Journal of Cleaner Production 309 (2021) 127343

Contents lists available at ScienceDirect

Journal of Cleaner Production

journal homepage: www.elsevier.com/locate/jclepro

# Mitigating environmental impacts of milk production via integrated maize silage planting and dairy cow breeding system: A case study in China

Xianlei Huang [a,b], Boyang Shi [a,c], Shu Wang [a,d], Changbin Yin [a,b,*], Linna Fang [a,b,**]

[a] Institute of Agricultural Resources and Regional Planning, Chinese Academy of Agricultural Sciences, Beijing, 100081, China
[b] Research Center for Agricultural Green Development in China, Beijing, 100081, China
[c] Laboratory of Chemical Engineering, Department of Applied Chemistry, University of Liège, Liège, Belgium
[d] Laboratory of Biomass and Green Technologies, Gembloux Agro-Bio Tech, University of Liège, 5030, Gembloux, Belgium

ARTICLE INFO

Handling editor: Cecilia Maria Villas Bôas de Almeida

*Keywords:*
Integrated maize silage planting and dairy cow breeding system (IPBS)
Life cycle assessment (LCA)
Environmental impacts
Milk production
Dairy farm

ABSTRACT

Environmental impacts of milk production are depending on the production efficiency of livestock and cropland. A mode of integrated maize silage planting and dairy breeding system (IPBS) has been widely promoted in China, as a promising way to recycle manure, reduce chemical fertilizer consumption and improve soil quality. However, quantitative environmental impacts and mitigation potential of this system remains unclear. In this study, based on life cycle assessment (LCA), environmental performance of non-IPBS and IPBS were compared: non-IPBS only involved dairy cow breeding, whereas maize silage planting was incorporated in IPBS. Results indicated that, although 60% of the surveyed dairy farms adopted IPBS, the self-sufficiency rate of maize silage was 57%. Compared with non-IPBS, IPBS had apparent potential in reducing global warming potential (−14%), acidification potential (−10%), eutrophication potential (−18%), non-renewable energy use (−10%), water use (−8%) and land use (−13%). It is estimated that, in China, 81% of dairy farms could adopt IPBS, resulting in a reduction of approximately 21% in greenhouse gas (GHG) emissions to compared with current situation, but the premise is that 2.0 million ha cropland should be applied for maize silage cultivation. Interestingly, environmental performance of IPBS was affected by the self-sufficiency rate of maize silage and restricted by milk yield and maize silage yield. Thus, mitigation of environmental impacts of milk production could be realized by combining a short-term strategy of increasing maize silage planting area in dairy farms and a long-term plan for technological improvements in the yield of crop and milk.

## 1. Introduction

From a global perspective, livestock contributes half of global agricultural gross domestic product (GDP) and it is responsible for economic benefits to at least 1.3 billion producers and retailers (Herrero et al., 2016; Thornton, 2010). On the other side, it is one of the foremost contributors to global resource consumption and gas emissions, because it occupies 33% of global cropland for animal fodder production, consumes 32% of global agricultural water, contributes 15% of global greenhouse gas (GHG) emissions, and produces over 20% of $NH_3$ emissions (Herrero et al., 2016; Wei et al., 2018). As global demand for ruminant meat and dairy products grows, GHG and $NH_3$ emissions from livestock will be expected to double by 2050 (Du et al., 2018), which indicates that environmental performance and resource utilization efficiency of livestock production must be enhanced to ensure the sustainable and cleaner livestock production in the future.

Though livestock production undertakes part of the responsibility for global GHG and $NH_3$ emissions, it has potential for improvement, especially in dairy industry. In milk production system, the sources of gas emissions can be divided into direct and indirect emissions (Kimberly et al., 2011): 1)direct emissions refer to gases produced by cows, including enteric fermentation and volatilization from excreted manure; 2)indirect emissions are gases that are either produced in fodder production or generated as a result of energy consumption during dairy cow breeding and the transportation of fodder and manure. Obviously, direct and indirect emissions are interconnected and are dependent on the production efficiency of livestock and cropland. An integrated maize silage planting and dairy cow breeding system (IPBS, for abbreviations

---

* Corresponding author. Institute of Agricultural Resources and Regional Planning, Chinese Academy of Agricultural Sciences, Beijing, 100081, China.
** Corresponding author. Institute of Agricultural Resources and Regional Planning, Chinese Academy of Agricultural Sciences, Beijing, 100081, China.
*E-mail addresses:* hxlzhg@sina.cn (X. Huang), yinchangbin@caas.cn (C. Yin), fanglinna@caas.cn (L. Fang).

https://doi.org/10.1016/j.jclepro.2021.127343
Received 16 February 2021; Received in revised form 25 April 2021; Accepted 28 April 2021

used in this study see Table 1) substantially increases the recycling rate of manure and maize silage, while simultaneously lowering chemical fertilizer consumption for maize silage planting and energy consumption in fodder and manure transportation (Fig. 1). These all have a positive effect on both environmental production and resource conservation.

In China, the high-speed development of intensive animal husbandry results in the massive construction of dairy farms. There are two typical milk production systems in China: Only dairy cow breeding system (non-IPBS) and IPBS (Fig. 1). During the feeding phase of non-IPBS, all of the maize silage is purchased, which raises the cost of fodder and energy consumption in transportation. Moreover, substantial amounts of dairy cow manure are wasted in this system. For the fodder planting phase, farmers generally prefer to use chemical fertilizer rather than organic fertilizer (Zhang et al., 2019, 2020), while recycling manure applied to cropland is a challenge. Thus, the decoupling of livestock and cropland is occurring, resulting in considerable amounts of manure and straw waste, which consequently leads to marine, terrestrial and atmospheric pollution (Jin et al., 2020). On the contrary, IPBS has been widely promoted in China as a promising way to realize the recycling of manure to fields, reduction of chemical fertilizer consumption and improvement of soil quality by manure application. Since 2017, the Chinese government has launched a program named "planting crops for feed instead of food" in 17 provinces (MOA, 2017), particularly for maize silage, to ensure a sufficient high-quality roughage and maximized milk yield. Generally, IPBS is defined as an agricultural system that is designed to be restorative and regenerative between livestock and cropping (Winans et al., 2017; Yin et al., 2013). Materials and substances flow through IPBS estimates to be more efficient.

Although the technical advantages of IPBS are widely recognized, there is still a lack of information on environmental performance. Life cycle assessment (LCA) is a method to quantitatively estimate environmental impacts of products, processes and activities (Michael et al., 2016). For milk production system, it has significant impacts on GHG emissions and energy consumption (Herrero et al., 2016; Wang et al., 2016, 2018), as well as on eutrophication, acidification, photochemical oxidant formation, land and water scarcity (Baldini et al., 2017; Battini et al., 2014; Eshel et al., 2014). In Europe, the self-sufficiency rate of fodder exceeds 70%, and this high rate has led to a reduction in GHG emissions, acidification and energy consumption (Matteo et al., 2013). Maize silage cultivation by dairy farm mitigates global warming, eutrophication, acidification and land use in New Zealand (Basset-Mens et al., 2009). The fodder production, enteric fermentation and manure management are the three major GHG emission sources in milk production system (Del et al., 2013; Wang et al., 2016). Fodder production is a significant determinant of global warming, energy use, land use and water consumption, while eutrophication and acidification are mainly affected by manure management (Wang et al., 2018). Besides GHG emission mitigation (Awasthi et al., 2019; Nkoa, 2014), IPBS has the additional advantages of reducing odors and water contamination by manure digestion, as well as improving crop residue utilization efficiency and soil fertility by manure application (Wei et al., 2000; Zheng et al., 2012).

