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Greenhouse Gas Mitigation of Rural Household Biogas Systems in China: A Life Cycle Assessment

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Abstract: Rural household biogas (RHB) systems are at a crossroads in China, yet there has been a lack of holistic evaluation of their energy and climate (greenhouse gas mitigation) efficiency under typical operating conditions. We combined data from monitoring projects and questionnaire surveys across hundreds of households from two typical Chinese villages within a consequential life cycle assessment (LCA) framework to assess net GHG (greenhouse gas) mitigation by RHB systems operated in different contexts. We modelled biogas production, measured biogas losses and used survey data from biogas and non-biogas households to derive empirical RHB system substitution rates for energy and fertilizers. Our results indicate that poorly designed and operated RHB systems in northern regions of China may in fact increase farm household GHG emissions by an average of 2668 kg·CO_2-eq·year^{-1}, compared with a net mitigation effect of 6336 kg·CO_2-eq per household and year in southern regions. Manure treatment (104 and 8513 kg·CO_2-eq mitigation) and biogas leakage (−533 and −2489 kg·CO_2-eq emission) are the two most important factors affecting net GHG mitigation by RHB systems in northern and southern China, respectively. In contrast, construction (−173 and −305 kg·CO_2-eq emission), energy substitution (−522 emission and 653 kg·CO_2-eq mitigation) and nutrient substitution (−1544 and −37 kg·CO_2-eq emission) made small contributions across the studied systems. In fact, survey data indicated that biogas households had higher energy and fertilizer use, implying no net substitution effect. Low biogas yields in the cold northern climate and poor maintenance services were cited as major reasons for RHB abandonment by farmers. We conclude that the design and management of RHB systems needs to be revised and better adapted to local climate (e.g., digester insulation) and household energy demand (biogas storage and micro power generators to avoid discharge of unburned biogas). More precise nutrient management planning could ensure that digestate nutrients are more effectively utilized to substitute synthetic fertilizers.

Keywords: biogas; manure storage; energy substitution; subsidy; small-scale; subsistence farming

1. Introduction

1.1. Rural Household Biogas Deployment

Rural household biogas (RHB) is an important source of renewable energy for smallholder farmers that utilizes human and animal waste products [1]. Several studies have demonstrated the economic and greenhouse gas (GHG) mitigation benefits of RHB, which compares well with other renewable energy resources [2]. RHB can also reduce pollution arising from alternative management of livestock manure and household waste, and the use of wood fuel, including odor, smoke and diseases [3–5]. In addition, the fermented substance is a quality fertilizer, rich in nutrients, which can be used to substitute synthetic fertilizers. Therefore, RHB is well suited to small-scale deployment in rural areas where it can be easily integrated into farm systems to improve their wider sustainability, and managed with relatively little operation and maintenance effort [6]. For this reason, RHB has been increasing in popularity in areas where smallholder farmers predominate, such as Southeast Asia, China and Africa. Considerable governmental funds and social capital have been invested in RHB in these regions [7–9]. For instance, the Chinese government regards RHB as a key solution to renewable energy generation in rural areas and has issued enormous stimulus policies (Figure S1). Nonetheless, there are increasing doubts about the real economic and environmental performance of RHB systems, and the cost-benefit return of government subsidies for RHB, and an increasing number of RHB systems in China are being abandoned. For example, in 2006 in China, the number of newly installed and abandoned facilities were 4.0 million and 0.34 million, respectively. These number increased to 5.2 and 0.64 million in 2009 respectively [10].

1.2. Environmental Performance

There is considerable debate about the energy output and GHG savings achieved by RHB in China. Liu et al. [11] estimated that Chinese RHB provided 832,749 TJ energy and avoided 73,158 Gg CO_2-eq of GHG emissions from fossil fuel and fuel woods between 1991 and 2005, concluding that governments should actively support RHB construction. However, Chen et al. [5] pointed out that only 19% of the RHB systems in China operate as designed. Lack of effective operational management and maintenance, and insufficient gas storage facilities, contribute to unstable operation and leakage of biogas—containing methane (CH_4) with a high global warming potential (GWP)—increasing net GHG emission [12]. Groenendaal et al. [13] pointed out RHB neither reduced fossil energy consumption, nor lowered fertilizer application in a number of typical Chinese villages studied. Sun et al. [14] considered that governmental subsidies were not well targeted to promote development of RHB in China, mainly because most investment was directed towards construction rather than operation and maintenance of new systems. Due to poor operation, many RHB have been abandoned, representing a poor return on public investment [15,16].

Contrasting results from previous studies were largely caused by different methodologies. RHB systems involve many farm and household activities, including animal production, crop production and household functions. They deliver a range of services including waste management, energy utilization and nutrient recycling, which determine economic and environmental benefits when compared against the counterfactual methods of delivering those services that RHB replaces. However, previous assessments of RHB typically addressed only one or two aspects at a time, such as GHG mitigation from energy substitution [11,17] or construction impacts [18]. Important factors that affect the environmental and economic performance, and RHB adoption, have often been ignored [18–20]. In particular, biogas leakage rates may in practice range up 100% of the biogas produced, at best significantly offsetting the GHG mitigation achieved through energy substitution [9], but may studies neglect biogas leakage because of difficulties in quantification [11,13]. In addition, there are large variations in estimates of biogas production—a critical variable for economic and environmental performance. For example, using statistical data on biogas tank sizes and raw material (substrate) inputs, Zhang et al. [17] estimated that average biogas production volume in RHB systems is approximately 400 m^3 per household per year, while Yang et al. [2] estimated the volume to be approximately 450 m^3 per household per year based on energy analysis [9]. According to the specific substrate mix and prevailing ambient temperatures across China represented in the ABEPE model, Tang et al. [21] estimated that biogas volumes vary from 130 to 1335 m^3 per household per year. Large uncertainties in biogas production and biogas leakage rates have hampered efforts to validate the economic and environmental sustainability of RHB, and to inform further development. Substitution of chemical fertilizers is also usually neglected when calculating the GHG mitigation potential of RHB systems [20].

