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Clare, A; Shackley, S; Joseph, S; Hammond, J; Pan, G; Bloom, A

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

ABBIE CLARE<sup>1,2</sup>, SIMON SHACKLEY<sup>1</sup>, STEPHEN JOSEPH<sup>3,4,5</sup>, JAMES HAMMOND<sup>6,7</sup>, GENXING PAN<sup>4</sup> and ANTHONY BLOOM<sup>1</sup>

<sup>1</sup>University of Edinburgh, Crew Building, Kings Buildings, Edinburgh EH9 3JN, UK, <sup>2</sup>Scotland's Rural College, Kings Buildings, Edinburgh EH9 3JG, UK, <sup>3</sup>Discipline of Chemistry, University of Newcastle, Callaghan, NSW 2308, Australia, <sup>4</sup>Nanjing Agricultural University, Nanjing 210095, China, <sup>5</sup>School of Materials Science and Engineering, University of New South Wales, Sydney, NSW 2052, Australia, <sup>6</sup>Key Laboratory of Biodiversity and Biogeography of East Asia, Kunming Institute of Botany, Chinese Academy of Sciences, Kunming 650201, China, <sup>7</sup>World Agroforestry Centre, East, and Central Asia, Kunming 650201, China

## 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<sub>2</sub>e<sup>-1</sup> 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<sub>2</sub>e per odt straw) is also lower than that of briquetting (1.35 MgCO<sub>2</sub>e per odt straw) and gasification (1.16 MgCO<sub>2</sub>e 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).

Correspondence: Abbie Clare, tel. +44 7745 701969, fax +44 131 662 0478, e-mail: Abbie.Clare@ed.ac.uk

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<sub>2</sub>, CO, CH<sub>4</sub>) that can be captured and used for energy (Brown, 2009).

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<sup>-1</sup> 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<sup>-1</sup>), 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<sub>2e</sub> per odt straw) outcomes associated with three straw utilization scenarios: straw briquetting and subsequent combustion for heat energy ( $S_{\text{Briq}}$ ); straw gasification for electrical energy ( $S_{\text{Gas}}$ ); and straw pyrolysis for biochar and electrical energy ( $S_{\text{Pyr}}$ ). These are compared to two baselines of straw reincorporation ( $S_{\text{Rein}}$ ) and straw burning ( $S_{\text{Burn}}$ ).  $S_{\text{Rein}}$  assumes that all straw is incorporated into the field whereas  $S_{\text{Burn}}$  assumes that straw is burned in-field.

### Technology scenario selection

Straw briquetting ( $S_{\text{Briq}}$ ) 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 ( $S_{\text{Gas}}$ ) 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 ( $S_{\text{Pyr}}$ ) 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  $S_{\text{Briq}}$ ,  $S_{\text{Gas}}$  and  $S_{\text{Pyr}}$  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  $\text{Mg}^{-1}$  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  $S_{\text{Briq}}$ ,  $S_{\text{Gas}}$  and  $S_{\text{Pyr}}$  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  $S_{\text{Briq}}$ ,  $S_{\text{Gas}}$  and  $S_{\text{Pyr}}$ , 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 $\text{yr}^{-1}$ )	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 $\text{kWh}^{-1}$ ); subsidized capital loans; tax breaks (Zhang <i>et al.</i> , 2014);	Feed-in-tariff for bioelectricity (\$0.12 $\text{kWh}^{-1}$ ); subsidized capital loans; tax breaks (Zhang <i>et al.</i> , 2014);
Local straw-burning subsidies	Avoided straw burning (\$28 $\text{Mg}^{-1}$ )	Avoided straw burning (\$28 $\text{Mg}^{-1}$ )	Avoided straw burning (\$28 $\text{Mg}^{-1}$ )

than this figure. The baseline agronomic value for biochar of \$110 Mg<sup>-1</sup> 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<sup>-1</sup> application rate. However, the more conservative estimate of Jeffery *et al.* (2011), assuming that a 10 Mg ha<sup>-1</sup> application stimulates 10% yield increases, gives biochar an agronomic value of just \$33 Mg<sup>-1</sup>. 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<sub>Pyr</sub>.