Therefore, evaluation of the environmental performance of milk production based on the whole system perspective is needed. Relevant research showed that environmental impacts were mainly dependent on the production efficiency of livestock and cropland (Basset-Mens et al., 2009; Fan et al., 2018; Zucali et al., 2018). Only by reducing environmental impacts from all processes, including fodder production, milk production and transportation, can the goal of low carbon production in dairy industry be achieved. IPBS is considered to have high potential in reduction of environmental impacts. However, existing studies paid insufficient attention to this work. An in-depth and accurate comparison of environmental performance between IPBS and non-IPBS in milk production system should be considered. Hence, based on life cycle assessment (LCA), this study compared the environmental performance of non-IPBS and IPBS in Shandong and Heilongjiang provinces in China, non-IPBS only contained dairy breeding, whereas maize silage planting was incorporated in IPBS. This study is for: 1)what are the environmental performances in milk production system under non-IPBS and IPBS; 2)what are the determinants of environmental performance in IPBS; 3)what are the potentials to reduce GHG emissions through IPBS. The practical significance of this study is selecting optimal technologies with lower environmental impacts, which is crucial for the proposes of relevant policies to support the government in promoting the low-carbon development of dairy industry.

**Table 1**
Main abbreviations used in this study.

| Abbreviation | Explanation |
|---|---|
| IPBS | Integrated maize silage planting and dairy cow breeding system |
| non-IPBS | Only dairy cow breeding system |
| LCA | Life cycle assessment |
| FPCM | Fat and protein corrected milk |
| GWP | Global warming potential |
| EP | Eutrophication potential |
| AP | Acidification potential |
| NREU | Non-renewable energy use |
| WU | Water use |
| LU | Land use |
| OLS | Ordinary least square |
| CNY | Chinese yuan, 1 CNY = 0.1534 USD (December 31, 2020) |

## 2. Materials and methods

### 2.1. IPBS in milk production system in study region

Milk production regions are mainly distributed in northern China, as well as the main maize cultivated area (Fig. 2). In 2019, China produced about 32 million tons of milk, with about 11 million dairy cows and 41 million ha of maize cropland (NBSC, 2020), of which about 5% was planted for maize silage. In 2019, Shandong and Heilongjiang province produced about 2.2 million tons and 4.7 million tons of milk, accounting for 7% and 15% of the total milk in China, respectively. There were 3.8 million ha of maize cropland in Shandong, and 5.9 million ha in Heilongjiang (NBSC, 2020). In terms of the policy, since 2016, two provinces have implemented a subsidy policy of maize silage planting, with a subsidy level of 20–50 CNY/t in Shandong and 60–100 CNY/t in Heilongjiang (DARH, 2020; SPG, 2016). By the end of 2020, the planting area of maize silage in Shandong and Heilongjiang have been over 333 and 46 thousand ha, respectively (DARH, 2020; SPG, 2016).

Thus, we conducted a dairy farm survey in Shandong and Heilongjiang provinces, including 223 dairy farms (i.e. 109 in Qingdao, Shandong and 114 in Harbin, Heilongjiang) via a face-to-face interview from August to December in 2020. A total of 189 valid dairy farms' data were obtained. There were mainly two milk production systems (non-IPBS and IPBS) in the study region, even across the country (Fig. 3). In non-IPBS, manure was composted into organic fertilizer, and was mainly consumed in orchards or vegetable fields. In IPBS, maize silage planting was incorporated, manure was composted into organic fertilizer, and then applied to maize silage cropland.

Table 2 presents basic information for various dairy farms. Milk yield for per lactating cow in non-IPBS and IPBS were 8372 and 8493 kg/year, individually. Although IPBS had a higher milk yield compared with non-IPBS, there was no significant difference after the T-test. Milk yield in Shandong was significantly higher than that in Heilongjiang, the reason might be the different diets (Table 4). Maize silage yield in IPBS was 43995 kg/ha, and the yield in Heilongjiang was significantly higher than in Shandong. An overview of the surveyed 189 dairy farms, 60.3% adopted IPBS with about 56.6% of self-sufficiency rate of maize silage.

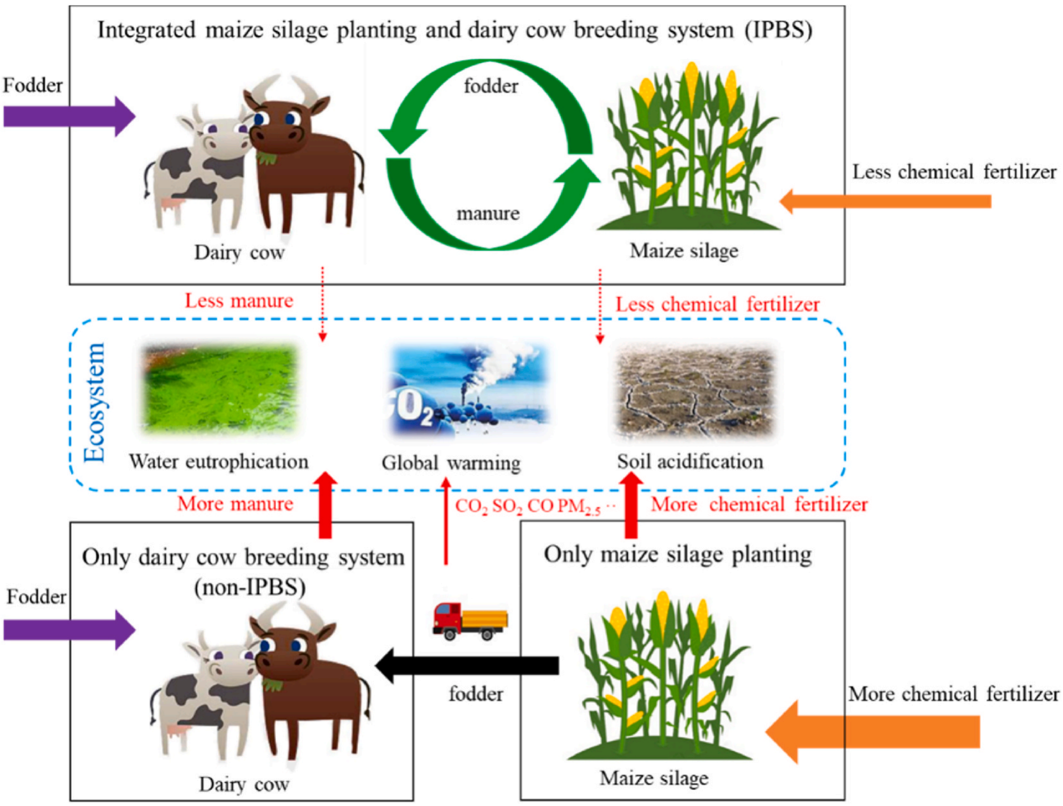


**Fig. 1.** Two typical milk production systems in China (IPBS: high recycling rate, low environmental impacts; non-IPBS: low recycling rate, high environmental impacts).