There remains a need to apply life cycle assessment (LCA) to evaluate the net environmental effects of Chinese RHB systems, considering the entire value chain (construction—operation—energy utilization—residue treatment), and utilizing data that represents realistic operating performance. Recent LCA studies in Europe have highlighted the need to expand system boundaries in order to capture important substitution effects such as fossil energy replacement, avoided manure storage and organic waste management (e.g., composting) and fertilizer substitution [22–26].

1.3. Study Objectives

The aim of this study is to evaluate the overall GHG balance of typical Chinese RHB systems from a life cycle perspective, and to inform practicable options to enhance GHG mitigation. Specific objectives are to: (1) apply LCA to evaluate the net GHG mitigation effect of RHB systems integrated into small household farms; (2) track RHB system deployment in different areas in relation to driving forces; (3) propose policy recommendations to improve the effectiveness of biogas GHG mitigation.

2. Materials and Methods

2.1. Characterisation of Typical Chinese RHB Systems

The volume of a typical Chinese RHB tank is approximately 8 m^3 (Figure 1), based on the daily waste output and energy needs of general small farm household. A typical family with 2 to 5 members and 1 to 3 pigs (or some other animals) generates 1 to 2 Mg of fresh feces and kitchen waste yearly in total (survey data [21]), equivalent of 300 to 500 m^3 biogas yearly (450 m^3 as mean value [21]). Chinese RHB tanks are usually constructed under the yard, and connected to the kitchen, animal houses and toilets so that the waste can enter the biogas tank naturally under gravity. Biogas residues and slurry are drawn out by farmers artificially, whilst biogas is piped to the cooking range, lamps and lanterns in the kitchen—and used primarily for cooking. Farmers may discharge surplus biogas that is not combusted after collection, since there is usually very limited biogas storage infrastructure (usually just the biogas tank head space). Unlike large-scale biogas plants, small-scale RHB systems in China do not require heating or stirring, so there is no external energy input or parasitic energy demand. Specialized service companies are responsible for the biogas system construction, repair and maintenance. The construction cost for a biogas digester is about 3000 RMB (~450 dollars), of which 800 to 1200 RMB is subsidized by the Chinese government.

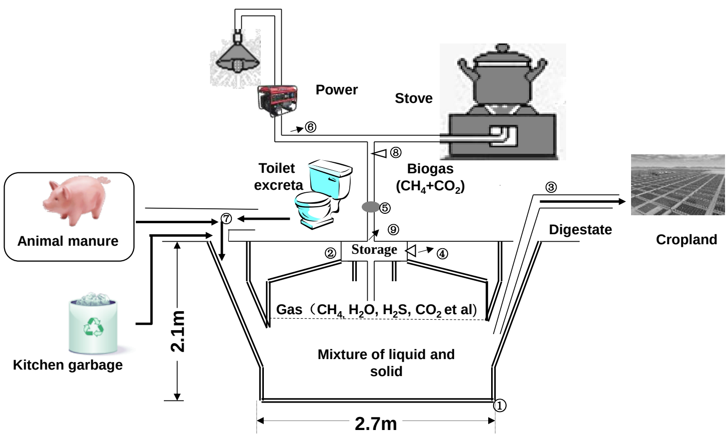

Figure 1. A typical digester is built below ground. Pipe joints, safety valve, purifier and other connectors are common gas leakage points. ① anaerobic digester, ② connecter in tank top (CTT), ③ digestate outlet, ④ safety valve, ⑤ purifier, ⑥ pipe joint, ⑦ inlet, ⑧ valve (switch).

2.2. Research Sites

This study involved the collection of data on RHB system operation from two villages between 2009 and 2014: Shuanghe village (Shu, N 30°24′, E 104°37′) (Jianyang City, Sichuan Province, Southwest China); Zhuwangzhuang village (Zhu, N 39°35′, E 118°35′) (Luannan County, Hebei Province, North China). The use of biogas in Sichuan Province dates back to 2000 years ago, during the Western Han Dynasty [27]. Nowadays, Sichuan is still one top regions for biogas use owing to the prevalence of intensive pig production, with 70 million pigs slaughtered annually (60% from intensive indoor systems, and 40% from small free-range farms with fewer than 50 animals) [10]. Importantly, the Sichuan climate is well suited to biogas production; the mean daily temperature is above 10 °C for between 240 and 280 days of the year (10 °C being the minimum temperature for significant biogas production). Hebei Province is another region where biogas has developed quickly owing to intensive animal production. Dairy farming is popular, and most dairy farms are small-scale, household units—i.e., 1 to 4 cows per family [10]. However, the mean daily temperature is above 10 °C for only 150 days per year (from April to September) in this Province.

2.3. Data Sources

Data in this study were obtained from household surveys, on-site monitoring of biogas consumption and leakage, and collected digestate quantities for the two villages. Household surveys were carried out twice in 2010 and 2014 respectively, between August and October (Table S1). Data from the 2010 survey (n = 95 and 303 for biogas and non-biogas households, respectively) were used to calculate a GHG balance; data from the 2014 survey were used to evaluate changes in RHB systems. For the surveys, all rural households were divided into two groups: households with biogas and households without biogas systems. We tried our best to visit every household in the villages. In 2014, we investigated the same households again (see Table S1 for more details). Although some household moved out, we still tried to track 50% of samples.

