capitalized upon the combination of a local straw-burning ban, related straw-burning avoidance subsidies (\$28 Mg^{-1} straw paid to businesses that use straw for livestock rearing, paper production or bioenergy generation) and national bioelectricity subsidies (Zhang *et al.*, 2014), to build a 4 MW pyrolysis unit and straw briquetting plant. Data on Sanli's economics, straw collection system and size guided the choice of parameters used to structure and assess the S_{Briq}, S_{Gas} and S_{Pyr} scenarios. Table 1 provides an overview of these parameters. More detailed information on technology configuration is available in the Data S1 (S9–S17, and Figures S1 and S2).

The Technology Readiness Levels (TRL) for each technology (straw briquetting, gasification and pyrolysis) are also estimated, based on expert opinion and observations of the deployment of these technologies in rural Chinese settings. A TRL is a scale from one to nine that indicates the maturity of a given technology (Mankins, 1995; UK Ministry of Defence, 2014). Table S1 provides a description for each TRL. Briquetting scores the highest (9), as a mature 'off the peg' technology, followed by gasification at stages 7–8, and then pyrolysis at stages 5–6.

Cost benefit analysis

Published literature, industry reports, policy documents, interviews and online market estimates were used to develop appropriate pricing structures for S_{Briq}, S_{Gas} and S_{Pyr}, adjusted to 2014 prices. The CBA combines these values to generate an estimate of scenario profitability from the perspective of a business or potential investor, taking account of government bioelectricity and avoided straw burning subsidies.

The agronomic value for biochar is estimated by combining data on the microeconomics of farms in Henan (Clare *et al.*, 2014) with data from the latest published meta-analyses on biochar's yield impacts (Jeffery *et al.*, 2011; Crane-Droesch *et al.*, 2013), the findings from which are also consistent with results from China-based biochar experiments (Wang *et al.*, 2012; Zhang *et al.*, 2012a). Biochar's agronomic value is calculated as the value of the yield improvement seen in one growing year, per unit of biochar applied, assuming that biochar is applied once and that its effects last across two growing seasons. It should be noted that this estimate does not take spreading and transportation costs into account, and that therefore the commercial sale price of biochar to farmers will need to be less

Table 1 Overview of technical parameters for briquetting, gasification and pyrolysis

	Briquetting	Gasification	Pyrolysis
Technology Readiness Level (TRL)	9	7–8	5–6
Lifetime of operation (yrs)	20	20	20
Straw processed (odt yr^{-1})	28 000	28 000	28 000
Annual output	28 000 Mg briquettes	26 680 MWh bioelectricity	8400 MWh bioelectricity; 8300 Mg biochar
Energy offset	Equivalent MJ heat energy from coal briquettes	Equivalent MWh electrical energy from central China's grid	Equivalent MWh electrical energy from central China's grid
National bioelectricity subsidies	None	Feed-in-tariff for bioelectricity (\$0.12 kWh^{-1}); subsidized capital loans; tax breaks (Zhang *et al.*, 2014);	Feed-in-tariff for bioelectricity (\$0.12 kWh^{-1}); subsidized capital loans; tax breaks (Zhang *et al.*, 2014);
Local straw-burning subsidies	Avoided straw burning (\$28 Mg^{-1})	Avoided straw burning (\$28 Mg^{-1})	Avoided straw burning (\$28 Mg^{-1})

than this figure. The baseline agronomic value for biochar of \$110 Mg^{-1} is calculated according to the latest meta-analysis by Crane-Droesch *et al.* (2013), who report a 10% yield increase for a 3 Mg ha^{-1} application rate. However, the more conservative estimate of Jeffery *et al.* (2011), assuming that a 10 Mg ha^{-1} application stimulates 10% yield increases, gives biochar an agronomic value of just \$33 Mg^{-1}. This is a significant price difference, and therefore the retail price of biochar is varied in the sensitivity analysis, reflecting this uncertainty and investigating the extent to which it impacts the overall profitability of S_{Pyr}.

Briquette market value is calculated based on the typical energy density of straw briquettes (McKendry, 2002; Roberts *et al.*, 2010b) and the value of this energy based on the spot price of coal in China at the time of writing (\$95 Mg^{-1}; Zhao & Che, 2012; Yang, 2014). Finally, the market value of bioelectricity is set in line with the current Chinese bioelectricity subsidy of \$0.12 kWh^{-1} (Zhang *et al.*, 2014).

The NPV of each scenario is calculated at the project level, over a 20 year lifetime, taking subsidized loans and tax breaks into account where relevant (Zhang *et al.*, 2014). The discount rate is set at 3.5% (Federal Reserve Bank of St Louis, 2014).

Life-cycle analysis

A GHG-oriented attributional LCA was performed, based on the ISO 14040 (2006) guidelines, and using a 100 year global warming potential. The three main GHGs were accounted for [carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O)], and these are henceforth displayed in terms of their carbon dioxide equivalent global warming potential (CO_2e), calculated according to IPCC guidelines of CO_2e equivalence as 25 for CH_4 and 298 for N_2O (IPCC, 2007). The GHG abatement potentials of S_{Briq}, S_{Gas} and S_{Pyr} were calculated using S_{Rein} as the baseline scenario, however the S_{Burn} scenario is also displayed for reference. The analysis initially focuses on directly-attributable CO_2e emissions from each phase of the life cycle (raw material acquisition, production, distribution, energy offset and dismantling processes) before moving on to consider the indirect CO_2e abatement potential of reduced soil N_2O emissions and avoided fertilizer use as a result of biochar application.

Soil N_2O reductions following biochar application have been widely debated for some years, however a recent meta-analysis (Cayuela *et al.*, 2014) provides greater clarity on the extent of this effect. Cayuela *et al.*, report that biochars derived from woody and herbaceous feedstocks, including agricultural straws, demonstrate the highest emission reduction potential, with a 27% reduction in N_2O emissions for a 1–2% (by soil weight) biochar application rate. Data from this study is then combined with a China-specific field trial demonstrating a similar effect (Zhang *et al.*, 2012b) to calculate the additional contribution that N_2O emission reduction may have on the S_{Pyr} LCA result.