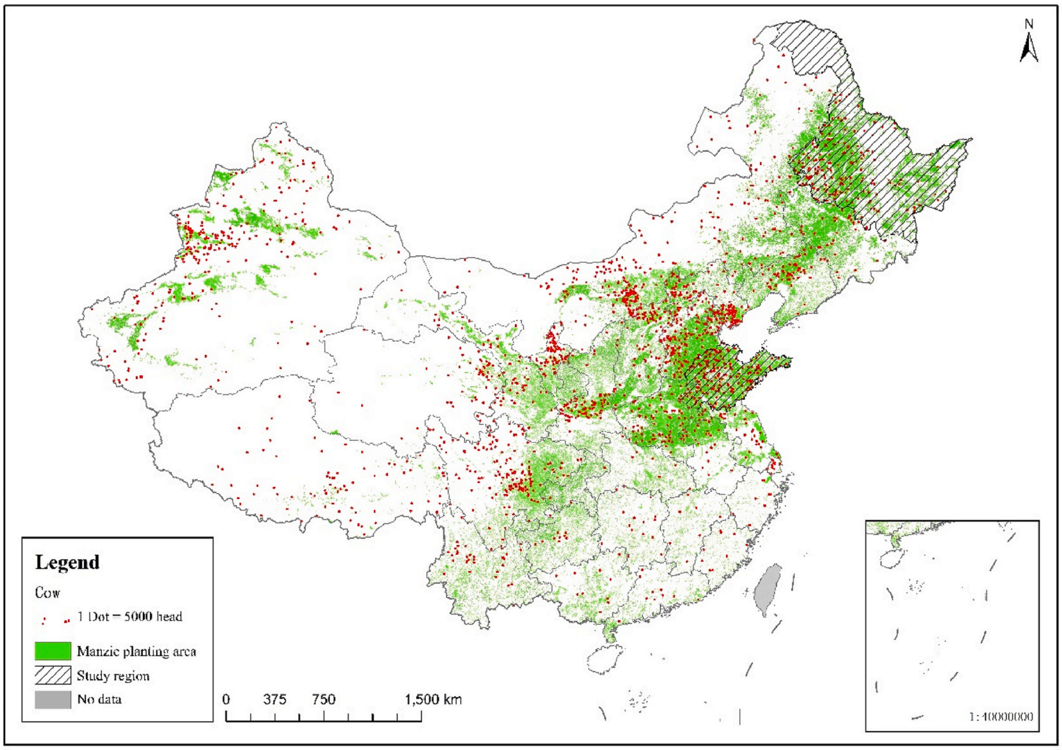


**Fig. 2.** Study region, cow distribution and maize planting area in China.

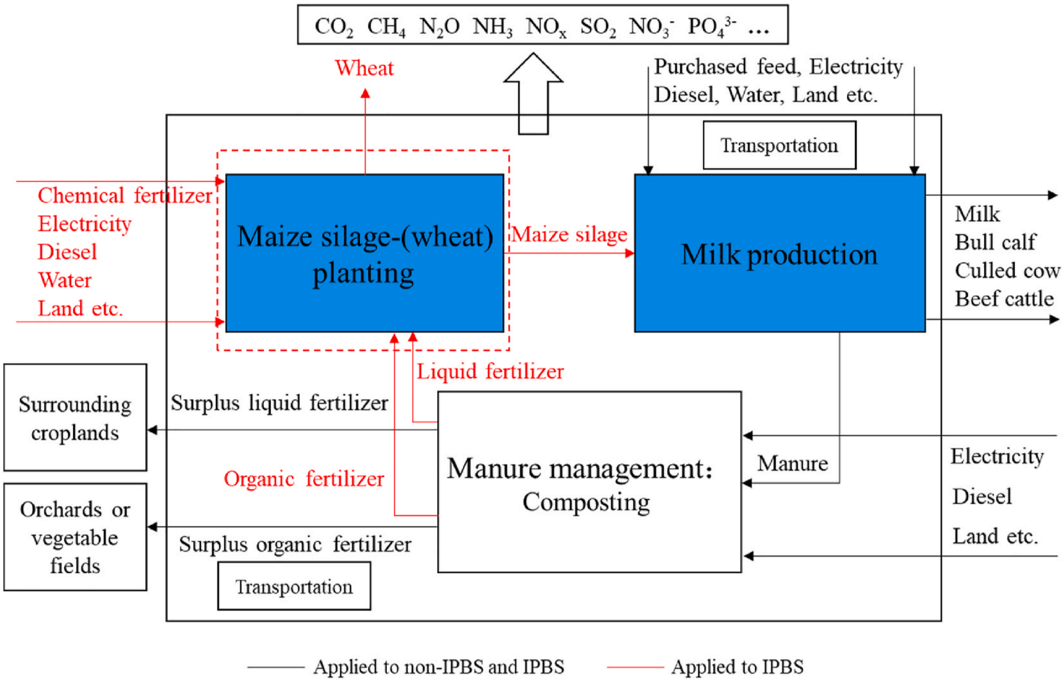


**Fig. 3.** System boundaries for IPBS and non-IPBS.

**Table 2**
The basic information of various dairy farms.

| | non-IPBS | | | IPBS | | |
|---|---|---|---|---|---|---|
| | ShanD | HeiLJ | Total | ShanD | HeiLJ | Total |
| Lactating cow | 81.4 | 109.2 | 95.1 | 129.6 | 128.9 | 129.2 |
| Non lactating cow | 24.5 | 51.0 | 37.6 | 30.8 | 57.0 | 46.7 |
| Finishing cow | 62.2 | 72.4 | 67.2 | 102.0 | 44.4 | 67.1 |
| Calf | 28.1 | 35.4 | 31.7 | 39.7 | 37.4 | 38.3 |
| Total cow | 196.1 | 270.8 | 233.0 | 302.2 | 267.7 | 281.3 |
| Milk yield (kg/year/cow) | 8588$^{a}$ | 8173$^{b}$ | 8383 | 8781$^{a}$ | 8334$^{b}$ | 8510 |
| Price of milk (CNY/kg) | 3.63 | 3.63 | 3.63 | 3.64 | 3.66 | 3.66 |
| Total cropland (ha) | 0 | 0 | 0 | 12.97 | 49.15 | 34.87 |
| Cropland for maize silage-(wheat) (ha)* | 0 | 0 | 0 | 10.35 | 35.41 | 25.52 |
| Wheat (kg/ha)* | - | - | - | 6927 | - | - |
| Maize silage (kg/ha)* | - | - | - | 39743$^{a}$ | 46768$^{b}$ | 43995 |
| Self-sufficiency rate of maize silage (%) | 0.0 | 0.0 | 0.0 | 31.8 | 72.7 | 56.6 |
| Sample size | 38 | 37 | 75 | 45 | 69 | 114 |

Note: “ShanD” Shandong province; “HeiLJ” Heilongjiang province;“CNY” Chinese yuan, 1 CNY = 0.1534 USD (December 31, 2020); “a” and “b” mean milk yield for per lactating cow and maize silage yield in two provinces by differ significantly at the 5% level; “*” the typical crop rotation system in Shandong is maize silage-wheat, and that in Heilongjiang is only maize silage.

### 2.2. *Goal and scope definition*

LCA has been widely applied in environmental assessment of agricultural production (Baldini et al., 2017; Sara et al., 2016; Wang et al., 2018). The key to applying LCA is how to establish a linkage between the inventory of inputs for the system and its potential environmental impacts. In this study, the LCA framework and methodology were introduced according to ISO 14040 standard (ISO, 2020). This study aimed to quantify environmental impacts of milk production system under non-IPBS and IPBS in Shandong and Heilongjiang provinces.