We chose 15 biogas households in each village, selected randomly according to the time of their RHB construction, to measure biogas usage. Consumption meters (GB/T 6968, Precision B by Chongqing Mountain City Gas Equipment Co., Ltd., Chongqing, China) were installed and monitored between August 2009 and August 2010, in biogas pipes 1 m away from biogas stoves, to measure all biogas combusted for cooking and heating (Figure 1). Because some of the households relocated (migrated to cities) or meters were damaged during the one-year period, we obtained data for 13 households in each village. Gas leakage from RHB systems, including release of surplus biogas by farmers, was estimated from the gap between gas production and gas usage. We estimated biogas production using the ABEPE model using animal numbers to estimate substrate volumes (details in Supplementary Materials). The ABEPE model estimates energy production according to biomass (substrate) input, energy conversion coefficients for each biomass type, and climate conditions.

We also measured gas leakage from common leakage positions (identified in Figure 1) across the 15 monitored RHB systems in each village, using a PGM-50Q composite gas detector (built by RAE Systems, Germany). Detection points included the safety valve, the biogas digester, pipeline connections, the biogas tank roof valve and its joints (Figure 1). Monitoring was undertaken over periods of four days, at 45-day intervals (eight times per RHB site in the experiment year). Measurements were taken at 10 a.m. on the first day, and repeated for three continuous days. Leakage was recorded as present or absent from each position at each RHB site. Meanwhile, we extracted samples of substrate inflows (primarily animal slurry) and digestate outflows from the 15 households, and undertook nutrient analyses. Nutrient concentrations were analysed using standard laboratory methods, i.e., total nitrogen (Kjeldahl method), total phosphorus (ultraviolet spectrophotometry) and total potassium (flame photometer, with Bao's method [28]).

2.4. Method to Caluculation GHG Mitigation

We established an LCA model to calculate GHG mitigation of Chinese RHB (Figure 1), comparing RHB households with non-biogas households at the single household level and according to the LCA principle of "cradle to grave" (Figure 2). The main raw materials entering the RHB systems were animal feces (as shown in the survey questionnaire, no straw or kitchen wastes were put into the RHB systems in either village). Sealed biogas tanks in which animal feces and urine decompose under anaerobic conditions substitute traditional manure storage systems such as uncovered anaerobic lagoons and manure stacks leading to important consequences for GHG emissions, especially CH_4 and N_2O (Table S2). Biogas generated in the tank is an alternative energy carrier that replaces traditional fuels including coal, fuel wood, electricity and natural gas. Due to different calorific values across fuels, and different energy transformation coefficients for different stoves (Table S3), GHG emissions vary by fuel source (Table S4). A fraction of biogas is not used because it leaks out of the system, releasing CH_4 to the atmosphere (a potent GHG with a global warming potential 25 times higher than CO_2). Solid and liquid digestate by-products can be re-used on farmland as fertilizer, replacing undigested manures and chemical fertilizers. Each type of fertilizer has different nutrient contents and different carbon footprints (carbon emission from their production, transportation, and field application) (Tables S2 and S8). RHB systems give rise to GHG emissions from two other sources: first, the production of cement and bricks used for the construction of biogas tanks gives rise to significant GHG emissions (large amounts of energy required to produce them); second, extra energy used for heating and stirring biogas tanks are popular in large scale biogas systems, which may incur GHG emissions if external fuels and electricity are used. However, the latter process energy is not required in simple RHB systems. Based on the aforementioned factors, the GHG balance was calculated (see Supplementary Materials for more details). Table 1 synthesizes the main data sources used to estimate activity data and emission factors.

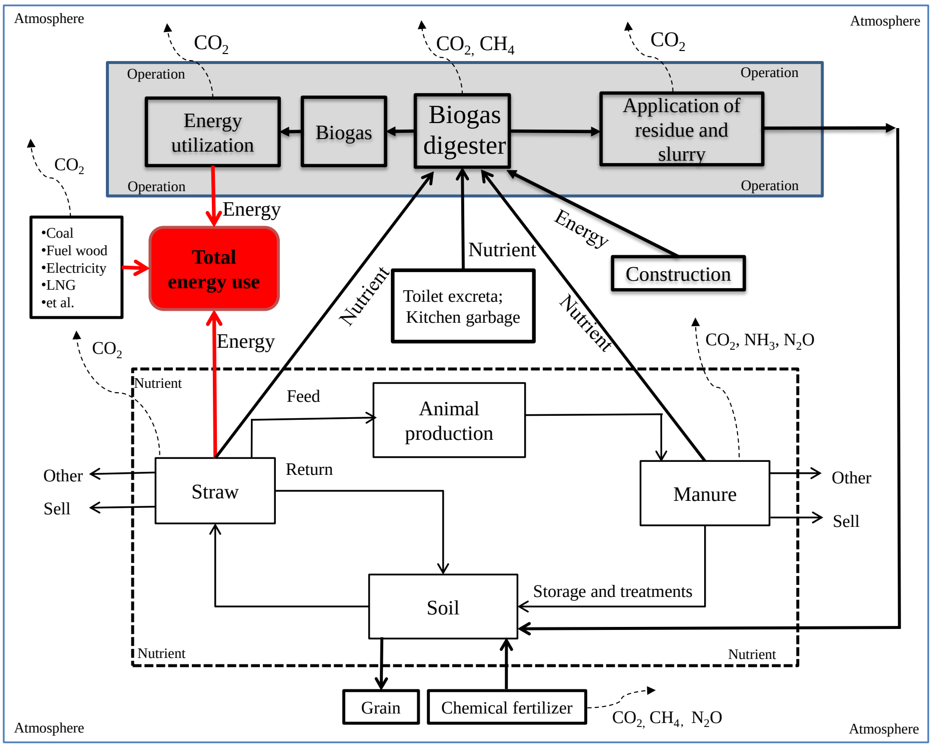

Figure 2. Life cycle assessment of GHG mitigation from biogas facilities. The solid line represents the digester operation flow, energy and materials, and the dotted arrow represents GHG emissions from biogas facility and associated farming system. The dotted box represents nutrient recycling within farming system.

