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Sustainable gasification–biochar systems? A case-study of rice-husk gasification in Cambodia, Part II: Field trial results, carbon abatement, economic assessment and conclusions

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ABSTRACT

In part I we described the gasification technology and characterised the physio-chemical properties and environmental impacts of the rice husk char (RHC) by-product. In part II we present summary results from field trials using the RHC, and provide an estimate of the carbon abatement and economic evaluation of the system. Statistically significant yield increases are demonstrated for RHC addition in irrigated rice cultivation (33% increase in paddy rice yield for a 41.5 t (dry weight) RHC application per hectare). The carbon abatement from the RHC addition is approximately 0.42 t CO_2 t^{-1} rice husk; including energy generation from gasification this increases to ca. 0.86 tCO_2 t^{-1}. Assuming a carbon value of \$5 t CO_2 t^{-1}, and agronomic value of \$3 t^{-1} RHC based on the field trials, the economic value of the RHC varies from \$9 t^{-1} (including only recalcitrant carbon) to \$15 t^{-1} (including avoided emissions from energy production). We summarise results from parts I and II, concluding that the gasification–biochar system meets many of the criteria of sustainability, but requires better waste water management and more field trials to demonstrate repeatable agronomic efficacy of RHC application.

1. Introduction

In Part I we presented information on the gasification technology that is being deployed in Cambodia for producing syngas to provide power in rice mills and ice-making factories. The gasifiers use rice husk as the fuel and produce ca. 35% rice husk char (RHC;dry weight proportion of feedstock). This RHC contains approximately 35% carbon (dry weight); ca. 30% of the carbon in the rice husk is conserved in RHC during the gasification process. The carbon is aromatic and largely unavailable to microbial or abiotic decomposition and mineralisation to CO_2 (Sohi et al., 2010); hence RHC has potential value as a way of sequestering carbon in the long-term (hundreds to thousands of years) and contributing to climate change mitigation. In part I we presented four criteria for assessing the sustainability of a gasification–biochar system, namely that it should: (a) produce and deploy biochar safely and without emitting excessive non-CO_2 greenhouse gases; (b) reduce net radiative forcing; (c) not increase inequality in access to, and use of, resources and (d) provide an adequate return on investment. In part I, we undertook an analysis of the physico-chemical properties of RHC, including an assessment of potential contaminants and other environmental and health and safety aspects. We concluded in part I that there are several health, pollution and contamination issues that require further investigation but which appear to be resolvable through process re-design and modification and appropriate regulation and management. We suggested, in particular, that RHC should not be mixed with the waste water from the process but should be discharged through a dry process using (for example) a screw auger.

In this part, we present results on utilisation of RHC in replicated field trials close to the gasification units. We then present and analyse information on the carbon abatement of the key components of the gasification–biochar system (though do not undertake a full life-cycle Assessment). We undertake a preliminary economic valuation of the use of RHC. A detailed economic analysis is not possible as yet due to the lack of experience and practice in the use of RHC. Finally, we pull together information from parts I and II in addressing the four criteria used to define sustainability of the gasification–biochar system.

2. Field trials using rice husk char

In order to assess the impact of the rice husk char (RHC) on agricultural systems, biochar pot and field trial plots were

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established in the North West of Cambodia , in Siem Reap and Pouk districts within Siem Reap Province, close to an ice-making factory that was powered by a gasifier using rice husk (EAP Sophat ice factory, Kralanh District). The hypothesised benefits of adding RHC to soil are the following: recalcitrant carbon storage, improved soil structure and properties, provision of nutrients, water retention, etc., hopefully resulting in higher agricultural yields (Dias et al., 2010; Sohi et al., 2010). At least some of the beneficial properties of biochar are probably related to its porous structure (see Fig. 1). The soil in the test area was sandy loam, sand or loamy sand (using the UK or USDA systems; Ashman and Puri, 2002).