A similar approach is taken to calculating additional GHG abatement as a result of avoided fertilizer application. Recent trials in China suggest that the application of a combined biochar-NPK-clay compound [a biochar-mineral-chemical-composite (BMCC)] may be an economic option for farmers, where ~25% of NPK is replaced by biochar, on a weight basis (Joseph *et al.*, 2013). This data is combined with data on the carbon intensity of China's domestic fertilizer production industry, which emits 13.5 $MgCO_2e$ MgN^{-1} fertilizer as compared to an average of 9.7 $MgCO_2e$ MgN^{-1} in Europe (Zhang *et al.*, 2013c). The nitrogen (N) fertilizer is assumed to contribute to a standard NPK (16:16:16) mix. Emissions from potassium (K) and phosphorus (P) production in synthetic fertilizers are excluded, as they are an order of magnitude lower (West & Marland, 2002). Figure 1 displays the processes included in the direct and indirect abatement potential calculations.

The CO_2e offsets from avoided fossil fuel energy are calculated according to the carbon emission factor (CEF) of the fuel that straw-derived bioenergy is expected to replace. Straw briquettes are assumed to replace coal briquettes that are typically burned for heat and/or cooking purposes in local applications such as homes, schools and hospitals. In S_{Gas} and S_{Pyr}, each MWh of bioelectricity produced is assumed to replace one MWh of electricity in the central grid, which services Henan province and has an estimated carbon intensity of 1.133 $MgCO_2e$ MWh^{-1} (World Resources Institute, 2014).

The details of GHG emissions associated with different phases of the lifecycle are given in the supplementary material (S9–S17). Many of the parameters used to estimate these emissions are considered uncertain, therefore published literature and expert opinion were also used to estimate the uncertainty range and probability distribution of each parameter. An uncertainty analysis was then undertaken using a Monte Carlo method. 10 000 simulations were performed to derive median points and 95% confidence intervals for $MgCO_2e$ emitted per odt feedstock. The impact of each parameter's value on the final result was investigated using sensitivity analysis.

Results

Economic viability of briquetting, gasification and pyrolysis

Removing both national bioelectricity and local avoided straw-burning subsidies renders S_{Briq}, S_{Gas} and S_{Pyr} unprofitable, with project NPVs of \$−2.88 million (m), \$−19.0 m, and \$−20.3 m, respectively (see black bars in Fig. 2). When including local avoided straw burning subsidies (see grey bars in Fig. 2), S_{Briq} becomes profitable (NPV \$7.34 m), whereas S_{Gas} and S_{Pyr} still generate significant losses (NPV \$−8.14 m and \$−9.36 m respectively). However, the inclusion of income from China's national bioelectricity subsidy programme (see white bars in Fig. 2) has a significant impact on S_{Gas} profitability (NPV \$12.60 m), increasing it above the unchanged S_{Briq} NPV (\$7.34 m). Meanwhile, S_{Pyr} remains unprofitable (NPV \$−1.84 m), due to the relatively lower electricity volume yielded per odt straw by pyrolysis as compared with gasification.

However the NPV of S_{Pyr} is strongly influenced by the agronomic value of biochar, which is one of the

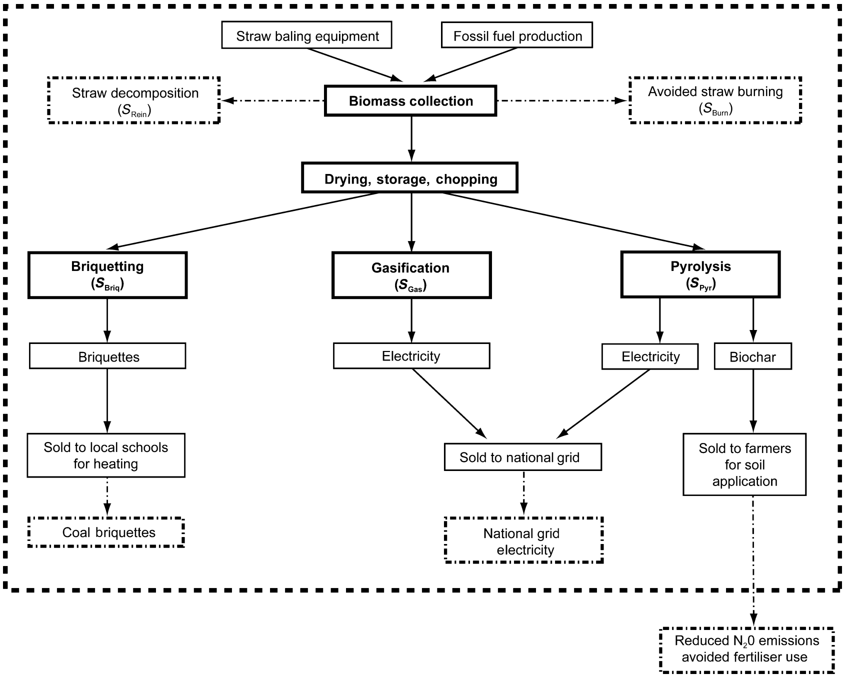

Fig. 1 Diagram of LCA boundaries: bold boxes indicate processes that emit CO_2e, dashed boxes indicate CO_2e offset or abatement processes. Processes within the bold dashed line are considered direct impacts of each scenario, and processes outside the bold dashed line are considered indirect impacts.