**Table 3**
Maize silage-(wheat) planting in IPBS.

| | | ShanD | HeiLJ | Total |
|---|---|---|---|---|
| Maize silage production* | Application of fertilizer (N) (kg/ha) | 133 | 128 | 130 |
| | Application of fertilizer ($P_2O_5$) (kg/ha) | 38 | 37 | 37 |
| | Application of fertilizer ($K_2O$) (kg/ha) | 29 | 28 | 28 |
| | Application of organic fertilizer (N) (kg/ha) | 251 | 240 | 244 |
| | Electricity (kWh/ha) | 635 | 266 | 412 |
| | Diesel (L/ha)$^{\#}$ | 56 | 56 | 56 |
| | Irrigation water ($m^3$/ha) | 1270 | 532 | 823 |
| | Maize silage (kg/ha) | 39743$^{a}$ | 46768$^{b}$ | 43995 |
| Wheat production* | Application of fertilizer (N) (kg/ha) | 136 | - | - |
| | Application of fertilizer ($P_2O_5$) (kg/ha) | 39 | - | - |
| | Application of fertilizer ($K_2O$) (kg/ha) | 29 | - | - |
| | Application of organic fertilizer (N) (kg/ha) | 252 | - | - |
| | Electricity (kWh/ha) | 735 | - | - |
| | Diesel (L/ha)$^{\#}$ | 56.5 | - | - |
| | Irrigation water ($m^3$/ha) | 1470 | - | - |
| | Wheat (kg/ha) | 6927 | - | - |
| Total Cropland (ha) | | 12.97 | 49.15 | 34.87 |
| Crop land in maize silage-(wheat) production (ha)$^{\&}$ | | 10.35 | 35.41 | 25.52 |
| Sample size | | 45 | 69 | 114 |

Note: “*” the typical crop rotation system in Shandong is maize silage-wheat, and that in Heilongjiang is only maize silage; “#” Due to the agricultural machinery was rented, and farm’s manager did not know the detailed diesel consumption in fodder production, those data according to Liu (2017); “a” and “b” mean maize silage yield in two provinces by differ significantly at the 5% level; “&” Some farms only use part of own land to produce maize silage-(wheat), and the system boundaries did not include the emissions from other crops production in this study.

#### 2.2.1. *System boundary*

System boundaries were set by using a “cradle-to-gate” approach, two milk production systems were involved, i.e., non-IPBS and IPBS

**Table 4**
Milk production in different systems (average value for per farm).

| | non-IPBS | | | IPBS | | |
|---|---|---|---|---|---|---|
| | ShanD | HeiLJ | Total | ShanD | HeiLJ | Total |
| Concentrate (t/year)* | 202.5 | 357.6 | 279.0 | 317.2 | 475.0 | 412.7 |
| Maize grain (t/year) | 98.6 | 95.9 | 97.3 | 155.2 | 93.7 | 118.0 |
| Soybean meal (t/year) | 45.5 | 25.2 | 35.5 | 71.6 | 45.4 | 55.7 |
| Wheat bran (t/year) | 38.1 | 2.4 | 20.5 | 59.7 | 1.1 | 24.2 |
| Purchased maize silage (t/year) | 886.5 | 2425.6 | 1645.8 | 977.6 | 2099.3 | 1656.5 |
| Produced maize silage (t/year)# | 0.0 | 0.0 | 0.0 | 419.5 | 1620.3 | 1146.3 |
| Alfalfa (t/year) | 97.0 | 87.7 | 92.4 | 148.9 | 82.0 | 108.4 |
| Oats (t/year) | 105.2 | 36.2 | 71.2 | 153.2 | 48.1 | 89.6 |
| Leymus chinensis (t/year) | 259.1 | 62.2 | 162.0 | 396.2 | 93.1 | 212.7 |
| Cotton seed meal (t/year) | 33.9 | 4.0 | 19.1 | 53.9 | 12.9 | 29.1 |
| FMCM (t/year) | 890.1 | 1160.7 | 1023.6 | 1480.7 | 1339.8 | 1395.4 |
| Milk protein content (%) | 3.32 | 3.37 | 3.34 | 3.31 | 3.35 | 3.33 |
| Milk fat content (%) | 3.78 | 4.02 | 3.90 | 3.77 | 4.01 | 3.92 |
| Bull calf (number/year) | 10.4 | 41.1 | 25.5 | 14.2 | 52.1 | 37.1 |
| Culled cow (number/year) | 18.2 | 24.0 | 21.1 | 21.1 | 26.3 | 24.2 |
| Beef cattle (number/year) | 1.6 | 6.5 | 4.0 | 2.4 | 3.4 | 3.0 |
| Electricity (kWh/year) | 86299 | 105598 | 95820 | 147297 | 105891 | 122235 |
| Diesel (L/year) | 1943 | 3406 | 2665 | 3028 | 3246 | 3160 |
| Water ($m^3$/year) | 9079 | 11945 | 10493 | 13601 | 12419 | 12886 |
| Land occupation (ha) | 1.26 | 1.48 | 1.37 | 1.97 | 2.18 | 2.10 |
| Sample size | 38 | 37 | 75 | 45 | 69 | 114 |

Note: "*" concentrate contains 30% soybean meal, 50% maize grain, 10% wheat bran and 10% others.

(Fig. 3). In non-IPBS, the pathway can be divided into: milk production, manure management, and transportation of fodder and manure. In IPBS, maize silage planting was incorporated, the pathway can be divided into: maize silage-(wheat) planting, milk production, manure management, and transportation of fodder and manure. In both systems, vitamin supplements, medicines and bovine semen were not included, because of the low contribution to overall impacts (Battini et al., 2014; Wang et al., 2018).

### 2.2.2. *Functional unit and allocation*

The mass (kg) or volume (L) of raw milk was normally selected as a functional unit (FU). However, in order to emphasize the nutritional function of milk, the functional unit (FU) was 1 kg of fat and protein corrected milk (FPCM), which could be estimated with the following formula (Baldini et al., 2017):

$$\text{FPCM}(kg) = Prodution(kg) \times [(0.116 * Fat\%) + (0.06 * Protein\%) + 0.337] \quad (1)$$

In this study, milk was regarded as the primary production, besides wheat (IPBS in Shandong), bull calf (both systems), culled cow (both systems), beef cattle (both systems) and organic fertilizer (both systems) were possible co-products. Thus, the overall environmental impacts should be burdened among the various products of the system. There were three typical allocation rules to distribute environmental impacts between milk and co-products (Baldini et al., 2017; Battini et al., 2014; Wang et al., 2018), including No allocation rule (all impact is allocated to the milk), Mass allocation rule (impact is based on the weight of the products leaving the system) and Economic allocation rule (impact is based on the value of the product sold). Different allocation rules lead to divergent results, Economic allocation rule is widely applied in present studies (Baldini et al., 2017; Battini et al., 2014; Wang et al., 2018), which is suitable for solving the problem of great difference in prices of co-products and coservices (Ardente et al., 2012). Therefore, the economic allocation rule was introduced in this study.

## 2.3. *Inventory analysis*

### 2.3.1. *Maize silage-(wheat) planting*

The typical crop rotation mode in Shandong is maize silage-wheat, and only maize silage in Heilongjiang. The inputs mainly included chemical fertilizers, organic fertilizers, pesticides, seeds, irrigation water and energy consumption for field operation. The products were wheat, wheat straw and maize silage (Table 3). In this study, wheat was sold, wheat straw was returned to field, maize silage was applied as fodder. IPBS significantly reduced the consumption of chemical fertilizer in maize silage planting. According to National Agricultural Product Cost-Benefit Data Compilation 2019 (NDRC, 2019), for only maize planting farm, the amount of applied N in chemical fertilizer was 384 kg/ha in Shandong and 338 kg/ha in Heilongjiang, both were higher than that in this study (Table 3). Obviously, the consumption of chemical fertilizer in IPBS was different from that of only crop production farm, which was the main factor affecting field emissions (Battini et al., 2014). Emissions in this process were evaluated according to Table A1.

### 2.3.2. *Milk production*

In this process, the difference between the non-IPBS and IPBS was the source of maize silage. In IPBS, part of the maize silage was produced by farms' own cropland, while all fodder was purchased in non-IPBS. Inputs were mainly fodder, water, electricity, diesel, infrastructures and purchased cow. Products were milk, bull calf, culled cow, beef cattle and manure (Table 4). Based on our survey, fodders for dairy cows were mainly concentrate, maize grain, wheat bran, maize silage, cotton seed meal, oats, alfalfa and leymus chinensis. Only a few farms added distiller's grain and rapeseed meal. Emissions from distiller's grain and rapeseed meal were exceedingly low, which can be neglected in this study. Emissions in milk production included emissions from purchased fodder production, energy consumption (e.g. fodder processing, cowshed lighting and ventilation, water heating, milk refrigeration) and animal enteric emissions. Energy consumption emissions, purchased fodder production emissions, and animal enteric emissions were calculated according to Table A1, Table A2 and Table A3, respectively.