Pot trials have been conducted using multiple cycles of crops (lettuce and cabbage) in biochar (20, 40 and 84 t ha^{-1} equivalent, dry weight (DW) basis) and non-biochar amended pots, both with and without compost and lake sediment additions. The principal variable measured is the yield of the crop (harvestable biomass). The results indicate a statistically significant positive yield response to biochar addition (Karve et al., 2010). Next, RHC was added to the soil before ploughing into fields cultivated with irrigated and rain-fed rice and a selection of horticultural crops. Three quarters of the RHC has a particle size between 0.5 and 2 mm, lending itself readily to soil application without excessive dust release.

For this pilot study, irrigated rice trials were planted in December 2009 and harvested in March 2010. It was decided to use a single large plot per application ($5 \times 10\ m^2$) with three application rates (fresh RHC weight of 10 t ha^{-1}, 30 t ha^{-1} and 60 t ha^{-1} equivalent). The results are not statistically representative due to lack of replication but give an initial indication of the impacts. The soil is sandy to a depth of ca. 8 cm (beneath which is clay) and acidic (pH 5.5) and flooded for rice cultivation to a depth of 0.5 m.

The RHC was scattered onto the surface by hand and then ploughed into the soil using traditional oxen-drawn ploughs. The RHC had a water content of 25% and therefore the oven dry biochar additions are 7.5 t ha^{-1}, 22.5 t ha^{-1} and 45 t ha^{-1}. The two lower additions resulted in a similar increase in yield of between 23 and 28%, while the higher addition increased yield by ca. 50%. There is a greater increase in yield per tonne of RHC added at the lower application level (3.7% increase per tonne) compared to the medium and high application levels (1–1.1% increase per tonne) suggesting that the incremental benefit of adding RHC declines with amount added.

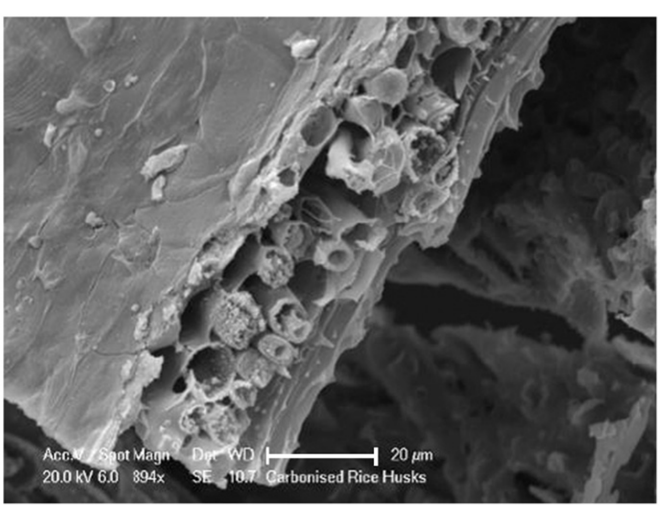

Fig. 1. Scanning electron microscope cross-section of rice husk char showing the presence of macro-pores.

Following these promising results, we then set up replicated field trials with irrigated rice in August 2010, harvested in late November 2010. Six plots (5 m × 5 m), three control and three amended with 41.5 t ha^{-1} (dry weight) RHC, were set up in a Latin-square layout on each farm, following the trial design in Haefele et al. (2011). There is 1 m gap between each plot and the border was discarded to avoid the edge effect. The results are shown in Fig. 2 and Table 1. The application of RHC increases both the paddy and straw yield. At farm 2, the paddy rice yield increase is ca. 10% but there is no statistically significant difference in yield in the plots with and without biochar addition (at a 95% confident interval). At farm 3, there is a statistically significant 33% increase in the paddy rice yield in the RHC amended plots compared to the controls ($p=0.033$). At farm 1 there is also a statistically significant increase in the yield of straw with and without RHC ($p=0.042$). The low yields of paddy and straw at farm 1 are explained by damage from rats that invaded the field. Farm 2 trials (unlike those at farms 1 and 3) used compost additions as well as RHC on both control and replicate plots. Positive yield results have also been obtained from studies of RHC additions to vegetable plots in the field, though these initial trials are non-replicated.