Table 1. Overview of activity data and emission factors used to calculate the net GHG balance of RHB systems, using empirical survey data from biogas and non-biogas households.

Process	Activity Data	Emission Factors
Construction	Survey, Table 2	Supplementary Materials
Biogas production	ABEPE model [21], Table S9	-
Biogas leakage	Difference between production and use	
Biogas combustion	Measurements and survey, Table S14	Table S4 (IPCC, 2006)
Energy substitution	Survey, Table S14 and energy densities (Tables S3 and S4)	Table S5 (Supplementary Materials)
Avoided manure storage	Survey, Table S12	Table S2 (IPCC, 2006)
Nutrient substitution	Survey, Table S16	Table S8 (Supplementary Materials)

3. Results

3.1. Overall GHG Mitigation of RHB Systems

The GHG balance of RHB systems differed massively between the villages of Zhu and Shu. In Zhu, RHB increased GHG emission by 2668 kg·CO_2-eq per household and year, while in Shu RHB decreased GHG emission by 6336 kg·CO_2-eq per household and year (Figure 3). Whilst previous studies have tended to focus on energy substitution, avoided manure storage and biogas leakage made the greatest relative contributions to mitigation and emission of GHGs, respectively. Construction emissions, also often ignored by previous studies [11], made a significant contribution to emissions. Avoided manure storage led to annual mitigation of 8513 kg·CO_2-eq per household in Shu and 104 kg·CO_2-eq in Zhu. There was wide variation in the relative importance of different GHG sources between the villages. For example, Zhu systems emitted on average 173 and 533 kg·CO_2-eq per household and year for construction and biogas leakage, respectively, while Shu emitted 305 and 2489 kg·CO_2-eq per household and year from these processes, respectively. Nutrient substitution did not led to GHG mitigation as expected, but increased GHG emissions. More detailed descriptions of GHG effects arising from key processes are given below.

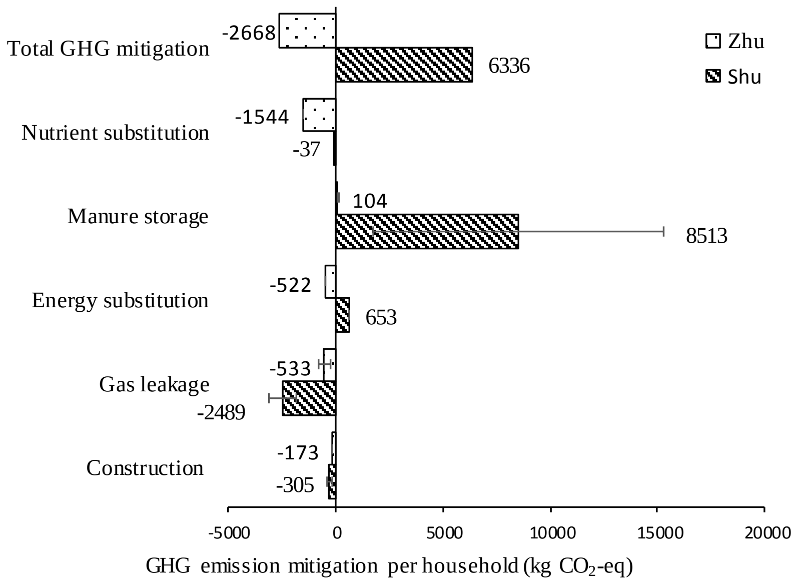

Figure 3. Annual GHG mitigation of household biogas in two villages. Positive (+) and negative (−) values represent GHG mitigation and net emission, respectively.

3.2. GHG Emissions from Construction

Annualized GHG emissions from construction were calculated based on embodied emissions from raw material inputs divided by the service life of the tank—equating to 173 and 305 kg·CO_2-eq per household and year in Zhu and Shu respectively (Table 2). The quantities of cement and brick used, as well as brick sizes, differed between the two study villages, resulting in different GHG emissions. Typical of northern Chinese villages, RHB construction in Zhu requires 200 large size bricks (400 × 200 × 200 mm) and 0.7 t cement per system; RHB construction in Shu requires 1200 small size bricks (240 × 115 × 65 mm) and 1.25 t cement per system. Longer service lives reduce annual average GHG emissions. Household survey data indicate that the average service lives are 1.0 year in Zhu and 7.6 years in Shu. One third of RHB systems in Shu (n = 95) exceeded their design service lives, whilst none did in Zhu (n = 78).

Table 2. Annualized GHG emissions from digester construction, expressed per household, and key data inputs used to calculate this compared against survey data on RHB operating life spans (up to the survey date).

Site	[a] Cumulative GHG Emission over Service Life (kg·CO_2-eq)	[b] Designed Service Life (Years)	[a] Annual GHG Emission with Service Life (kg·CO_2-eq)	[c] Actual Service Life of RHB: Share of Plants Operated According to Real Using Life (RF, Years) (%)			
				RF ≤ 1	1 < RF ≤ 5	5 < RF ≤ 10	10 < RF ≤ 20
Zhu	862.3	5	172.5 ± 0	97.4	2.6	0	0
Shu	2146.8	10	305 ± 140	15.8	51.6	11.6	21.1

Note: Source: the authors. Data are presented as means ± SD. [a] indicates 15 samples in both Zhu and Shu. [b] indicates designed service life in all biogas samples in Zhu or Shu (including 15 in Zhu and 15 in Shu). [c] indicates 78 and 95 biogas plants respectively in Zhu and Shu.