### 2.3.3. *Manure management*

Based on our survey, all dairy cows were Holstein. Excreted parameters of manure were demonstrated in Table A4. Various manure management leads to differences in emissions, composting was used in non-IPBS and IPBS. The inputs of this process were mainly electricity and diesel, output was organic fertilizer (Table 5). Emissions were from manure composting and energy consumption in organic fertilizer production. Emissions from manure composting were reckoned according to Table A5. Energy consumption emissions were determined according to Table A1.

### 2.3.4. *Fodder and organic fertilizer transportation*

Differences between non-IPBS and IPBS were the transport distance of maize silage and organic fertilizer in this process. In IPBS, due to the self-produce of maize silage, part of organic fertilizer was used in farm's own cropland, the transport distance was less than 1 km. In non-IPBS, purchased maize silage came from the local farm about 30 km away, organic fertilizer was sold to orchard or vegetable farmers about 20 km

**Table 5**
Manure management in different systems.

| | non-IPBS | | | IPBS | | |
|---|---|---|---|---|---|---|
| | ShanD | HeiLJ | Total | ShanD | HeiLJ | Total |
| Solid fraction (t/year) | 1658 | 2266 | 1958 | 2589 | 2359 | 2450 |
| Liquid fraction (t/year) | 490 | 663 | 575 | 773 | 738 | 752 |
| Organic fertilizer (t/year) | 645 | 881 | 761 | 984 | 897 | 931 |
| Application of manure on own cropland (%) | - | - | - | 28.9 | 74.8 | 56.7 |
| Electricity (kWh/year) | 1161 | 1586 | 1371 | 1812 | 1651 | 1715 |
| Diesel (L/year) | 2213 | 3024 | 2613 | 3455 | 3148 | 3269 |
| Land occupation (ha) | 0.14 | 0.23 | 0.18 | 0.41 | 0.21 | 0.29 |
| Sample size | 38 | 37 | 75 | 45 | 69 | 114 |

on average. Concentrate, soybean meal, maize grain, and wheat bran came from local market, with the transport distance of approximately 50 km. Cotton seed meal came from Gansu province with about 1830 km to Qingdao and 2650 km to Harbin. Leymus chinensis came from Heilongjiang province, and the transport distance was about 1720 km to Qingdao and 200 km to Harbin. Oats and alfalfa came from USA, transport distance reached about 18000 km to Qingdao and 20000 km to Harbin. The inventory data on transport were derived from the Ecoinvent Database v3.7 (EC, 2020).

### 2.4. Life cycle impact assessment

In order to get a comprehensive understanding of environmental impacts in both systems, the midpoint characterization method was introduced in this study (Fig. 4), during which emissions and extractions were weighted to represent their contribution to each midpoint category (Michael, 2016). Then the following environmental impact categories were analyzed: global warming potential (GWP), eutrophication potential (EP), acidification potential (AP), non-renewable energy use (NREU), land use (LU) and water use (WU). Characterization factors for all the impact categories were summarized in Table A6.

### 2.5. The linear regression model

After calculating the environmental performance in IPBS, the next step is to identify determinants that influence environmental performance in this system (Fig. 5). Because the GWP, AP, EP, NREU, WU and LU were continuous variables, thus Ordinary Least Square (OLS) can be used in this study. The equation is as follows:

$$Y_i = \alpha + \beta Z_i + \varepsilon_i \tag{2}$$

where $Y_i$ is an explained variable, i.e. GWP, AP, EP, NREU, WU and LU for per $kg^{-1}$ FPCM, $\alpha$ is the constant term, $Z_i$ is the a vector of explanatory variables, $\beta$ represents the coefficient of explanatory variables, $\varepsilon_i$ is the error term. Definitions and descriptive statistics of the explanatory variables were presented in Table 6.

## 3. Results

In this study, the average economic allocation factor in milk production system was 85.0% and 83.8% in non-IPBS and IPBS, respectively. The total environmental performance in milk production system was calculated for 189 dairy farms in both systems (Table 7). Environmental consequence of IPBS was better than non-IPBS (Fig. 6). Compared with non-IPBS, IPBS had great potential in reducing environmental impacts and resource use, with a reduction ratio range from 8% to 18% over various categories.

### 3.1. Global warming potential

The GWP for 1 kg of FPCM production was 1.11–1.40 kg $CO_2$ eq. in two systems (Table 7). Those values were in line with previous researches in China, with a range from 0.73 to 1.77 kg $CO_2$ eq. (Ledgard et al., 2019; Wang et al., 2016, 2018; Liu et al., 2020; Zhou, 2017), but were higher than in Europe or New Zealand (Battini et al., 2014; Ledgard et al., 2019). In non-IPBS, GWP in Shandong was lower than that in Heilongjiang, mainly because that Shandong had a higher milk yield. In IPBS, the GWP in Shandong was higher than that in Heilongjiang, due to a higher self-sufficiency rate of maize silage in Heilongjiang.

Interestingly, compared with non-IPBS, by incorporating self-produce maize silage in IPBS, total GWP decreased by 14%. In specific, GWP from fodder production process (including produced and purchased) decreased by 24% (Fig. 6). The reason might be the reduced consumption of chemical fertilizer replaced by organic fertilizer during maize silage production, which mitigates $CO_2$ and CO emissions. GWP