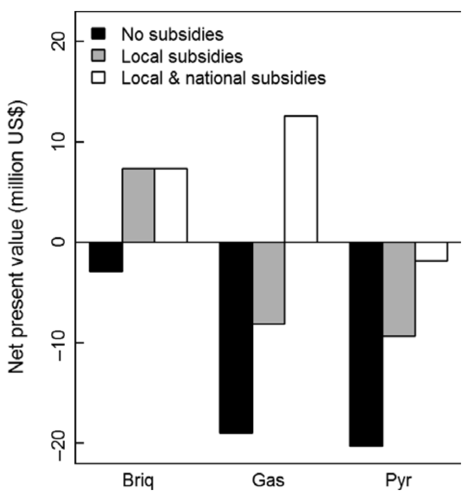

Fig. 2 Net present value (million US$), with and without Chinese government subsidies, for S_{Briq}, S_{Gas} and S_{Pyr}.

most uncertain parameters modelled in this CBA. At the baseline agronomic value of \$110 Mg^{-1}, (based on the results of Crane-Droesch *et al.* (2013)) the S_{Pyr} NPV (including all available subsidies) is \$−1.84 m. However, assuming the more conservative estimate of \$33 Mg^{-1}, (based on the results of Jeffery *et al.* (2011)) the S_{Pyr} NPV drops even further to \$−10.1 m. For S_{Pyr} to break even, biochar must sell for \$128 Mg^{-1} if all other factors remain equal, or for \$206 Mg^{-1}, if bioenergy subsidies are excluded. For the NPV of S_{Pyr} to equal that of S_{Gas}, biochar must sell for \$238 Mg^{-1}. Interestingly, in 2014 Sanli New Energy Company reported their biochar retail price as \$259 Mg^{-1}, which exceeds the break-even prices that we report as being necessary for pyrolysis profitability. However, this high sale price is at odds with current understanding of biochar's agronomic value in soil (as outlined above) and studies on agricultural economics and farmer-perspectives of biochar in the area (Clare *et al.*, 2014).

Direct CO_2e abatement potential of briquetting, gasification and pyrolysis

Figure 3 outlines the CO_2e abatement potential of S_{Burn}, S_{Briq}, S_{Gas} and S_{Pyr}, including only direct processes in the analysis, all implicitly compared against S_{Rein} as the baseline scenario. The results suggest that, when including offsets from avoided fossil-fuel energy emissions (see black bars in Fig. 3), S_{Briq} offers the greatest carbon abatement (1.35 $MgCO_2e$ per odt straw) followed by S_{Gas} (1.16 $MgCO_2e$ per odt straw) and S_{Pyr} (1.06 $MgCO_2e$ per odt straw). This carbon abatement potential increases by 0.04 $MgCO_2e$ per odt straw for each scenario, if referenced to the S_{Burn} baseline rather than S_{Rein}. Interestingly this means that, despite only

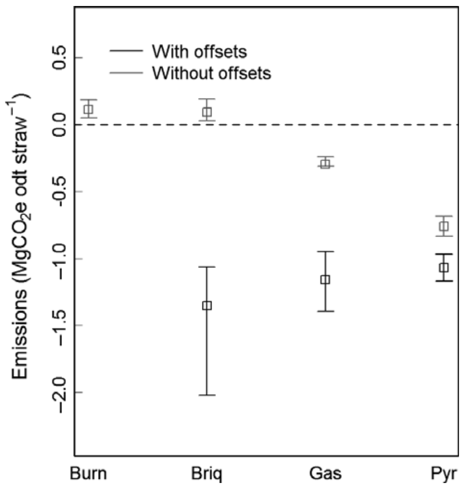

Fig. 3 Median and confidence interval estimates of $MgCO_2e$ abated per odt straw processed in S_{Briq}, S_{Gas} and S_{Pyr}, including and excluding offsets from avoided fossil-fuel energy (black bars and grey bars, respectively). Uses S_{Rein} as the baseline, and displays S_{Burn} for reference.

receiving local and not national subsidies, S_{Briq} appears to offer the greatest CO_2e abatement potential. However, S_{Briq} also displays the most variance in its carbon abatement, as a result of the wide variability in data available for comparing emissions from straw and coal briquettes in small stoves (Zhang *et al.*, 2000; Wang *et al.*, 2013).

If emissions offsets from avoided fossil fuel use are not included (see grey bars in Fig. 3), both S_{Gas} and S_{Pyr} still provide some carbon abatement. In the case of S_{Gas} this is because approximately 20% of feedstock carbon is initially stabilized in the ashy char produced during the gasification process (Lu & Zhang, 2010b) with 90% remaining stable over the 100 year time-scale of this analysis (Cross & Sohi, 2013). In the case of S_{Pyr}, 50% of feedstock carbon is initially stabilized in the biochar, with 80% of that amount (39% of the initial feedstock carbon) still remaining in the soil after 100 years (Singh *et al.*, 2012; Crombie *et al.*, 2013). This persistence is a pertinent point, as it can be argued that offset fossil fuel emissions are not avoided for long, because the fossil fuel still remains to be consumed. From these perspectives, it can therefore be argued that S_{Pyr} offers a more permanent GHG reduction than the other options.

Indirect CO2e abatement potential of pyrolysis

The application of biochar to agricultural land may contribute to the abatement potential of S_{Pyr} via indirect processes, which generally have a higher level of uncertainty and variability than the direct factors already discussed. This can result from reduced certainty regarding biochar's impact on a given outcome (i.e., in the case of biochar's effect on N_2O emissions) and/or because the process relies on human behaviour change (i.e., the reduction in fertilizer application, or the application of biochar to land). Indirect environmental consequences of biochar application have been variously reported in past LCA studies (Roberts *et al.*, 2010a; Hammond *et al.*, 2011; Sparrevik *et al.*, 2013), but recent evidence has improved the evidence base for the effect magnitude that might be expected for a given biochar application rate. Specifically, two indirect effects that have received increased attention are reduced N_2O emissions from soil and improved fertilizer use efficiency.

Reduced N_2O emissions from soil. Table 2 combines data from a recent meta-analysis of biochar's impact on soil N_2O emissions (Cayuela *et al.*, 2014) with the baseline and reduced N_2O emission reductions reported in a China-based biochar field trial (Zhang *et al.*, 2012a). According to these data, and assuming a one-year effect of biochar on N_2O emissions, the abatement potential of S_{Pyr} could be increased by 0.004–0.012 $MgCO_2e\ yr^{-1}$. This represents a 1% increase in S_{Pyr}'s abatement potential, and we therefore suggest that the absolute contribution of biochar-induced soil N_2O emission reductions is relatively small.