In summary, we have obtained promising results from the non-replicated and replicated pot and field trials with irrigated rice and vegetables. The replicated trials on rice indicate a statistically significant yield increase of ca. 33% for a 41.5 t ha^{-1} application. One other study using RHC at a similar rate in rice fields in SE Asia showed a statistically significant increase of between 16 and 35% in poor infertile soils, but no significant increase in better quality soils (Haefele et al., 2011). One hypothesis is that it is the nutrient addition from RHC that accounts for the yield improvements given that no additional chemical fertiliser is being added to soil in these plots. This would help to explain why no significant yield improvement occurred at farm 2, where the compost amendment would have been supplying nutrients. In biochar applications of 4, 8 and 16 t ha^{-1} to upland rice fields in Laos, Asai et al. (2009) found that statistically significant grain yield increases occurred, but only in soils with low available phosphorus and where sufficient nitrogen was available. The role of nutrients in explaining the benefits of biochar addition is currently under debate amongst soil scientists and agronomists, with some authors pointing to the variable, and frequently low, concentration of nutrients in biochars (Chan and Xu, 2010; Kimetu et al., 2008) and looking to other phenomenon in explanation of the effect, e.g. stimulation of microorganisms in association with biochar.

3. Carbon abatement assessment

Eq. (1) can be used to calculate the net carbon abatement (removal of, plus avoided, CO_2) arising from the use of rice husk for gasification–biochar and for alternative applications (combustion, direct field incorporation on dry soil or direct field incorporation into flooded rice fields; see Table 2):

$$CO_2na = CO_2av + CO_2recal + CO_2avoid - CO_2rel \quad (1)$$

where CO_2na is net carbon dioxide equivalent (eq.) abatement, CO_2av is carbon dioxide emissions avoided by replacement of fossil fuels, CO_2recal is recalcitrant carbon in the long-term (>100 years), CO_2avoid is carbon dioxide equivalent emissions avoided by thermochemical conversion rather than waste disposal with methane production (incorporation of rice husk in irrigated fields) and CO_2rel is carbon dioxide released by the biomass feedstock processing (all expressed in t CO_2eq. t^{-1} feedstock).