3.3. GHG Mitigation from Manure Storage

The magnitude of GHG mitigation resulting from avoided storage of animal waste depends on the amount of waste diverted into the RHB system, emission factors for traditional storage methods (see Section 1.3 in Supplementary Materials). Results showed that RHB decreased GHG emissions by 104 and 8513 kg·CO_2-eq per household and year in Zhu and Shu, respectively (Figure 3). There were three main factors leading to the great differences in GHG emission. First was different sources of animal waste (cow manure in Zhu and pig manure in Shu). Second, the amount of animal manure put into RHB was different in two villages, i.e., 2.99 t in Zhu and 5.20 t in Shu (Table 3). In Shu, each household produced 5.6 Mg of manure per year on average, of which 95% were put into RHB and 5% entered uncovered anaerobic lagoon (Table S12). Third, the traditional manure treatment methods differ between the two villages, with manure stacks in Zhu and uncovered anaerobic lagoons in Shu. The methane conversion factors (MCF) in these two systems vary greatly, at 2% and 77%, respectively (Table S2), leading to CH_4 emission factors of 0.19 $m^3 \cdot kg^{-1}$ for pig manure and 0.42 $m^3 \cdot kg^{-1}$ for cow slurry, respectively. Consequently, the RHB system only had a small GHG mitigation effect for manure storage in Zhu.

Table 3. GHG reduction from manure input for biogas over a 12-month period, per household.

Site	Treatment	Total Manure (Mg, FM)	Manure Affected by RHB (Mg, FM)	Main Storage Style	CH_4 (kg·CO_2-eq)	N_2O (kg·CO_2-eq)	GHG Emission Mitigation (kg·CO_2-eq)
Zhu	Biogas	69.3 ± 43.6	2.99 ± 0.79	RHB	0	0	104 ± 27
	Non-biogas	38.8 ± 37.4	-	Stack	46 ± 12	58 ± 15	-
Shu	Biogas	5.6 ± 5.0	5.2 ± 4.1	RHB	0	0	8513 ± 6778
	Non-biogas	4.9 ± 5.5	-	Lagoon	8437 ± 6718	75 ± 60	-

Note: Source: the authors. Data are presented as means ± SD.

3.4. GHG Mitigation from Energy Substitution

The monitoring results for biogas use in the 15 households with gas meters showed that each household used 47 m^3 (20–77 m^3) biogas on average in Zhu, and 173 m^3 (102–341 m^3) in Shu. These biogas volumes are equivalent to 33 and 123 kg of standard coal (kg coal equivalent, kg·ce) in Zhu and Shu respectively (Table 4). If the efficiency of biogas stove is 60%, then the effective heat provided by biogas would equate to between 20 and 74 kg of coal, respectively (Table S14). These quantities of energy and small compared with overall household energy use, and the use of traditional fuels was actually higher in biogas than non-biogas households in Zhu (Table 4). The type of fuels substituted also differed between the two villages. In Shu, biogas appeared to substitute straw and fuel woods, presumably owing to the labor intensity of collecting these fuels and the air pollution caused by burning these fuels indoors (Table 4, Tables S3 and S4), whilst there is no obvious substitution effect in Zhu. Consequently, households with RHB systems had GHG emissions from energy generation that were 522 kg·CO_2-eq per household and year higher than for households without RHB systems in Zhu. In Shu, RHB systems appeared to reduce energy-related GHG emissions by 653 kg·CO_2-eq per household and year (Table 4).

Table 4. Per household energy consumption and associated GHG emissions in the two study villages, expressed per household over a 12-month period.

Energy Type	Zhu Village				Shu Village			
	Primary Energy Consumption (kg·ce)		GHG Emission (kg·CO_2-eq)		Primary Energy Consumption (kg·ce)		GHG Emission (kg·CO_2-eq)	
	Biogas (n = 78)	Non-Biogas (n = 153)	Biogas (n = 78)	Non-Biogas (n = 153)	Biogas (n = 95)	Non-Biogas (n = 150)	Biogas (n = 95)	Non-Biogas (n = 150)
Electricity	122 ± 58	100 ± 47	1293 ± 610	1061 ± 497	98 ± 48	75 ± 49	1033 ± 505	794 ± 514
LPG	51 ± 35	52 ± 32	13 ± 3	14 ± 8	-	-	-	-
Straw	-	-	-	-	386 ± 363 *	481 ± 227	1828 ± 1717 *	2277 ± 1074
Firewood	-	-	-	-	194 ± 183 *	311 ± 166	1005 ± 947 **	1605 ± 859
Coal	2052 ± 663	1961 ± 746	5556 ± 1795	5308 ± 2020	-	-	-	-
Biogas	33 ± 15	-	43 ± 19	-	123 ± 52	-	157 ± 67	-
Total	2258	2113	6905	6383	801	867	4023	4676

Note: Source: the authors. Data are presented as means ± SD, and "-" means does not exist in the energy structure of the village, * (**) indicates $p < 0.05$ (0.01) between biogas and non-biogas farmers with independent samples t test.

3.5. GHG Mitigation from Nutrient Substitution

Compared to the non-biogas households, the biogas households in Zhu and Shu both increased their nutrient inputs, which increased GHG emissions from nutrient applications by 1544 and 37 kg·CO_2-eq per household and year, respectively (Table 5), with similar crop yields ($p > 0.05$) (Table S16).

Table 5. Cropland nutrient applications for biogas and non-biogas households in two villages, expressed per household over a 12-month period.