Improved fertilizer use efficiency. If biochar were to aid the reduction of fertilizer application in China, the resulting GHG mitigation potential is large. Using data from Joseph *et al.* (2013) and Zhang *et al.* (2013c) we calculate that each Mg of biochar that replaces chemical fertilizer could abate an additional 1.33 $MgCO_2e$, and thus that each odt of straw feedstock being used to produce biochar could abate an additional 0.39 $MgCO_2e$.

Including these indirect effects of biochar application on avoided emissions from soil N_2O and fertilizer use reduction, the total abatement potential of S_{Pyr} increases to 1.46 $MgCO_2e$ per odt straw, which puts it ahead of

Table 2 Calculations of avoided N_2O emissions per tonne feedstock pyrolysed

Biochar application rate (%)	0.5*	2*	1–2†
% N_2O reduction from baseline	−40	−51	−27
N_2O avoided (kg per odt)	0.021	0.007	0.007
Abatement potential ($MgCO_2e$ per odt)	0.012	0.004	0.004

*Data taken from Zhang *et al.* (2012a).
†Data taken from Cayuela *et al.* (2014).

both S_{Gas} (1.16 $MgCO_2e$ per odt straw) and S_{Briq} (1.35 $MgCO_2e$ per odt straw) in terms of carbon abatement.

Sensitivity analysis

Figures 4 and 5 graphically display the results of sensitivity analysis undertaken on key parameters influencing the NPV and carbon abatement potential, respectively, of the S_{Briq}, S_{Gas} and S_{Pyr} scenarios. Both figures present the baseline NPV/carbon abatement value and a surrounding range, calculated by varying key economic/carbon abatement parameters by ±20%, whilst keeping all other parameter values constant. The parameter values used in these sensitivity analyses are available in S19 and S20 of Data S1.

Figure 4 displays the influence of the following economic parameters on the overall NPV for each scenario: straw price, local straw burning subsidies, capital cost, labour cost, and the sale price of outputs (briquettes; electricity; electricity and biochar, for S_{Briq}, S_{Gas} and S_{Pyr}, respectively). All NPVs displayed include the financial support currently available from both local and national subsidy programmes.

The results in Fig. 4 suggest that sales prices for output products are very influential on the overall economic viability of briquetting, gasification and pyrolysis projects. Likewise, varying the capital cost of pyrolysis and gasification units has a significant impact on the economic viability of S_{Gas} and S_{Pyr}, even tipping S_{Pyr} into profitability where capital costs alone decrease by 20%. This is particularly relevant when considering the early stage of technological readiness of pyrolysis and the subsequent drop in capital cost that might be expected as this technology reaches higher stages of

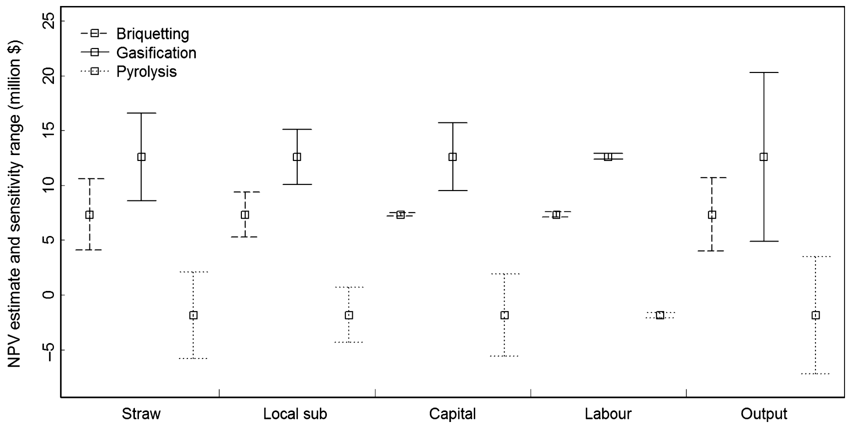

Fig. 4 Baseline NPV estimates (million US$) and sensitivity analyses for key parameters determining the economic viability of briquetting, gasification and pyrolysis. Ranges are produced by independently varying key parameters (*x*-axis) by ±20% and recording the impact on the overall NPV value.

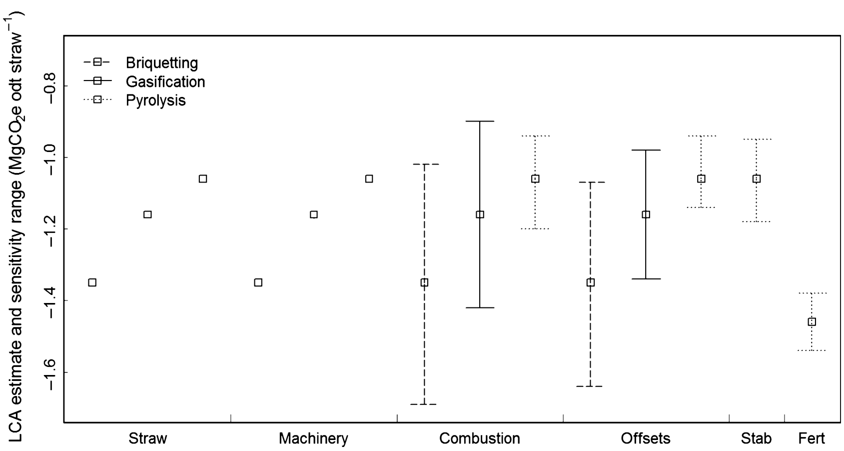

Fig. 5 Baseline carbon abatement estimates ($MgCO_2e$ abated odt per straw) and sensitivity analyse for key parameters determining the carbon abatement potential of briquetting, gasification and pyrolysis. Ranges are produced by independently varying key parameters (*x*-axis) by ±20% and recording the impact on overall carbon abatement potential.

maturity (Utterback, 1996; Shackley *et al.*, in press). However, it must also be noted that the top range of S_{Pyr}'s NPVs do not overlap with the bottom range of the NPVs of S_{Briq} or S_{Gas}, suggesting that pyrolysis will require significant improvements in multiple economic parameters before it can compete with briquetting or gasification.