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<sup>-1</sup>; 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<sup>-1</sup> (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<sub>2</sub>), methane (CH<sub>4</sub>), and nitrous oxide (N<sub>2</sub>O)], and these are henceforth displayed in terms of their carbon dioxide equivalent global warming potential (CO<sub>2</sub>e), calculated according to IPCC guidelines of CO<sub>2</sub>e equivalence as 25 for CH<sub>4</sub> and 298 for N<sub>2</sub>O (IPCC, 2007). The GHG abatement potentials of S<sub>Briq</sub>, S<sub>Gas</sub> and S<sub>Pyr</sub> were calculated using S<sub>Rein</sub> as the baseline scenario, however the S<sub>Burn</sub> scenario is also displayed for reference. The analysis initially focuses on directly-attributable CO<sub>2</sub>e 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<sub>2</sub>e abatement potential of reduced soil N<sub>2</sub>O emissions and avoided fertilizer use as a result of biochar application.

Soil N<sub>2</sub>O 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<sub>2</sub>O 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<sub>2</sub>O emission reduction may have on the S<sub>Pyr</sub> 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<sub>2</sub>e MgN<sup>-1</sup> fertilizer as compared to an average of 9.7 MgCO<sub>2</sub>e MgN<sup>-1</sup> 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<sub>2</sub>e 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<sub>Gas</sub> and S<sub>Pyr</sub>, 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<sub>2</sub>e MWh<sup>-1</sup> (World Resources Institute, 2014).

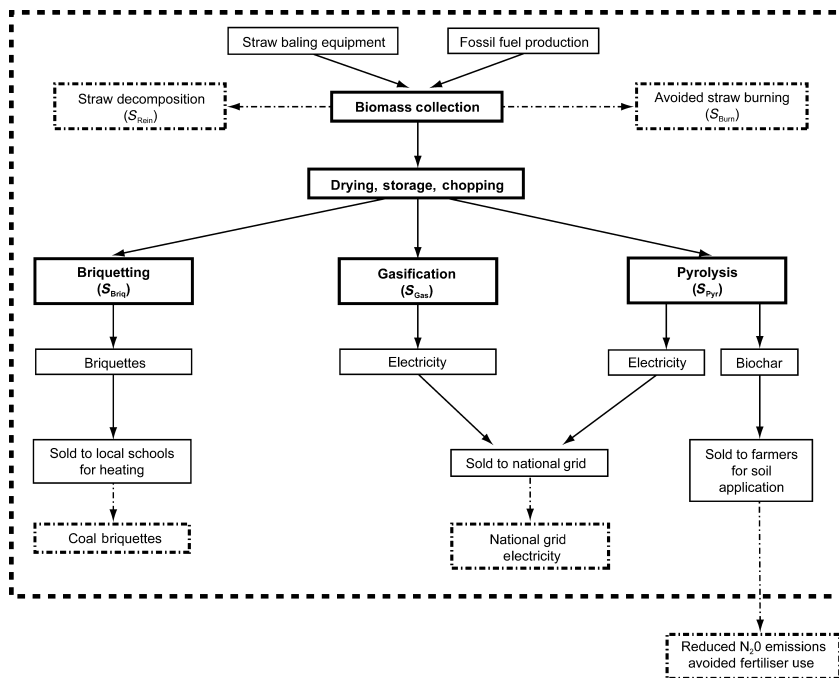
The details of GHG emissions associated with different phases of the lifecycle are given in the supplementary material (S9–S17). Many of the parameters used to estimate these emissions are considered uncertain, therefore published literature and expert opinion were also used to estimate the uncertainty 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<sub>2</sub>e emitted per odt feedstock. The impact of each parameter's value on the final result was investigated using sensitivity analysis.