Nutrient Source	Nutrient Type	Zhu		Shu	
		Biogas (n = 78)	Non-Biogas (n = 153)	Biogas (n = 95)	Non-Biogas (n = 150)
Manure (kg)	N	165 ± 108	119 ± 105	-	34 ± 38
	P_2O_5	46 ± 30	33 ± 29	-	44 ± 49
	K_2O	56 ± 37	40 ± 39	-	19 ± 21
Digestate # (kg)	N	17.6 ± 0.0	-	33.5 ± 6.6	-
	P_2O_5	2.7 ± 0.0	-	39.0 ± 7.1	-
	K_2O	9.5 ± 0.0	-	18.5 ± 4.5	-
Straw (kg)	N	3 ± 10	1 ± 3	6 ± 16 *	3 ± 7
	P_2O_5	2 ± 5	1 ± 2	6 ± 10 **	3 ± 4
	K_2O	9 ± 12	6 ± 12	15 ± 39 **	5 ± 12

Table 5. *Cont.*

Nutrient Source	Nutrient Type	Zhu		Shu	
		Biogas (n = 78)	**Non-Biogas (n = 153)**	**Biogas (n = 95)**	**Non-Biogas (n = 150)**
Chemical fertilizer (kg)	N	608 ± 393	501 ± 247	99 ± 45	98 ± 58
	P_2O_5	245 ± 150 **	180 ± 85	39 ± 21	34 ± 21
	K_2O	250 ± 144 **	191 ± 91	2 ± 8	1 ± 5
Total nutrient input (kg)	N	794 ± 393 **	621 ± 247	139 ± 45	135 ± 58
	P_2O_5	295 ± 150 **	214 ± 85	84 ± 21	81 ± 21
	K_2O	325 ± 144 **	237 ± 91	36 ± 8 **	25 ± 5
Total GHG emission (kg·CO_2-eq)		6999 ± 3404 **	5455 ± 2135	1237 ± 378	1200 ± 485

Note: Source: the authors. Data are presented as means ± SD, and '-' means does not exist in the energy structure of the village, * (**) means $p < 0.05$ (0.01) between biogas and non-biogas farmers with independent samples t test. # means the sample number, and the value of biogas and non-biogas farmers in Zhu and Shu is 15.

Digestate use has no effect on chemical fertilizer application in either village (Table 5), perhaps reflecting loss of nutrients in the digesters (Table S7) and a preference for convenient chemical fertilizers in biogas households that usually have a higher income than non-biogas households (Table S13). Biogas substitution of straw as a fuel in Shu results in higher rates of return of straws back to arable soils (Tables S17 and S18).

3.6. GHG Emissions from Biogas Leakage and Loss

Biogas leakage caused an average of 533 and 2489 kg·CO_2-eq emission per household per year in Zhu and Shu, respectively (Figure 3). Each RHB could theoretically produce 76 and 298 m^3 biogas in Zhu and Shu, respectively (Figure 4c). Nevertheless, the actual annual consumption volumes of biogas were 47 and 176 m^3 in Zhu and Shu respectively (accounting for 61% and 59% of the theoretical biogas production volumes, respectively, Figure 4a). We considered the remaining biogas to be lost to the atmosphere as direct leakage (Figure 4c, Equation (S5) in Supplementary Materials).

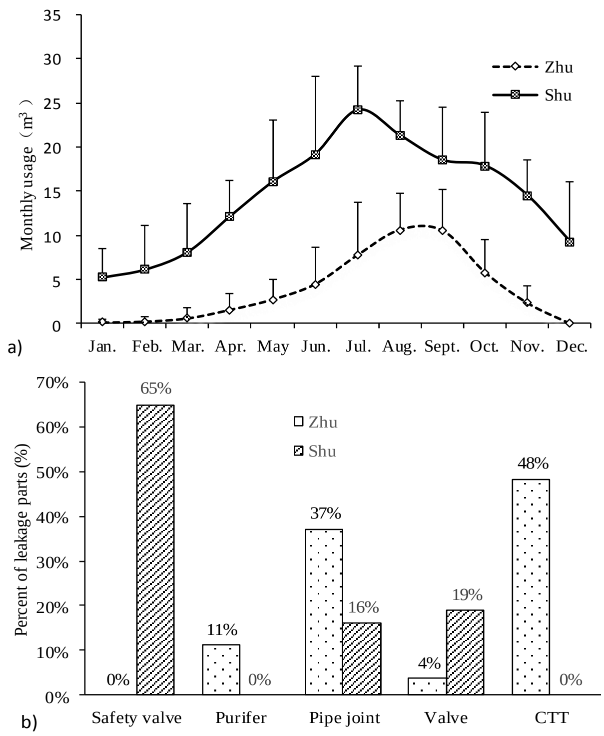

Figure 4. *Cont.*

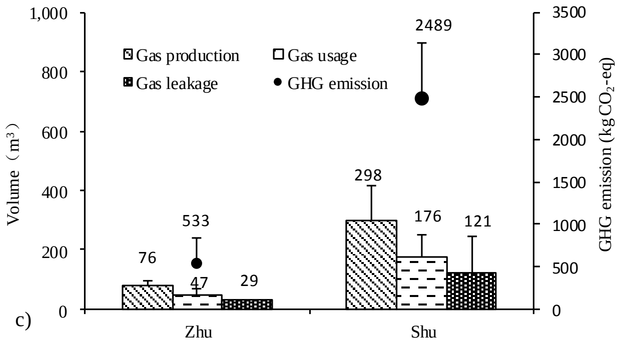

Figure 4. Gas usage by month (**a**) and leakage of household biogas by location (**b**) and by volume (**c**) in the two study villages during a 12-month period. Notes: CTT indicates connecter in tank top; the bars indirate Standard Deviation; 13 biogas household in Zhu and Shu were monitored.