Figure 5 displays the results of a ±20% sensitivity analysis conducted on the following key parameters influencing the carbon abatement potential of S_{Briq}, S_{Gas} and S_{Pyr}: straw collection emissions; embedded emissions within machinery; direct emissions from the combustion of straw briquettes/gasification of straw/pyrolysis of straw; offset emissions from avoided fossil fuel energy; the stability of carbon sequestered within biochar; and offset emissions from avoided fertilizer use. The results suggest that direct emissions from combustion of straw briquettes/gasification of straw/pyrolysis of straw, and offset emissions from avoided fossil fuel use, have the greatest impact on the carbon abatement potential of each scenario. This suggests that gasification and pyrolysis units must be well designed, maintained and managed by staff with appropriate expertise, and that improvements to the efficiency of boilers that combust straw briquettes could also improve their carbon abatement potential. Variation in emissions from straw collection and machinery/building construction has a negligible impact on overall carbon abatement balance. However, variability in fertilizer use and the stability of carbon sequestered within biochar have modest effects on the overall carbon abatement potential of S_{Pyr}.

Carbon abatement cost-effectiveness

In light of the Chinese government's carbon intensity reduction targets, it is important to consider the cost-effectiveness of S_{Briq}, S_{Gas}, and S_{Pyr} in terms of CO_2e abatement. Our results show that all three technologies require assistance from carbon pricing to break-even, although S_{Briq} requires a significantly lower price than S_{Gas} and S_{Pyr} (see Table 3, where S_{Pyr} (D) includes only direct effects and S_{Pyr} (I) includes both direct and indirect processes discussed in this paper). Requiring a carbon price of \$7 $MgCO_2e$ per abated, S_{Briq} is the only technology studied here that can produce carbon abatement for less than \$25 $MgCO_2e^{-1}$, as outlined in the Stern Report (Stern, 2006). Moreover, early price indications from China's nascent emissions trading scheme (which currently covers five municipal areas and two provinces; (Lo, 2012) suggest that domestic carbon prices (currently ranging between \$5 and \$20 $MgCO_2$ per abated) would only provide sufficient support to make S_{Briq} profitable (Song & Lei, 2014).

Table 3 Comparing CO_2e abatement cost effectiveness for briquetting, gasification and pyrolysis

	S_{Briq}	S_{Gas}	S_{Pyr} (D)	S_{Pyr} (I)
Subsidy required (\$ tonne feedstock per processed)	5	34	36	36
Subsidy required (\$ $MgCO_2e^{-1}$ abated)	7	61	71	51

Discussion

We find that the briquetting of straw for sale as a local fuel in heating and cooking appliances to be the most efficient use of China's straw residue resources. S_{Briq} has the greatest carbon abatement potential (1.35 $MgCO_2e$ per odt straw as compared to 1.16 and 1.06 $MgCO_2e$ per odt for S_{Gas} and S_{Pyr}, respectively), and the highest economic abatement efficiency (requiring a relatively small carbon price of \$7 $MgCO_2e^{-1}$ abated, compared to \$61 $MgCO_2e^{-1}$ or \$51–71 $MgCO_2e^{-1}$ abated, for S_{Gas} and S_{Pyr}, respectively.) Straw briquetting also has the highest technology readiness level (TRL), making it attractive for small businesses and village level industry. This technology also leads to the direct use of biomass energy for heat in boilers and heating systems of local communities, thus negating the need for expensive equipment and avoiding the inevitable energy wastage when converting heat energy into electricity.

However, the apparent success of straw briquetting is subject to two important caveats. Firstly, this scenario relies on the sale of straw briquettes to local households, schools and hospitals for combustion in relatively inefficient, small-scale boilers and stoves. However, as China's energy system modernizes, there may be a move towards more efficient district heating and power systems, which will reduce market demand for straw briquettes to be processed and sold in this way. Secondly, the heat energy produced from locally sold briquettes is not as fungible as electricity, which is socially a highly valued commodity.

This may explain why current Chinese bioenergy subsidies focus on bioelectricity generation, and supports our finding that national bioelectricity subsidies increase the NPV of gasification (NPV \$12.60 m) above that of briquetting (NPV \$7.34 m). However, pyrolysis remains unprofitable even when receiving local and national subsidy support (NPV \$−1.84 m). For pyrolysis and associated biochar production to be able to compete with alternative uses of feedstock such as briquetting and gasification, the agronomic value of biochar will need to increase considerably. The current evidence suggests that biochar has an agronomic value of approximately \$110 Mg^{-1} in central, grain-growing Chinese provinces such as Henan (Crane-Droesch *et al*., 2013; Clare *et al*., 2014). However, we find that biochar must sell for at least \$238 Mg^{-1} in the presence of subsidies for the NPV of S_{Pyr} to equal that of S_{Gas}, which is far above what current research suggests is its agronomic value in the first year after application. Moreover, our LCA analysis suggests that pyrolysis is unlikely to attract financial support from the Chinese government on carbon abatement grounds alone, unless the abatement potential of indirect processes such as avoided fertilizer use are included and can be increased.

There are three considerations that may affect these findings. The first relates to the indirect mitigation potential of avoided fertilizer use. In fact, fertilizer application rates in China are so high that fertilizer application can be reduced by up to 27% with no impact on yields, and without requiring biochar application (Huang *et al*., 2008). This calls into question the necessity of biochar to stimulate this particular indirect carbon abatement mechanism because, although replacement of NPK with biochar to produce a biochar-mineral-chemical-composite (BMCC) could theoretically reduce fertilizer application rates (Joseph *et al*., 2013; Clare *et al*., 2014), biochar is not essential to achieving this goal.