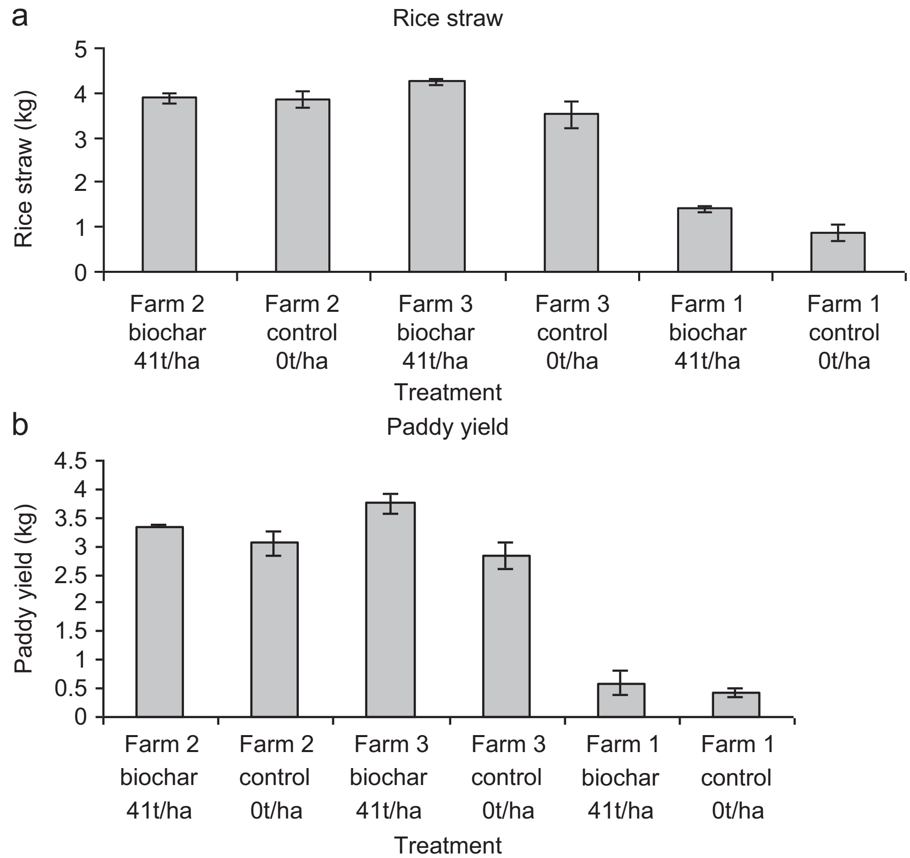

Fig. 2. The impact of RHC addition on (a) rice straw yield and (b) paddy yield (error bars shown; kg per plot).

Table 1
Results from the rice trials in Cambodia for a 41 t ha^{-1} application of RHC (dry weight). Compost was added to the soil at farm 2 (both control and with biochar plots). Hence results cannot be compared directly to those at farms 1 and 2.

Farm	Mean paddy			Mean straw		
	Control (t ha^{-1})	With biochar (t ha^{-1})	p	Control (t ha^{-1})	With biochar (t ha^{-1})	p
Farm 1	0.26	0.37	0.493 (n.s.)	0.54	0.88	0.042 (sig)
Farm 2	1.91	2.10	0.235 (n.s.)	2.42	2.43	0.955 (n.s.)
Farm 3	1.77	2.35	0.033 (sig)	2.20	2.65	0.076 (n.s.)

n.s. = not significant, sig = significant at $p \leq 0.05$.

Meanwhile

$$CO_2recal_{(100)} = BM_{tot} \times BC_{yield} \times CO_{2tot} \times CSF \tag{2}$$

where BM_{tot} is biomass total dry weight, BC_{yield} is biochar yield (ratio), CO_{2tot} is total CO_2 eq. content of fresh biochar and CSF is Carbon Stability Factor over 100 years (all expressed in t CO_2eq. t^{-1} feedstock):

$$CSF = 1 - C_{lab} - C_{unstab} \tag{3}$$

C_{lab} is the fraction of carbon that is labile (lost in a few weeks) and C_{unstab} is the fraction of carbon that is unstable as determined by accelerated ageing methods.

For RHC, CSF is 1–0.0036–0.07 = 0.926 (part I). For 1 t of rice husk, CO_{2recal} is therefore $1 \times 0.35 \times 0.35 \times 0.926 = 0.1134 \times 3.667 = 0.416$ t CO_2 t^{-1} feedstock for a 35% char yield.

In order to calculate the net CO_2 equivalent (eq.) abatement, the relevant baseline needs to be estimated. There are a number of different potential uses of rice husk:

(a) incorporation into soils under aerobic conditions;
(b) incorporation into soils under anaerobic conditions;
(c) open-burning on the field;
(d) combustion for bio-energy generation.

Table 2 presents the results for burning of rice husk in fields, for gasification–biochar, for aerobic decomposition and a mixture of aerobic and anaerobic decomposition in field. Approximately 10% of biomass carbon applied to flooded rice fields is emitted as methane (Haefele et al., 2011; Knoblauch et al., 2010). For every 1 t of applied biomass (straw or rice husk) with a C concentration of 40% (determined for oven dry straw) about 40 kg of carbon is therefore converted into methane, producing 53 kg of CH_4. Assuming a Global Warming Potential for CH_4 of 23, this corresponds to 3352 kg CO_2eq. per tonne straw/husk. The rest of the carbon is assumed to decompose aerobically. Biomass replacement is the assumption here, since the rice husk is a residue from agricultural systems and it can be assumed that rice will continue to be cultivated at similar levels in future years. In other words, CO_2 emissions arising from the biotic or abiotic conversion of

Table 2
Assessment of carbon abatement and emissions associated with key life-cycle stages of the gasification–biochar system.