## Results

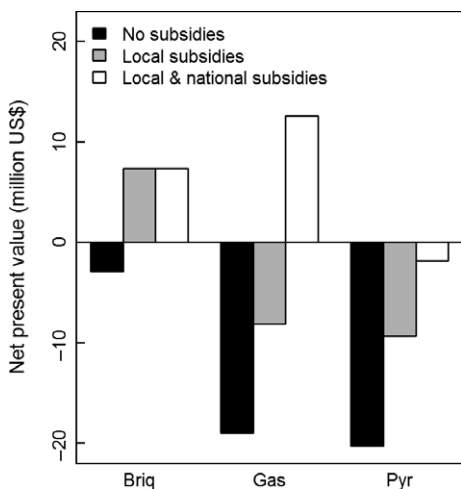
### Economic viability of briquetting, gasification and pyrolysis

Removing both national bioelectricity and local avoided straw-burning subsidies renders S<sub>Briq</sub>, S<sub>Gas</sub> and S<sub>Pyr</sub> 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<sub>Briq</sub> becomes profitable (NPV \$7.34 m), whereas S<sub>Gas</sub> and S<sub>Pyr</sub> 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<sub>Gas</sub> profitability (NPV \$12.60 m), increasing it above the unchanged S<sub>Briq</sub> NPV (\$7.34 m). Meanwhile, S<sub>Pyr</sub> 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<sub>Pyr</sub> 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<sub>2</sub>e, dashed boxes indicate CO<sub>2</sub>e 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.



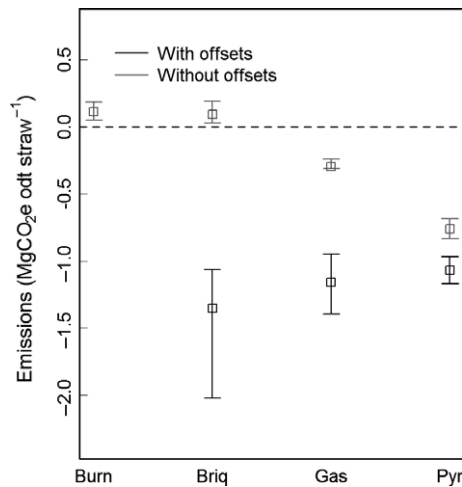
**Fig. 2** Net present value (million US\$), with and without Chinese government subsidies, for  $S_{Briq}$ ,  $S_{Gas}$  and  $S_{Pyr}$ .

most uncertain parameters modelled in this CBA. At the baseline agronomic value of \$110 Mg<sup>-1</sup>, (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<sup>-1</sup>, (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<sup>-1</sup> if all

other factors remain equal, or for \$206 Mg<sup>-1</sup>, if bioenergy subsidies are excluded. For the NPV of  $S_{Pyr}$  to equal that of  $S_{Gas}$ , biochar must sell for \$238 Mg<sup>-1</sup>. Interestingly, in 2014 Sanli New Energy Company reported their biochar retail price as \$259 Mg<sup>-1</sup>, 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<sub>2</sub>e abatement potential of briquetting, gasification and pyrolysis*

Figure 3 outlines the CO<sub>2</sub>e 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<sub>2</sub>e per odt straw) followed by  $S_{Gas}$  (1.16 MgCO<sub>2</sub>e per odt straw) and  $S_{Pyr}$  (1.06 MgCO<sub>2</sub>e per odt straw). This carbon abatement potential increases by 0.04 MgCO<sub>2</sub>e 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  $\text{MgCO}_2\text{e}$  abated per odt straw processed in  $S_{\text{Briq}}$ ,  $S_{\text{Gas}}$  and  $S_{\text{Pyr}}$ , including and excluding offsets from avoided fossil-fuel energy (black bars and grey bars, respectively). Uses  $S_{\text{Rein}}$  as the baseline, and displays  $S_{\text{Burn}}$  for reference.

receiving local and not national subsidies,  $S_{\text{Briq}}$  appears to offer the greatest  $\text{CO}_2\text{e}$  abatement potential. However,  $S_{\text{Briq}}$  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  $S_{\text{Gas}}$  and  $S_{\text{Pyr}}$  still provide some carbon abatement. In the case of  $S_{\text{Gas}}$  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  $S_{\text{Pyr}}$ , 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  $S_{\text{Pyr}}$  offers a more permanent GHG reduction than the other options.

#### Indirect $\text{CO}_2\text{e}$ abatement potential of pyrolysis

The application of biochar to agricultural land may contribute to the abatement potential of  $S_{\text{Pyr}}$  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  $\text{N}_2\text{O}$  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  $\text{N}_2\text{O}$  emissions from soil and improved fertilizer use efficiency.