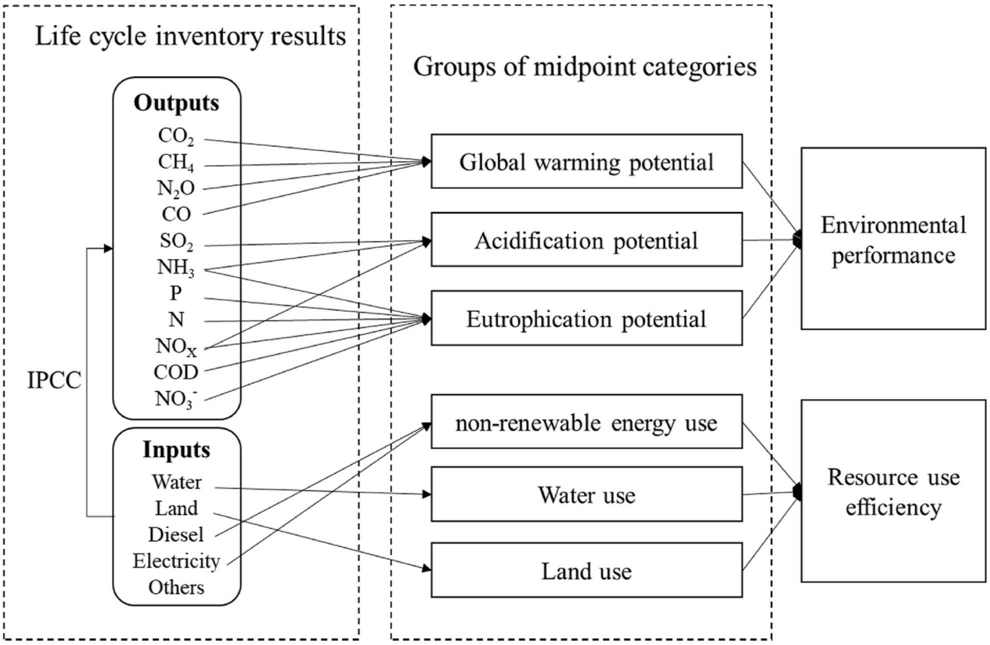


**Fig. 4.** Life cycle impact assessment framework.

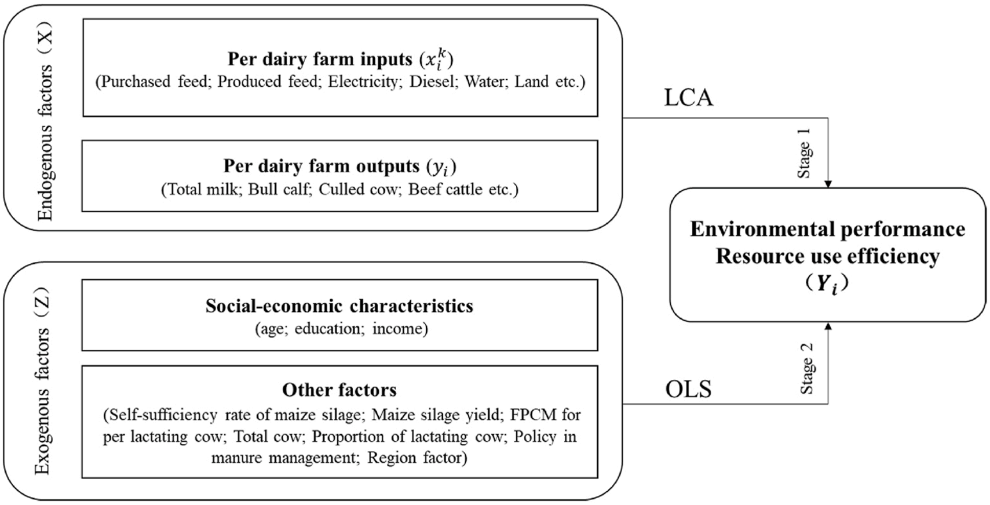


**Fig. 5.** Analytical framework in IPBS.

**Table 6**
Definitions and descriptive statistics of explanatory variables.

| Variables | Definition | Mean | SD |
|---|---|---|---|
| Age | Age of dairy farm owner(year) | 46.6 | 6.9 |
| Primary | Education of dairy farm owner (1: primary school; 0: others) | 0.10 | 0.30 |
| Junior | Education of dairy farm owner (1: junior high school; 0: others) | 0.56 | 0.50 |
| Senior | Education of dairy farm owner (1: senior high school and above; 0: others) | 0.34 | 0.48 |
| Income | Annual dairy farm income ( × 1000 CNY) | 6015 | 5646 |
| Self-sufficiency rate of maize silage | Self-sufficiency rate of maize silage dairy farm (%) | 56.6 | 39.4 |
| Maize silage yield | Maize silage yield in dairy farm (kg/ha) | 43995 | 6810 |
| FPCM for per lactating cow | FPCM for per lactating cow (kg/head) | 8435 | 1000 |
| Total dairy cow | Total dairy cow in dairy farm (head) | 281 | 198 |
| Proportion of lactating cow | Proportion of lactating cow (%) | 45.2 | 11.8 |
| Policy in manure management* | The execution of policies for ban on manure direct discharge is strict or not by local government (1: yes; 0: no) | 0.92 | 0.27 |
| Shandong | (1: Shandong; 0: other provinces) | 0.39 | 0.49 |
| Heilongjiang | (1: Heilongjiang; 0: other provinces) | 0.61 | 0.49 |

Note: “*” Policy in manure management refers to policies for prevention and control of pollution from manure direct discharge (e.g. in 2013, “Regulations on prevention and control of pollution from large-scale livestock and poultry breeding” was issued by the State Council of China (SC, 2013)), if dairy farm owner considered that the execution of policies for ban on manure direct discharge is strict by local government, we define “Policy in manure management” as “1: yes”.

from transportation process decreased by 7%, due to the self-produce can reduce the energy consumption in fodder transportation. The most significant contributor to GWP was fodder production, accounting for about 44% of the total environmental impacts, which means improving fodder production process was the key to achieve GHG emissions reduction in milk production system. Incorporating cropland to produce maize silage in dairy farms had great potential in reducing GHG emissions.

### 3.2. *Acidification potential*

The AP for 1 kg of FPCM production was 15.84–18.89 g $SO_2$ eq. in two systems (Table 7). The results were in line with previous researches in China, with a variation from 12.1 to 30.0 $SO_2$ eq. (Wang et al., 2018; Cao, 2012; Zhou, 2017), but higher than that from researches in Europe or Canada (Arsenault et al., 2009; Matteo et al., 2013).

Compared with non-IPBS, the total AP for IPBS decreased 10%. During the process, AP from fodder production and manure management decreased by 12% and 10%, respectively (Fig. 6). The reason was that self-produce maize silage could reduce chemical fertilizer consumption by replacing organic fertilizer. Meanwhile, cropland is conducive to manure recycling, which has less discharged manure than non-IPBS, thus leading to lower $SO_2$ emissions. In the whole process, about 32% of total AP derived from animal emissions, and about 30% of total AP derived from fodder production. Obviously, compared with animal emissions, reducing AP from fodder production was more achievable. Incorporating cropland to produce maize silage in dairy farms is an effective measure to reduce the total AP in milk production system.

**Table 7**
Results for all the environmental impacts in both systems (per kg$^{-1}$ FPCM).

| | non-IPBS | | | IPBS | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| | ShanD | HeiLJ | Total | ShanD | Reduced* | HeiLJ | Reduced* | Total | Reduced* |
| GWP (kg $CO_2$eq.) | 1.33 | 1.40 | 1.37 | 1.25 | −6.0% | 1.11 | −20.7% | 1.17 | −14.0% |
| AP (g $SO_2$eq.) | 17.6 | 18.9 | 18.2 | 17.1 | −2.8% | 15.8 | −16.4% | 16.3 | −10.4% |
| EP (g $PO_4^{3-}$eq.) | 8.52 | 9.09 | 8.80 | 7.72 | −9.4% | 6.85 | −24.6% | 7.19 | −18.3% |
| NREU (MJ) | 4.56 | 4.62 | 4.59 | 4.29 | −5.9% | 4.03 | −12.8% | 4.13 | −10.0% |
| WU (L) | 403 | 425 | 414 | 383 | −5.0% | 380 | −10.6% | 381 | −8.0% |
| LU ($m^2$) | 1.46 | 1.60 | 1.53 | 1.37 | −6.2% | 1.30 | −18.8% | 1.33 | −13.1% |
| Sample size | 38 | 37 | 75 | 45 | | 69 | | 114 | |

Note: “*” the decreased percentage of environmental impacts compared with non-IPBS.