Secondly, there are anecdotal reports of two factories in central China producing 60 000Mg yr^{-1} of BMCC products for local agricultural markets. Field trials in China have recently suggested that BMCCs (which premix low application rates of biochar with inorganic fertilizer and clay) can produce yield increases of up to 40% (Joseph *et al*., 2013). Applying this data to agricultural market conditions in Henan province, biochar's value as a soil amendment would be \$5740 Mg^{-1}, increasing the S_{Pyr} NPV to over 50 times that of S_{Gas}. If these results are reproducible, this is a significant game-changer for the field of biochar research and application, however extensive field trials are necessary to ensure that such impacts can be replicated consistently.

Thirdly, the technological advancement, appropriate management and successful deployment of pyrolysis and gasification technologies will have an important impact both on their carbon abatement and economic potential. Improved technological maturity and deployment should improve the conversion efficiency from straw energy to electrical energy and/or biochar. This is a significant determinant of the overall economic viability and emissions balance of S_{Gas} and S_{Pyr}, both by increasing the units of economic output produced per unit of feedstock, and by avoiding emissions of strong climate forcing GHGs resulting from incomplete combustion. Also, the 'technological readiness' of pyrolysis currently lags behind gasification, making it potentially more risky and less attractive for investors. As such, innovative technological advancements are needed for pyrolysis technology to compete with gasification and briquetting, both in terms of economic viability and carbon abatement potential.

Whether Chinese policy makers provide financial support to the advancement of pyrolysis technology is likely to depend on the outcomes of biochar trials related to land remediation [currently a significant issue in China (Chen, 2007; Khan *et al*., 2008; Bian *et al*., 2013)] and to BMCCs for food production. If early BMCC trial results of high yield impacts for low biochar application and reduced fertilizer application rates (Joseph *et al*., 2013) are further substantiated, the Chinese government may consider pyrolysis/biochar technology as something that policy should support, even if this increases competition for straw feedstocks in China.

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References

Beesley L, Moreno-Jiménez E, Gomez-Eyles JL, Harris E, Robinson B, Sizmur T (2011) A review of biochars' potential role in the remediation, revegetation and restoration of contaminated soils. *Environmental Pollution*, **159**, 3269–3282.

Bian R, Chen D, Liu X *et al*. (2013) Biochar soil amendment as a solution to prevent Cd-tainted rice from China: results from a cross-site field experiment. *Ecological Engineering*, **58**, 378–383.

Biederman L, Harpole WS (2013) Biochar and its effects on plant productivity and nutrient cycling: a meta-analysis. *Global Change Biology Bioenergy*, **5**, 202–214.

Brown R (2009) Biochar production technology. In: *Biochar for Environmental Management - Science and Technology* (eds Lehmann J, Joseph S), pp. 127–146. Earthscan, London, UK.

Cao G, Zhang X, Zheng F (2006) Inventory of black carbon and organic carbon emissions from China. *Atmospheric Environment*, **40**, 6516–6527.

Cayuela ML, van Zwieten L, Singh BP, Jeffery S, Roig A, Sánchez-Monedero M (2014) Biochar's role in mitigating soil nitrous oxide emissions: a review and meta-analysis. *Agriculture, Ecosystems & Environment*, **191**, 5–16.

Chan K, Xu Z (2009) Biochar–nutrient properties and their enhancement. In: *Biochar for Environmental Management: Science and Technology* (eds Lehmann J, Joseph S), pp. 67–84. Earthscan, London, UK.

Chen J (2007) Rapid urbanization in China: a real challenge to soil protection and food security. *Catena*, **69**, 1–15.

Clare A, Barnes A, McDonagh J, Shackley S (2014) From rhetoric to reality: farmer perspectives on the economic potential of biochar in China. *International Journal of Agricultural Sustainability*, **12**, 440–458.

Crane-Droesch A, Abiven S, Jeffery S, Torn MS (2013) Heterogeneous global crop yield response to biochar: a meta-regression analysis. *Environmental Research Letters*, **8**, 44–49.

Crombie K, Mašek O, Sohi SP, Brownsort P, Cross A (2013) The effect of pyrolysis conditions on biochar stability as determined by three methods. *Global Change Biology Bioenergy*, **5**, 122–131.

Cross A, Sohi SP (2013) A method for screening the relative long-term stability of biochar. *Global Change Biology Bioenergy*, **5**, 215–220.

Dong N (2012) *Support Mechanisms for Cofiring Secondary Fuels*. IEA Clean Coal Centre, London, UK.

Duan F, Liu X, Yu T, Cachier H (2004) Identification and estimate of biomass burning contribution to the urban aerosol organic carbon concentrations in Beijing. *Atmospheric Environment*, **38**, 1275–1282.

Fan M, Shen J, Yuan L, Jiang R, Chen X, Davies WJ, Zhang F (2012) Improving crop productivity and resource use efficiency to ensure food security and environmental quality in China. *Journal of Experimental Botany*, **63**, 13–24.

Federal Reserve Bank of St Louis (2014) Discount rate for China. Available at: http://research.stlouisfed.org/fred2/series/INTDSRCNM193N (accessed 6 March 2014).

Gan L, Yu J (2008) Bioenergy transition in rural China: policy options and co-benefits. *Energy Policy*, **36**, 531–540.

Haberstroh PR, Brandes JA, Gélinas Y, Dickens AF, Wirick S, Cody G (2006) Chemical composition of the graphitic black carbon fraction in riverine and marine sediments at sub-micron scales using carbon X-ray spectromicroscopy. *Geochimica et Cosmochimica Acta*, **70**, 1483–1494.

Hammond J, Shackley S, Sohi S, Brownsort P (2011) Prospective life cycle carbon abatement for pyrolysis biochar systems in the UK. *Energy Policy*, **39**, 2646–2655.