Indicator	Burning in field	Gasification–biochar (with electricity generation)	Direct incorporation of rice husk into field—aerobic decomposition[b]	Direct incorporation of rice husk into field—anaerobic + aerobic decomposition[c]
Starting feedstock mass (t)	1	1	1	1
Carbon content at start (t)	0.38	0.38	0.38	0.38
C content at end (stabilised) (t)[a]	0.0114	0.1095	Yr. 0: 0.38 Yr. 1: 0.14 Yr. 2: 0.05 Yr. 3: 0.02 Yr. 4: 0.007	Yr. 0: 0.34 Yr. 1: 0.13 Yr. 2: 0.05 Yr. 3: 0.02 Yr. 4: 0.006
$CO_{2recal(100)}$ ($t\ CO_2\ t^{-1}$)	0.042	0.416	0.026	0.022
CO_{2av}—avoided emissions (replacement of fossil fuels)				
Avoided diesel per hour (l)[d]		23		0
Avoided CO_2 per hour		61		
Rice husk consumption rate per hour (kg)[e]		138		
CO_{2av} ($tCO_2\ t^{-1}$)	0	0.44	0	0
Carbon emission factor: 2.6391 kg CO_2 per litre (diesel)				
CO_{2rel}—CO_2 released by biomass processing				
CO_{2rel} ($t\ C\ t^{-1}$)	0.3686	0.2705	0.373	0.334
CO_{2rel} ($t\ CO_2\ t^{-1}$)	1.35	0.99	1.38	1.23
Total CO_2 abatement per tonne feedstock (t) assuming no biomass replacement				
CO_{2na} ($t\ CO_2\ t^{-1}$)	−1.31	−0.13	−1.35	−1.21
Total CO_2 abatement per tonne feedstock (t) assuming biomass replacement				
CO_{2na} ($t\ CO_2\ t^{-1}$)	0.042	0.86	0.026	0.022
CO_2 abatement from decomposition of biomass to methane assuming 40 kg of carbon is converted into 53 kg of methane (assuming a GWP for CH_4 of 23)				
CO_2 (CH_4) ($t\ CO_2\ t^{-1}$)			0	−3.352
Total CO_2 abatement per tonne feedstock (t; assuming biomass replacement)				
CO_2 total ($tCO_2\ t^{-1}$)	0.042	0.86	0.026	−3.33

[a] Bronzeoak (2003).
[b] Assumes an exponential decay function with a decay constant of 1.0. Empirical data on aerobic decomposition of rice husk over a three year period has been presented by Knoblauch et al. (2010) for a range of soils in Germany. The percentage of carbon mineralised to CO_2 ranged from 77.8% to 99.8% over 2.9 years depending on soil type. The % C mineralisation from the use of the decay function used in Table 2 is 95%, which is close to the mid-range value from the four soils tested by Knoblauch et al. (2010).
[c] Assumes an exponential decay function with a decay constant of 0.75. The experimental results of Knoblauch et al. (2010) suggest a lower mineralisation rate, with a low value of 30.9% C remaining after 2.9 years incubation and a high value of 54.3%.
[d] Based upon information from SME Renewables.
[e] Based upon information from the technology supplier that 6 kg of rice husk is used to replace 1 l of diesel (Ankur, 2010; Nagori, 2010) and verified by SME Renewables.

organic carbon to CO_2 are not accounted for here as it is assumed that an equivalent quantity of CO_2 will be taken up by plant photosynthesis in subsequent growth cycles.