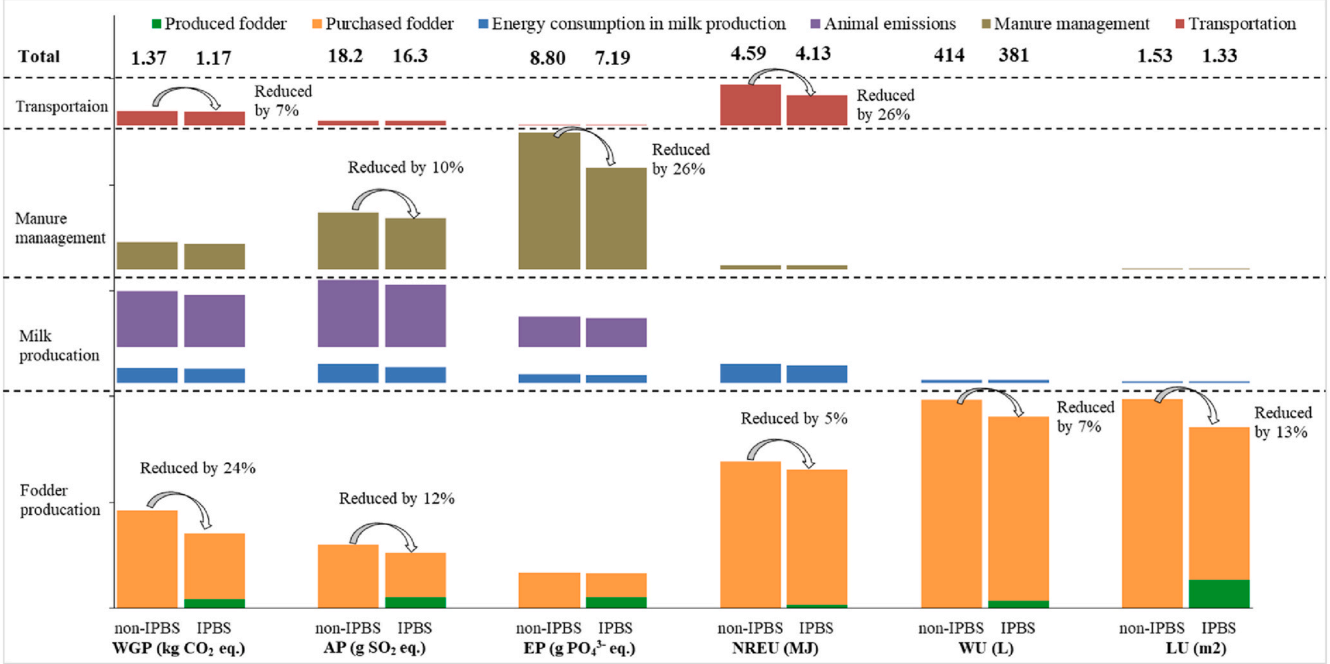


**Fig. 6.** The contribution shares to environmental performance and resource use (per $kg^{-1}$ FPCM).

### 3.3. Eutrophication potential

The EP for 1 kg of FPCM production was 6.85–9.09 g $PO_4^{3-}$ eq. in two systems (Table 7). Results were in line with previous researches in China, ranging from 2.2 to 9.27 g $PO_4^{3-}$ eq. (Wang et al., 2018; Cao, 2012; Zhou, 2017), and were similar to that of Europe (Matteo et al., 2013), but were higher than that in Canada (Arsenault et al., 2009). EP has been significantly improved in milk production system benefited from the promulgation of effective policies relevant to manure management since 2013 (SC, 2013).

Total EP of IPBS decreased by 18% compared with non-IPBS. Manure management decreased by 26% (Fig. 6). The reason might be that IPBS has its own cropland to dispose manure, especially liquid fraction, which has less discharge than non-IPBS, leading to lower loss of N, P and COD. Due to the separation of farms and orchards (vegetable fields), 40% of manure was discharged (Wang et al., 2018), which put a heavy burden on the environment. The largest contributor to EP was manure management, accounting for 62% of the total environmental impacts. The results indicated that controlling manure management was responsible for achieving EP reduction in milk production system, dairy farms with their own cropland can dispose manure and avoid manure leakage in transportation.

### 3.4. Non-renewable energy, water and land use

The NREU for 1 kg of FPCM production was 3.59–4.56 MJ in two systems (Table 7). Those values were in line with previous researches in China (Wang et al., 2018; Liu et al., 2020), but were higher than that in Europe or Canada (Arsenault et al., 2009; Matteo et al., 2013). Compared with non-IPBS, the NREU in IPBS decreased by 10%. Transportation process and fodder production process decreased by 26% and 5%, respectively (Fig. 6). Self-produce maize silage can save energy consumption in transportation. Meanwhile, dairy farms used organic fertilizer to reduce chemical fertilizer consumption, which omitted energy consumption from chemical fertilizer production. The most significant contributor to NREU was fodder production, accounting for 71% of the total energy consumption. This means that reducing NREU from fodder production was the key to achieve NREU reduction in milk production system.

The WU for producing 1 kg of FPCM was 380–425 L in two systems (Table 7). Those values were higher than in Europe or U.S. (Sultana et al., 2014), but still in line with previous researches in China (Wang et al., 2018; Liu et al., 2020). Most of the fodder was produced in a low-rainfall region, so it required more irrigation water. IPBS decreased 8% compared with non-IPBS, fodder production decreased by 7% (Fig. 6). Because the application of liquid fraction substituted irrigation water in maize silage planting, and fodder production process accounting for 98% of the total WU. This means that incorporating cropland to produce maize silage in farms is an effective measure to reduce water use in milk production system.

The LU for producing 1 kg of FPCM was 1.30–1.60 $m^2$ in two systems (Table 7). Those values were in line with previous researches in China (Wang et al., 2018; Liu et al., 2020), but were higher than in Europe or American (Eshel et al., 2014; Matteo et al., 2013). Generally, IPBS decreased by 13%, fodder production process also decreased by 13% (Fig. 6). Obviously, applying organic fertilizer increased the crop yield and led to less cropland area. About 98% of total land use is derived from fodder production, so incorporating cropland to produce maize silage in farms is an effective measure to reduce LU in milk production system.

### 3.5. Determinants of environmental performance and resource use in IPBS

In IPBS, regression results were presented in Table 8. The T-values indicated eight explanatory variables: i.e., Primary, Income, Self-sufficiency rate of maize silage, Maize silage yield, FPCM for per lactating cow, Total cow, Proportion of lactating cow and Policy in manure management significantly influenced environmental performance and resource use in IPBS.

There was a significant relationship between self-sufficiency rate of maize silage and environmental performance for per $kg^{-1}$ FPCM, which meants that farms with a high self-sufficiency rate of maize silage tended to lower environmental impacts in milk production system. Because producing maize silage had a lower environmental impact compared with purchasing. This result was in line with the previous literature (Basset-Mens et al., 2009; Matteo et al., 2013). Primary had a significant impacts on GWP, AP, EP, NREU, WU and LU. Income had significant impacts on GWP, AP and EP. Maize silage yield and FPCM for per