Han J, Mol APJ, Lu Y, Zhang L (2008) Small-scale bioenergy projects in rural China: lessons to be learnt. *Energy Policy*, **36**, 2154–2162.

Houben D, Evrard L, Sonnet P (2013) Mobility, bioavailability and pH-dependent leaching of cadmium, zinc and lead in a contaminated soil amended with biochar. *Chemosphere*, **92**, 1450–1457.

Huang J, Hu R, Cao J, Rozelle S, Korea S (2008) Training programs and in-the-field guidance to reduce China's overuse of fertilizer without hurting profitability. *Journal of Soil and Water Conservation*, **63**, 163–165.

IPCC (2007) Climate change 2007: working group I: the physical science basis. IPCC Fourth Assessment Report: Climate Change.

ISO (2006) *ISO 14040: Environmental Management—Life Cycle Assessment— Principles and Framework*. International Standards Organisation, Geneva.

Jeffery S, Verheijen FG, van der Velde M, Bastos AC (2011) A quantitative review of the effects of biochar application to soils on crop productivity using meta-analysis. *Agriculture, Ecosystems & Environment*, **144**, 175–187.

Jiang D, Zhuang D, Fu J, Huang Y, Wen K (2012) Bioenergy potential from crop residues in China: availability and distribution. *Renewable and Sustainable Energy Reviews*, **16**, 1377–1382.

Joseph S, Graber E, Chia C *et al.* (2013) Shifting paradigms on biochar: micro/nano-structures and soluble components are responsible for its plant growth promoting ability. *Carbon Management*, **4**, 323–343.

Khan S, Cao Q, Zheng YM, Huang YZ, Zhu YG (2008) Health risks of heavy metals in contaminated soils and food crops irrigated with wastewater in Beijing, China. *Environmental Pollution (Barking, Essex: 1987)*, **152**, 686–692.

Kimetu JM, Lehmann J, Ngoze SO *et al.* (2008) Reversibility of soil productivity decline with organic matter of differing quality along a degradation gradient. *Ecosystems*, **11**, 726–739.

Kirkels AF, Verbong GPJ (2011) Biomass gasification: still promising? A 30-year global overview. *Renewable and Sustainable Energy Reviews*, **15**, 471–481.

Lal R (2002) Soil carbon sequestration in China through agricultural intensification, and restoration of degraded and desertified ecosystems. *Land Degradation & Development*, **13**, 469–478.

Lashari MS, Liu Y, Li L *et al.* (2013) Effects of amendment of biochar-manure compost in conjunction with pyroligneous solution on soil quality and wheat yield of a salt-stressed cropland from Central China Great Plain. *Field Crops Research*, **144**, 113–118.

Lehmann J, Skjemstad J, Sohi S *et al.* (2008) Australian climate–carbon cycle feedback reduced by soil black carbon. *Nature Geoscience*, **1**, 832–835.

Lin RQ, Song DL (2002) Utilizing status and problems of crop straw on Guangdong province (in Chinese). *Soil and Environmental Sciences*, **11**, 110.

Liu X, Zhang Y, Han W *et al.* (2013a) Enhanced nitrogen deposition over China. *Nature*, **494**, 459–462.

Liu X, Zhang A, Ji C *et al.* (2013b) Biochar's effect on crop productivity and the dependence on experimental conditions - a meta-analysis of literature data. *Plant and Soil*, **373**, 583–594.

Lo AY (2012) Carbon emissions trading in China. *Nature Climate Change*, **2**, 765–766.

Lu W, Zhang T (2010a) Life-cycle implications of using crop residues for various energy demands in China. *Environmental Science & Technology*, **44**, 4026–4032.

Lu W, Zhang T (2010b) Supporting Information: life-cycle implications of using crop residues for various energy demands in China. *Environmental Science and Technology*, 1–62.

Mankins JC (1995) *Technology Readiness Levels: A White Paper*. NASA, USA.

Masiello C (1998) Black carbon in deep-sea sediments. *Science*, **280**, 1911–1913.

McKendry P (2002) Energy production from biomass (Part 2): conversion technologies. *Bioresource Technology*, **83**, 47–54.

Pan G (2008) Soil organic carbon stock, dynamics and climate change mitigation of China (in Chinese). *Advances in Climate Change Research*, **5**, 1–7.

Roberts KG, Gloy B, Joseph S, Scott NR, Lehmann J (2010a) Life cycle assessment of biochar systems: estimating the energetic, economic, and climate change potential. *Environmental Science & Technology*, **44**, 827–833.

Roberts KG, Gloy BA, Joseph S, Scott NR, Lehmann J (2010b) Life cycle assessment of biochar systems: estimating the energetic, economic, and climate change potential: supplementary information. *Environmental Science & Technology*, **44**, S1–S32.

Shackley S, Hammond J, Gaunt J, Ibarrola R (2011) The feasibility and costs of biochar deployment in the UK. *Carbon Management*, **2**, 335–356.

Shackley SJ, Clare AJ, Joseph S, McCarl BA, Schmidt H-P (in press) Chapter 31: economic evaluation of biochar systems: current evidence and challenges. In: *Biochar for Environmental Management* (eds Lehmann J, Joseph S). Earthscan, London, UK.

Singh N, Abiven S, Torn MS, Schmidt MWI (2012) Fire-derived organic carbon in soil turns over on a centennial scale. *Biogeosciences*, **9**, 2847–2857.

Sohi S, Lopez-capel E, Krull E, Bol R (2009) Biochar, climate change and soil: a review to guide future research. *CSIRO Land and Water Science Report*, **5**, 17–31.

Song R, Lei H (2014) Emissions trading in China: first reports from the field. World Resources Institute website. Available at: http://www.wri.org/blog/2014/01/emissions-trading-china-first-reports-field (accessed 1 August 2014).

Sparrevik M, Field JL, Martinsen V, Breedveld GD, Cornelissen G (2013) Life cycle assessment to evaluate the environmental impact of biochar implementation in conservation agriculture in Zambia. *Environmental Science & Technology*, **47**, 1206–1215.