From Table 2, the total net carbon abatement from rice husk gasification (including the avoided CO_2 emissions from the diesel fuel that is replaced by use of rice husk) is 0.86 $t\ CO_2\ t^{-1}$ rice husk. In the case of burning and aerobic decomposition, the carbon in the rice husk is mineralised through abiotic and/or biotic processes and can be assumed to be taken up in subsequent plant growth. Where rice husk are added to irrigated paddy fields and some anaerobic decomposition takes place, the net carbon abatement is 3.55 $t\ CO_2\ t^{-1}$ husk, even assuming biomass replacement. Hence, compared to alternative uses of the rice husk, the gasification–RHC option results in net carbon equivalent abatement of ca. 0.4 $t\ CO_2\ t^{-1}$ (cf. husk as a fuel), ca. 0.83 $t\ CO_2\ t^{-1}$ (cf. husk burnt in field or aerobic decomposition) and ca. 4.41 $t\ CO_2\ t^{-1}$ (cf. anaerobic and aerobic decomposition; Table 2). A survey in a local area would be required to establish the appropriate baseline against which to compare the gasification–RHC option. If rice husk were used 50% as fuel and 50% aerobic decomposition or burnt as the baseline, then the net carbon abatement from gasification-RHC would be 0.62 $t\ CO_2\ t^{-1}$ rice husk and the additional carbon abatement benefit. Assuming an average use of 4000 t of rice husk per gasifier per year, and 35 operational gasifiers, this amounts to ca. 87 kt CO_2 abatement for the existing fleet. With a nominal, conservative carbon price of \$5 tCO_2^{-1}, this carbon abatement could be worth \$0.4 million, though in reality there would be difficulties in attracting carbon finance. Given that ca. 1.5 million tonnes of rice husk are produced per annum in Cambodia, a cautious estimate of the theoretical carbon eq. abatement from the use of rice husk – if it could all be gasified – amounts to 630 kt $CO_2\ yr^{-1}$ (not accounting for the offset emissions from energy generation and assuming a biomass replacement, i.e. carbon-neutral, baseline).

We have not undertaken a complete life-cycle assessment. The upstream emissions in producing and transporting the feedstock have been omitted. We have relied upon SME Renewable Energy Ltd.'s data on the replacement of diesel fuel (see part I) and not fully accounted for process-based emissions. For example, tar that is produced during the process is collected and burnt, hence generating additional carbon equivalent emissions that have not been accounted for here. Upstream emissions associated with construction of the plant are not accounted for, nor are transport emissions associated with movement of the rice husk (if appropriate) and of the char from the factory to the field. These transport emissions are likely to be low, however, because rice mills are located in rice-growing areas and suitable agricultural locations are very likely to be situated close to rice husk gasifiers given the highly rural character of Cambodia. If we assumed a 25 km round trip, and a carbon emission factor (CEF) of 0.272 $kg\ CO_2\ km^{-1}$ for a 3.25 t diesel-fuelled vehicle, this would be ca. 3.5 $kg\ CO_2^{-1}$ per tonne char—which is under 0.5% of the overall abatement. Our baseline measurements are also overly simplistic and do not account for the production of soot, black carbon, N_2O, CH_4 and other climate forcing gases during biomass burning in field. A more detailed analysis of the full life-cycle emissions, by Field and Tanger, based upon actual equipment and field measurements is available (Karve et al., 2010) and will be published separately.

We have, furthermore, not included some potentially important additional carbon abatement from the inclusion of biochar in soils arising from the priming of soil organic carbon in soils (Sohi et al., 2006; Liang et al., 2010). There may also be some suppression of other soil GHG flux (nitrous oxide or methane emissions) arising from biochar addition (Yanai et al., 2007; Singh et al., 2010) but we have not attempted to include these due to high uncertainty. The field trials are being undertaken with subsistence farmers, who largely do not use synthetic fertilisers (or else use very small quantities), which tend to be the largest source of N_2O (although organic amendments do also generate some trace gas emissions). There is a high degree of uncertainty associated with the indirect impacts of biochar in soils, with estimates in the literature ranging from 0 to 40% of the overall CO_2 eq. abatement on a life-cycle basis (Shackley et al., 2012). Without empirical studies of the impacts of the RHC upon the specific soils in Cambodia and under typical agricultural management practices, we decided that it would be preferable to leave out the indirect impacts.