*Reduced  $\text{N}_2\text{O}$  emissions from soil.* Table 2 combines data from a recent meta-analysis of biochar's impact on soil  $\text{N}_2\text{O}$  emissions (Cayuela *et al.*, 2014) with the baseline and reduced  $\text{N}_2\text{O}$  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  $\text{N}_2\text{O}$  emissions, the abatement potential of  $S_{\text{Pyr}}$  could be increased by 0.004–0.012  $\text{MgCO}_2\text{e yr}^{-1}$ . This represents a 1% increase in  $S_{\text{Pyr}}$ 's abatement potential, and we therefore suggest that the absolute contribution of biochar-induced soil  $\text{N}_2\text{O}$  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  $\text{MgCO}_2\text{e}$ , and thus that each odt of straw feedstock being used to produce biochar could abate an additional 0.39  $\text{MgCO}_2\text{e}$ .

Including these indirect effects of biochar application on avoided emissions from soil  $\text{N}_2\text{O}$  and fertilizer use reduction, the total abatement potential of  $S_{\text{Pyr}}$  increases to 1.46  $\text{MgCO}_2\text{e}$  per odt straw, which puts it ahead of

**Table 2** Calculations of avoided  $\text{N}_2\text{O}$  emissions per tonne feedstock pyrolysed

Biochar application rate (%)	0.5*	2*	1–2†
% $\text{N}_2\text{O}$ reduction from baseline	–40	–51	–27
$\text{N}_2\text{O}$ avoided (kg per odt)	0.021	0.007	0.007
Abatement potential ( $\text{MgCO}_2\text{e}$ 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<sub>2</sub>e per odt straw) and  $S_{Briq}$  (1.35 MgCO<sub>2</sub>e per odt straw) in terms of carbon abatement.

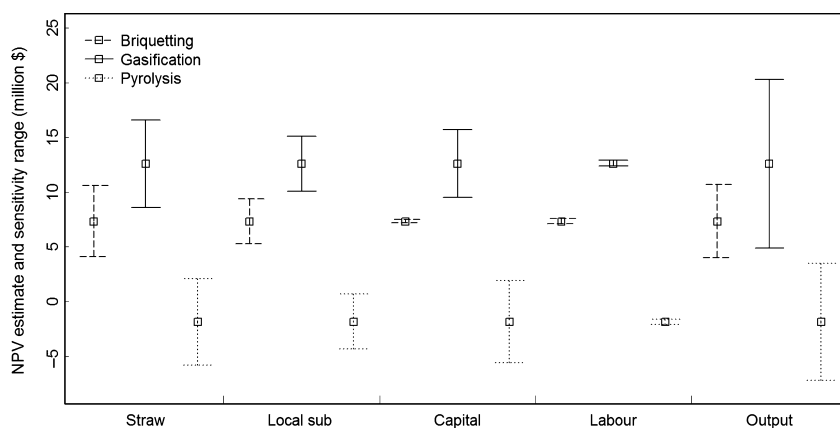
### Sensitivity analysis

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  $\pm 20\%$ , whilst keeping all other parameter values constant. The parameter values used in these sensitivity analyses are available in S19 and S20 of Data S1.

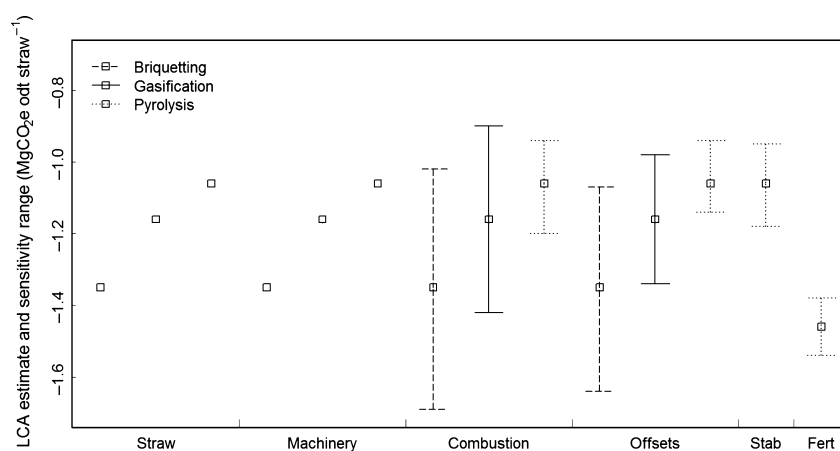
Figure 4 displays the influence of the following economic parameters on the overall NPV for each scenario:

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  $\pm 20\%$  and recording the impact on the overall NPV value.