Steiner C, Glaser B, Geraldes WT, Lehmann J, Blum WEH, Zech W (2008) Nitrogen retention and plant uptake on a highly weathered central Amazonian Ferralsol amended with compost and charcoal. *Journal of Plant Nutrition and Soil Science*, **171**, 893–899.

Stern N (2006) *The Economics of Climate Change*. HM Treasury, London, UK.

Tang H, Qiu J, Van Ranst E, Li C (2006) Estimations of soil organic carbon storage in cropland of China based on DNDC model. *Geoderma*, **134**, 200–206.

UK Ministry of Defence (2014) Technology readiness levels (TRLs) in the project lifecycle. Available at: http://www.publications.parliament.uk/pa/cm201011/cmselect/cmsctech/619/61913.htm#note221 (accessed 23 April 2014).

Utterback JM (1996) *Mastering the Dynamics of Innovation*. Harvard Business Press, Boston, USA.

Van Zwieten L, Kimber S, Downie A, Morris S, Petty S, Rust J, Chan KY (2010) A glasshouse study on the interaction of low mineral ash biochar with nitrogen in a sandy soil. *Australian Journal of Soil Research*, **48**, 569–576.

Wang J, Pan X, Liu Y, Zhang X, Xiong Z (2012) Effects of biochar amendment in two soils on greenhouse gas emissions and crop production. *Plant and Soil*, **360**, 287–298.

Wang Q, Geng C, Lu S, Chen W, Shao M (2013) Emission factors of gaseous carbonaceous species from residential combustion of coal and crop residue briquettes. *Frontiers of Environmental Science & Engineering*, **7**, 66–76.

West TO, Marland G (2002) A synthesis of carbon sequestration, carbon emissions, and net carbon flux in agriculture : comparing tillage practices in the United States. *Agriculture, Ecosystems & Environment*, **91**, 217–232.

Woolf D, Amonette JE, Street-Perrott FA, Lehmann J, Joseph S (2010) Sustainable biochar to mitigate global climate change. *Nature Communications*, **1**, 56.

World Resources Institute (2014) Getting every ton of emissions right. Available at: http://www.wri.org/sites/default/files/calculation_spreadsheet_of_china_regional_grid_emission_factors.xlsx (accessed 3 May 2014).

Wu L, Chen J, Zhu XD, Xu YP, Feng B, Yang L (2001) Straw-burning in rural areas of China: caused and controlling strategy (in Chinese). *China Population, Resources and Environment*, **11**, 110–112.

Yan X, Ohara T, Akimoto H (2006) Bottom-up estimate of biomass burning in mainland China. *Atmospheric Environment*, **40**, 5262–5273.

Yang J (2014) China benchmark spot coal price drops first time in three months. Available at: http://www.bloomberg.com/news/2014-01-06/china-benchmark-spot-coal-price-drops-first-time-in-three-months.html (accessed 6 January 2014).

Yao Z, Wang SH, Jiang XH (2001) The current situation and approach of return straw to field in suburb of Shanghai (in Chinese). *Agro-Environment and Development*, **3**, 40–41.

Yu Z (2003) The developing trend of resources treatment of crop stalk in Fuzhou city (in Chinese). *Fujian Environment*, **20**, 31–32.

Zhang J, Smith K, Ma Y *et al.* (2000) Greenhouse gases and other airborne pollutants from household stoves in China: a database for emission factors. *Atmospheric Environment*, **34**, 4537–4549.

Zhang A, Cui L, Pan G *et al.* (2010) Effect of biochar amendment on yield and methane and nitrous oxide emissions from a rice paddy from Tai Lake plain, China. *Agriculture, Ecosystems & Environment*, **139**, 469–475.

Zhang F, Cui Z, Fan M, Zhang W, Chen X, Jiang R (2011) Integrated soil-crop system management: reducing environmental risk while increasing crop productivity and improving nutrient use efficiency in China. *Journal of Environmental Quality*, **40**, 1051–1057.

Zhang A, Liu Y, Pan G, Hussain Q, Li L, Zheng J, Zhang X (2012a) Effect of biochar amendment on maize yield and greenhouse gas emissions from a soil organic carbon poor calcareous loamy soil from Central China Plain. *Plant and Soil*, **351**, 263–275.

Zhang A, Bian R, Pan G *et al.* (2012b) Effects of biochar amendment on soil quality, crop yield and greenhouse gas emission in a Chinese rice paddy: a field study of 2 consecutive rice growing cycles. *Field Crops Research*, **127**, 153–160.

Zhang F, Chen X, Vitousek P (2013a) An Experiment for the World. *Nature*, **497**, 33–35.

Zhang Q, Zhou D, Zhou P, Ding H (2013b) Cost analysis of straw-based power generation in Jiangsu Province, China. *Applied Energy*, **102**, 785–793.

Zhang W-F, Dou Z-X, He P *et al.* (2013c) New technologies reduce greenhouse gas emissions from nitrogenous fertilizer in China. *Proceedings of the National Academy of Sciences of the United States of America*, **10**, 8375–8380.

Zhang Q, Zhou D, Fang X (2014) Analysis on the policies of biomass power generation in China. *Renewable and Sustainable Energy Reviews*, **32**, 926–935.

Zhao G, Che K (2012) A study on predicting coal market price in china based on time sequence models. *International Journal of Business and Social Science*, **3**, 31–36.

Zimmerman AR, Gao B, Ahn M-Y (2011) Positive and negative carbon mineralization priming effects among a variety of biochar-amended soils. *Soil Biology and Biochemistry*, **43**, 1169–1179.

Supporting Information

Additional Supporting Information may be found in the online version of this article:

Data S1. Supporting information text file, containing embedded tables and figures. This provides detailed referencing for the data used to construct the life-cycle analyses and cost-benefit analyses for the briquetting, gasification, and pyrolysis scenarios.