The net carbon abatement of RHC at 0.86 t CO_2 t^{-1} feedstock is similar to the value obtained in other studies of biochar, e.g. (Hammond et al., 2011) at 1–1.4 t CO_2eq. t^{-1} and (Roberts et al., 2010) at 0.8 t CO_2eq. t^{-1} for a range of feedstocks. Those studies have assumed pyrolysis as the thermochemical conversion technology, rather than gasification. As noted in part I, the gasification of rice husk shares some similarities with pyrolysis, this probably being due to the shielding of the carbon matter by the silica shell (see part I) and the overall carbon conservation is ca. 30% compared to ca. 50% in slow pyrolysis and ca. 2–10% in typical gasification. The high silica content also reduces the C content of the biochar (35% by dry mass compared to typical values of 60–90% C). As a consequence, the contribution of the stabilised C in the char to the overall net C abatement is lower than in many pyrolysis–biochar systems (at 49% compared to 50–80% if the indirect effects of biochar in soils are ignored), but the generation of renewable energy through gasification is more efficient than is assumed for many pyrolysis technologies. Therefore, the contribution of the offset carbon dioxide emissions from bioenergy generation is higher for the gasification—than for the pyrolysis–biochar system.

4. Economic assessment

It is not the aim of this paper to assess the economic viability of gasification units—their rapid adoption in Cambodia in rice mills and ice making factories strongly suggests that they are an attractive investment and enjoy a good return on investment in this specific context (see also Field and Tanger's analysis in Karve et al. (2010) and forthcoming). Whether they are economically attractive investments in other countries is another matter, due to market, technology, commercial and pricing differences. Furthermore, if the value of rice husk increases due to competition for their use in biomass combustion facilities (as has happened elsewhere) the existing economic viability of gasifiers in Cambodia could be impacted.

Since the mid-2000s to the present time, RHC has largely been a free by- or waste-product. Whether RHC will continue to be free will depend upon whether there is sufficient demand for the material to create a market where it can be bought and sold. Some mills have succeeded in selling RHC to farmers at price of 300–400 riel per 25 kg bag, or ca. \$3–4 per tonne. A few other buyers have emerged, e.g. garden centres, which use RHC as a potting medium. Other mills have given RHC away to farmers or for use in land reforming, e.g. to fill in ponds.

If we assume a value of \$250 per tonne of unmilled paddy rice, then the value of 41.5 t ha^{-1} RHC application at farm 3 (Table 1) is \$145. Making the very crude assumption of a linear relationship between RHC application level and yield, this would imply an agronomic value per tonne of RHC of \$3.5, which is, interestingly, similar to the purchase price of RHC at some mills. If we assume a low transport and application cost of ca. \$0.5 t^{-1}, the value to the farmer of RHC (assuming it is free) would be ca. \$3 t^{-1}. There is some evidence from our earlier non-replicated field trials of a leveling off of the yield benefits of biochar with RHC application levels; hence it might be the case that lower application rates (e.g. 10 t ha^{-1}) would provide a higher return per hectare than higher application rates (e.g. 40 t ha^{-1}) and make the application of RHC a more attractive proposition to the farmer.

A further possible revenue stream is from sale of carbon credits under the Clean Development Mechanism (CDM) or the voluntary carbon market—both are speculative at present because biochar is not currently included within these mechanisms. Assuming a carbon price of \$5 per tonne CO_2 avoided or removed, and the availability of RHC at zero-cost, the value of a tonne of RHC is the product of price CO_2 t^{-1}, the inverse of the char yield and total CO_2 abatement t^{-1} feedstock; i.e. $5 \times 1/0.35 \times 0.86 =$ ca. \$12 t^{-1} RHC. (Note that this value would be ca. \$6 t^{-1} RHC if only the stabilised carbon in the char is accounted for and the avoided fossil fuel emissions not included.)

If a baseline of anaerobic decomposition were accepted, then the value could be significantly greater—to ca. \$57 t^{-1}. Hence, a 10 t ha^{-1} application rate could generate an income of \$60 ha^{-1} (RHC abatement only; increasing to \$120 ha^{-1} if bioenergy is included and to \$570 ha^{-1} against a baseline of anaerobic decomposition). This compares to an estimate of the agronomic value from the field trials of ca. \$30 ha^{-1}. The total value of RHC (carbon abatement plus agronomic benefit) is between \$9 t^{-1} (char carbon only) and \$15 t^{-1} (including also offset emissions from bioenergy; or \$63 t^{-1} for an avoided anaerobic decomposition baseline). Potentially, therefore, RHC can be a valuable addition to farm incomes through improving yields and especially if a carbon value for the RHC could be realised. Because the carbon is fixed during the gasification process, incorporation into the field *per se* does not increase the carbon abatement (excluding indirect effects of the biochar in the soil). Hence, it would be necessary to include the gasification operation within the project boundary in addition to the field incorporation in order to acquire any carbon financing for the biochar.