Biogas leakage occurs in many parts of RHB systems (Figure 1). However, measurement data demonstrate that the positions and reasons of gas leakage in two villages were different. In Zhu, biogas leakage occurred at 48% of CTT and 37% of pipeline joint measurements, apparently due to poor quality materials. In Shu, 65% of households deliberately discharged biogas for safety reasons (to avoid excess tank pressure). There is a strong positive relationship between temperature and biogas production [20]. The largest monthly biogas usage of 24 m^3 arose in Jul in Shu, coinciding with the highest rate of gas leakage (Figure 4a and questionnaire). In the absence of biogas storage tanks, farmers discharged surplus biogas to avoid the risk of explosion (Figure 4b). We also found that biogas leakage occurred during 19% of valve and 16% of pipe joint detections in Shu systems. These numbers were lower than those in Zhu, perhaps because there was a biogas service station in Shu.

4. Discussion

4.1. GHG Mitigation Effectiveness of RHB

Application of a consequential LCA approach to measurement and survey data on RHB operations and overall energy use, fertilizer use and manure storage in farm households with and without RHB systems showed significant differences in the apparent GHG mitigation effectiveness of RHB between two Chinese villages. RHB systems increased farm household GHG emissions by 2668 kg·CO_2-eq·year^{-1} in north China, but reduced farm household GHG emissions by 6336 kg·CO_2-eq·year^{-1} in south China. These results contrast with previous studies that have tended to indicate large GHG mitigation from RHB systems, but that typically considered only a few of the farm household processes altered by the introduction of RHB systems [5,11]. We demonstrated the importance of capturing key factors such as biogas service life, biogas leakage rate, nutrient substitution and manure storage within LCA system boundaries—in addition to the more typical emphasis on energy substitution rate [11,17]. Ideally, site-specific data are required to reflect large differences in incurred and avoided management practices, and biogas yield potential.

GHG emissions from RHB construction equated to 860–2147 kg·CO_2-eq per household and year (Table 2), which is considerably lower than construction emissions reported for medium and large biogas digesters—e.g., 112 Mg·CO_2-eq according to Yabe [18]. However, when these emissions are normalized per m^3 of biogas over the relatively short service life of RHB systems, Chinese RHB systems emit approximately 3.6 kg·CO_2-eq per m^3 biogas, compared with 0.21 CO_2-eq per m^3 biogas for small-scale plastic biogas units, and 2.4 kg·CO_2-eq per m^3 for large biogas digesters in Latin America [29]. In Zhu and Shu, RHB systems are designed to operate for between 5 and 10 years (Table 2), compared with 20 years for Latin American systems [29].

Biogas leakage is another hotspot that is often neglected [30]. Previous studies have tried to monitor and quantify biogas leakage [31,32], which IPCC [33] highlights can range from 0% to 100%

of total production. Bruun et al. [9] estimated that the volume of biogas leakage in China overall represented 40% of biogas production, using literature data. Dhingra et al. [34] proposed that Chinese RHB leakage amounted to only 4.5 m^3 biogas per household per year, based on path-integrated concentration measurements and VRPM technology. These rough estimations ignored artificial biogas discharge by farmers and did not clarify where the leakage occur. Quantification of these factors within this study provides an important evidence base to dramatically improve the performance of small-scale biogas systems in China.

Because RHB systems are constructed underground, it is impossible to install measuring instruments directly [32]. In addition, since the rural households also purposely discharge surplus biogas without a pre-defined time schedule mainly to respond to potential safety risks (Section 3.6), it is very challenging to quantify the volume of biogas leakage and loss. In this study, we quantified biogas leakage by estimating the difference between the production volume and the actual consumption volume of biogas. The ABEPE model, used to estimate biogas production, has been validated in previous studies, including Tang et al. [21], Batzias [35], and Jingura et al. [36].

Adequate characterization of the background (or reference system), including all counterfactual processes, is critical to accurately estimate net GHG mitigation using a consequential LCA approach (Figure 2). Wide variation in farm practices across regions means that it is imperative to obtain farm-, or at least regionally-, specific data on counterfactual management processes such as manure handling. We found the real amounts of cow and manure treated by RHB systems were 2990 and 5200 kg in Zhu and Shu, respectively, and that the farmers' traditional manure storage methods of stacking and uncovered anaerobic lagoons led to vastly different GHG mitigation attributable to avoided manure storage in the study villages. These values differed considerably from default values used in previous studies [11,27]. We found that use of digestate from RHB systems does not reduce synthetic fertilizer use, in contrast with previous research indicating a high substitution potential [1,27]. Biogas usage in the two villages was only 47 to 176 m^3 per household and year, which is far below that estimated in previous studies such as Liu et al. [11], and appears to predominantly replace firewood and crop straw (in Shu) rather than fossil fuels as indicated in previous research [4,5,17]. Consequently, energy substitution at best makes only a small contribution to GHG mitigation compared with avoided manure management, confirming results of previous slurry biogas LCA studies in Europe [24–26].

4.2. Factors Affecting Development of RHB

Over the study period (2010–2014), the number of RHB systems in operation increased slightly in Shu but decreased dramatically in Zhu, from 93 to 25 (Table S1). Our analyses reveal some important barriers for RHB adoption, confirming previous studies [14], such as the quality of digester construction [34] and biogas technology [15], ease of operation and maintenance [9] and socio-economic barriers such as level of education and household finances [19]. Based on our findings, we suggest that more efficient deployment of RHB systems requires careful evaluation of the local climate (biogas yield) and farm management practices (manure quantities and existing storage systems), and adequate local service support. We make the following recommendations to increase the viability and GHG mitigation effectiveness of small-scale biogas systems in China.