**Fig. 5** Baseline carbon abatement estimates (MgCO<sub>2</sub>e 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  $\pm 20\%$  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  $S_{\text{Pyr}}$ 's NPVs do not overlap with the bottom range of the NPVs of  $S_{\text{Briq}}$  or  $S_{\text{Gas}}$ , 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  $\pm 20\%$  sensitivity analysis conducted on the following key parameters influencing the carbon abatement potential of  $S_{\text{Briq}}$ ,  $S_{\text{Gas}}$  and  $S_{\text{Pyr}}$ : 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  $S_{\text{Pyr}}$ .

#### Carbon abatement cost-effectiveness

In light of the Chinese government's carbon intensity reduction targets, it is important to consider the cost-effectiveness of  $S_{\text{Briq}}$ ,  $S_{\text{Gas}}$ , and  $S_{\text{Pyr}}$  in terms of  $\text{CO}_2\text{e}$  abatement. Our results show that all three technologies require assistance from carbon pricing to break-even, although  $S_{\text{Briq}}$  requires a significantly lower price than  $S_{\text{Gas}}$  and  $S_{\text{Pyr}}$  (see Table 3, where  $S_{\text{Pyr}}$  (D) includes only direct effects and  $S_{\text{Pyr}}$  (I) includes both direct and indirect processes discussed in this paper). Requiring a car-

bon price of \$7  $\text{MgCO}_2\text{e}$  per abated,  $S_{\text{Briq}}$  is the only technology studied here that can produce carbon abatement for less than \$25  $\text{MgCO}_2\text{e}^{-1}$ , 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  $\text{MgCO}_2\text{e}$  per abated) would only provide sufficient support to make  $S_{\text{Briq}}$  profitable (Song & Lei, 2014).

#### 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.  $S_{\text{Briq}}$  has the greatest carbon abatement potential (1.35  $\text{MgCO}_2\text{e}$  per odt straw as compared to 1.16 and 1.06  $\text{MgCO}_2\text{e}$  per odt for  $S_{\text{Gas}}$  and  $S_{\text{Pyr}}$ , respectively), and the highest economic abatement efficiency (requiring a relatively small carbon price of \$7  $\text{MgCO}_2\text{e}^{-1}$  abated, compared to \$61  $\text{MgCO}_2\text{e}^{-1}$  or \$51–71  $\text{MgCO}_2\text{e}^{-1}$  abated, for  $S_{\text{Gas}}$  and  $S_{\text{Pyr}}$ , 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

**Table 3** Comparing  $\text{CO}_2\text{e}$  abatement cost effectiveness for briquetting, gasification and pyrolysis

	$S_{\text{Briq}}$	$S_{\text{Gas}}$	$S_{\text{Pyr}}$ (D)	$S_{\text{Pyr}}$ (I)
Subsidy required (\$ tonne feedstock per processed)	5	34	36	36
Subsidy required (\$ $\text{MgCO}_2\text{e}^{-1}$ abated)	7	61	71	51

approximately \$110 Mg<sup>-1</sup> 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<sup>-1</sup> in the presence of subsidies for the NPV of S<sub>Pyr</sub> to equal that of S<sub>Gas</sub>, 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 60 000 Mg yr<sup>-1</sup> of BMCC products for local agricultural markets. Field trials in China have recently suggested that BMCCs (which pre-mix 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<sup>-1</sup>, increasing the S<sub>Pyr</sub> NPV to over 50 times that of S<sub>Gas</sub>. 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<sub>Gas</sub> and S<sub>Pyr</sub>, 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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### 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.