5. Conclusion

In part I, we set out four criteria by which the sustainability of a gasification–biochar system could be evaluated. To what extent have these criteria been met?

(a) Criterion 1: produces and deploys biochar safely and without emitting non-CO_2 greenhouse gases, which would obviate the carbon abatement benefit. Questions remain regarding the safety of RHC for human health. More work needs to be done but it is likely that appropriate precautions and practices can limit the risks adequately. Uncertainty remains in whether such precautions would be implemented and enforced, however. Likewise, there are pollution and contamination issues associated with the production and storage of RHC. Our evaluation to date relates only to the use of RHC, *not* to mixtures of RHC and sludge from the settling ponds. Such sludge contains high quantities of PAHs, some of which are known carcinogens. Before gasifiation–biochar systems could be further promoted as a sustainable option, far more effective and comprehensive clean-up of the black water, sludge, tars and other waste streams will be necessary. Issues such as burning of tars and sludge in a non-controlled fashion are of concern. Recently, technology provider Ankur Scientific has developed gasifiers that discharge the char in dry form (Nagori, 2010). A small part of this dry char discharge can itself be used for water treatment but the large part can be put to use as biochar as the dry discharge eliminates the contact between the char and the dirty process water (see part I). This innovation is likely to be important in improving the sustainability of the GBS but may be difficult, if not impossible, to retrofit on already-installed units.

(b) Criterion 2: reduces net radiative forcing. The existing system performs well under this criterion with a net abatement of 0.42 t CO_2 t^{-1} rice husk (and double this if the avoided fossil fuel emissions from bioenergy generation are included). On the other hand, if gasifiers emerge that utilise locally sourced wood that is not replaced (as is already happening), then any benefit is probably foregone in carbon released on land-conversion (Walker et al., 2010).

(c) Criterion 3: does not increase inequality in access to and use of resources. Where RHC is given away for free, or at a very low cost, the overall effect should be to bring income to subsistence farmers; hence the system would score well on this criterion. RHC could increase in cost, however, as gasifier operators come to realise their value to farmers. Commercial producers could buy up the majority of the supply, thereby denying the subsistence farmers the opportunity to benefit from the availability of RHC. Gasifiers using locally available biomass could reduce the availability of such biomass to households that rely upon such biomass for cooking. A further equity issue relates to the provision by the farmer of the rice husk, effectively for free, to the rice mill. Some of the new-found value of the rice husk should be shared with the farmer (Parnphuneesup and Kerr, 2011).

(d) Criterion 4: provides an adequate return on investment. This is currently hard to evaluate because of uncertainty in agronomic value and in whether a carbon market might develop for biochar deployment. Where RHC is 'free' its use almost certainly does provide a good prospect of a small positive return on investment at least under the soil and agronomic context we examined. This value will increase if a carbon market can develop, which allows for the recalcitrant carbon and offset fossil fuel emissions to be valued in carbon markets. The carbon abatement value of RHC looks to be several times larger than the agronomic value. However, the gasification unit needs to be included in the project boundary for any carbon financing, since it is during gasification that carbon fixation takes place (Monfries, 2011). This could make meeting the requirements of project additionality under the CDM rules difficult, however, since installation of gasifiers at rice mills in Cambodia is already economically viable.

The results presented suggest that the gasification–biochar system studied here, making use of a readily available agricultural residue (rice husk), offers potential not only as a way of effecting the long-term storage of carbon but also in improving crop productivity. The gasification–biochar system can also, potentially, effect a more sustainable disposal route for RHC, which otherwise may be a local pollutant. More research is required on the agronomic benefits of RHC on a range of crops and soils and on the health and environmental risks arising from RHC production, storage and deployment.

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