It is necessary to re-design RHB systems for operation in colder climates. Current common RHB technology cannot ensure biogas production at a temperature below 10 °C [37]. One of the main reasons for successful deployment of RHB system in Shu was the adequate temperature (above 10 °C for 83% of days in a year) (Figure 4a). In other parts of the world such as Vietnam, RHB systems also work successfully owing to adequate climatic temperatures [38]. In regions with cold winters such as Zhu (only 58% of days in the year above 10 °C), low temperature appears to impede efficient operation of RHB systems and leads to abandonment (Figure S2). Adaptations that could improve the performance of RHB systems in cold climates include insulating and/or heating the biogas digesters—though this would require good biogas yields to justify the investment.

In future, it may be more appropriate to install fewer, larger biogas systems to accommodate the shift towards larger, more intensive farms as Chinese farming consolidates. Current RHB systems are limited in size (~8 m^3) and can only treat small quantities of wastes. Farmers also expressed a reluctance to transport manure back to small RHB systems from nearby fields, especially as labor is getting more scarce in rural areas—manure stacking is the preferred management option for this reason (Figure S2). Nonetheless, the consolidation of livestock production in larger intensive farms is likely to improve the economic viability, and potentially the efficiency of, biogas systems. A recent UK study found that larger biogas plants were more efficient and achieved larger biogas yields [26].

Our third recommendation is to develop infrastructure and markets that enable biogas production to be matched with household energy requirements [6]. The small digester size and low operating efficiency lead to unstable energy supply from RHB systems, i.e., oversupply in summer and insufficient supply in winter. The installation of biogas storage tanks and small electricity generators could avoid deliberate discharge of biogas (as in Shu) and ensure that more fossil energy is substituted. In some areas, larger digesters may be appropriate, as described above. But RHB is also well suited to remote mountainous areas where main fuels are straws and fuel woods that can be efficiently replaced with biogas, also leading to health benefits. Insulated or heated biogas tanks may be needed.

Our final recommendation is to design servicing packages that balance cost to householders with benefits via increased biogas use, whilst maintaining profitability for service companies. Although small-scale RHB construction is attractive in terms of small initial investment (3000 yuan) and potential economic returns (Shu), maintenance, repair and operation requires substantial time and money. Monitored leakage rates highlighted a need for frequent inspection and repair (about 50 RMB per RHB per year). Digestate needs to be extracted one to three times per year at a cost of 60 RMB per time. In regions without service stations, farmers must seek service stations in distant counties or cities, greatly increasing costs (Zhu). Even in Shu where there was a local service company, poor service was still a primary reason for abandoning RHB systems (Figure S2), suggesting that servicing is a weak link. Subsidies may be necessary to encourage servicing, as well as construction.

5. Conclusions

RHB can be an effective measure to reduce the fossil fuel consumption, GHG emissions from manure storage, and chemical fertilizer inputs. However, poorly operated RHB systems can increase GHG emissions, aggravate nutrient surpluses on farmland, increase labor input and cause economic loss. Low temperature is the primary reason for poor performance and abandonment of RHB systems in North China, while lack of service support for maintenance is a primary reason for abandonment in Southern China. The design of RHB systems needs to be carefully matched to local conditions (e.g., insulated in cold climates) and farmers' needs with respect to manure management and energy requirements. Technical options such as small pumps to draw out digestate from digesters could improve operational efficiency and reduce labor requirements for RHB systems. There is certainly a continuing role for RHB systems to serve small subsistence farms, but biogas systems are likely to become more efficient and profitable as they are deployed in smaller numbers of larger livestock farms as livestock farming in China intensifies. Government support for the deployment of biogas storage equipment and micro power generators could significantly improve household energy substitution, helping to correct the spatial and temporal disconnect between biogas supply and demand, and also reduce the biogas discharge that is so detrimental to net GHG mitigation. Additional support for RHB servicing could also significantly improve operating efficiency, GHG mitigation and economic viability. Finally, widespread adoption of more precise nutrient management planning would ensure that digestates are utilized efficiently to realize potential fertilizer substitution effects.

Supplementary Materials: The followings are available online at www.mdpi.com/1996-1073/10/2/239/s1, Figure S1: Increases in biogas digester units (cumulative) and government support provided as subsidies (cumulative), Figure S2: Reasons for abandoning RHB, including abandoning construction or an operation RHB. Percentage of reasons exceeded 100% because some farmers gave more than one reason for stopping a RHB,

Table S1: The numbers of biogas and non-biogas farms and main categories in the survey, Table S2: GHG emission factors for manure management, Table S3: Calculation parameters of energy consumption, Table S4: GHG emission factor of energy consumption, Table S5: GHG emission factors of energy extraction and transportation, Table S6: Contents of nutrients in manures, digestate and wastes, Table S7: Nutrient retention of biogas digesters in the two villages, Table S8: GHG emission during chemical fertilizer production and transportation, Table S9: The estimated biogas yields of households, Table S10: Cropland and livestock, Table S11: Nutrient content of different crop straws, Table S12: Manure treatment in Zhu and Shu in 2009/2010, Table S13: Household income and average years of education completed by the head of the household, Table S14: Energy consumption by source in biogas and non-biogas households in the two villages in 2009/10, Table S15: Estimated energy consumption of all households including biogas and non-biogas farmers in 2009/10, Table S16: Fertilization of biogas and non-biogas household in two villages in 2009/10, Table S17: Straw use in different ways of the two groups of farmers in 2009/10, Table S18: Straw nutrient content and use in 2009/10.

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