**Table 8**
Determinants of environmental performance and resource use in IPBS (per kg$^{-1}$ FPCM).

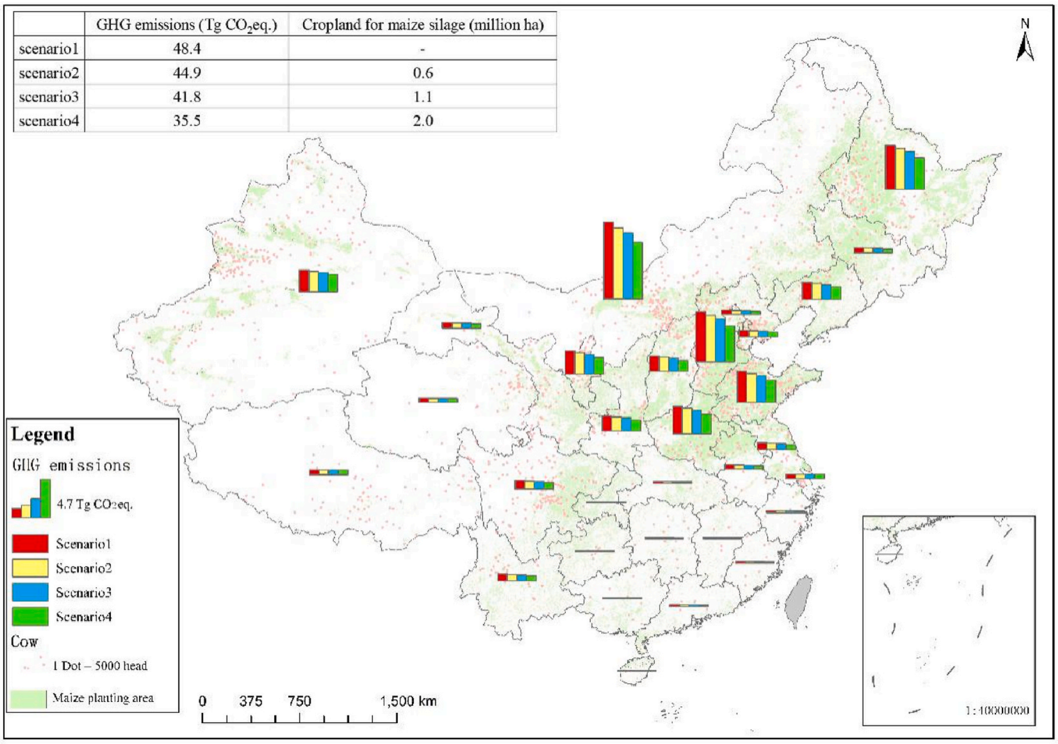
| Variables | GWP | AP | EP | NREU | WU | LU |
|---|---|---|---|---|---|---|
| | Coefficients | Coefficients | Coefficients | Coefficients | Coefficients | Coefficients |
| Age | 0.00126 | 0.00642 | 0.00225 | 0.00697 | 0.90482 | 0.00220 |
| Primary | 0.14547*** | 1.66817** | 0.46940* | 0.92643*** | 91.64402*** | 0.21957*** |
| Senior | −0.05206 | −0.77787* | −0.24693 | −0.26264 | −27.27408 | −0.10195** |
| Income | −0.00003** | −0.00035* | −0.00017** | −0.00011 | −0.00840 | −0.00001 |
| Self-sufficiency rate of maize silage | −0.00176*** | −0.01356** | −0.01753*** | −0.00463* | −0.20423** | −0.00217*** |
| Maize silage yield | −0.00007* | −0.00097** | −0.00026 | −0.00039* | −0.04105 | −0.00016*** |
| FPCM for per lactating cow | −0.00005*** | −0.00094*** | −0.00046*** | −0.00004 | −0.00685 | −0.00007** |
| Total cow | 0.00090** | 0.00983** | 0.00499*** | 0.00305 | 0.24022 | 0.00051 |
| Proportion of lactating cow | −0.01354*** | −0.19547*** | −0.06956*** | −0.04068*** | −3.84057*** | −0.01671*** |
| Policy in manure management | −0.26897*** | −3.11847*** | −1.40661*** | −1.08825*** | −95.25807*** | −0.30424*** |
| Shandong | −0.01781 | −0.35551 | −0.23566 | −0.25769 | −36.52545* | −0.11028* |
| Constant | 2.65637*** | 38.74199*** | 16.91119*** | 8.32837*** | 788.01570*** | 3.47260*** |
| F statistic | 34.90 | 37.00 | 50.61 | 13.55 | 10.72 | 21.42 |
| Prob > F | 0.0000 | 0.0000 | 0.0000 | 0.0000 | 0.0000 | 0.0000 |
| R-squared | 0.7901 | 0.7996 | 0.8452 | 0.5936 | 0.5362 | 0.6979 |
| Adjusted R-squared | 0.7674 | 0.778 | 0.8285 | 0.5498 | 0.4862 | 0.6653 |
| Sample size | 114 | 114 | 114 | 114 | 114 | 114 |

Note: "***" significant at the 10% level; "**" significant at the 5% level; "*" significant at the 10% level.

lactating cow had significant impacts on environmental performance and resource use of milk production system, which means that a high yield of maize silage and milk tended to lower environmental impacts and resource use. The lactating cow had a significant impact on environmental performance, which means that improving herd structure, and controlling the proportion of lactating cows in farms can reduce environmental impacts of milk production. Those results were in line with the previous literature (Wang et al.,2016, 2018). The total cow had a significant impact on environmental performance, which means that enlarging the cow scale will increase environmental impacts of milk production system, which was in line with the previous literature (Delgado et al., 2008). Policy in manure management had a significant impact on environmental performance and resource use of milk production system, due to the strict execution of policies for ban on manure direct discharge by local government can benefit for improving manure management in dairy farms.

### 3.6. *GHG mitigation potentials in milk production system through IPBS*

According to results, the dairy cows' farmland demand, cow distribution and maize planting area in China, four scenarios were contrived and compared in this study (Fig. 7). In scenario1, all dairy farms adopted non-IPBS, which represented the maximum GHG emissions. In scenario2, 50% of farms were adopted IPBS with a 50% of self-sufficiency



| | GHG emissions (Tg $CO_2$eq.) | Cropland for maize silage (million ha) |
|---|---|---|
| scenario1 | 48.4 | - |
| scenario2 | 44.9 | 0.6 |
| scenario3 | 41.8 | 1.1 |
| scenario4 | 35.5 | 2.0 |

**Fig. 7.** GHG mitigation potentials in milk production system in China through IPBS.
Note: "scenario1" all dairy farms were adopted non-IPBS; "scenario2" 50% of dairy farms were adopted IPBS with a 50% of self-sufficiency rate of maize silage; "scenario3" 93% of dairy farms were adopted IPBS with a 50% of self-sufficiency rate of maize silage; "scenario4" 81% of dairy farms were adopted IPBS with a 100% of self-sufficiency rate of maize silage.

rate of maize silage. This scenario was in line with the current milk production system in China, representing current GHG emissions. Scenario3 represented that the theoretical highest adoption rate of IPBS can be achieved 93% when the self-sufficiency rate of maize silage was 50% in this system. Scenario4 represented that the theoretical highest adoption rate of IPBS can be achieved 81% when the self-sufficiency rate of maize silage was 100% in this system.

In 2019, GHG emissions from milk production system in scenario1 were 48.4 Tg $CO_2$ eq., accounting for 16% of total livestock GHG emissions (FAOSTAT, 2019). If 50% of dairy farms were adopted IPBS, GHG emissions were 44.9 Tg $CO_2$ eq., scenario2 had a reduction of 7% compared with scenario1, which was in line with previous researches (Wang et al., 2019a). About 0.6 million ha cropland planted maize silage in this scenario, accounting for 2% of China's total maize planting area. In scenario3, there was 1.1 million ha cropland and the GHG emissions were 41.8 Tg $CO_2$ eq., with a reduction of 7% compared with scenario2. In scenario4, GHG emissions were 35.5 Tg $CO_2$ eq., with a reduction of 21% compared with scenario2. About 2.0 million ha cropland was used for maize silage planting in this scenario, accounting for 5% of the total maize planting area in China. Obviously, IPBS had great potential in reducing GHG emissions.

## 4. Discussion

In order to achieve the goal of a cleaner milk production system, improving the environmental performance and resource use efficiency is particularly significant. There were about 32 million tons of milk produced in 2019 in China (NBSC, 2020). Dairy farms of various sizes were spread across the country, with a total of 11 million dairy cows (NBSC, 2020). For the Chinese government's dairy industry development goals, China will produce 45 million tons of milk by 2025 (MOA, 2018), with an increase of 40% from 2019. Although China is the third largest milk producer in the world, milk production with low productivity has high environmental impacts compared with developed countries (Bai et al., 2013; Eshel et al., 2014). According to our study, compared with non-IPBS, the IPBS had great potential in reducing GWP, AP, EP, NREU, WU and LU in milk production system (Table 7), with a noticeable reduction ratio range from 8% to 18% in different environmental impact categories. This means that IPBS is a valid option for environmental impacts mitigation and resource use efficiency improvement. Meanwhile, the Chinese government attaches great importance to silage maize planting (MOA, 2017). It is very likely that more and more dairy farms will adopt IPBS in the future, which is meaningful for the sustainability of milk production system.

IPBS had great potential in reducing GHG emissions. In China, producing more milk with a low-carbon production has become a grand challenge to achieve the specific Paris Agreement commitments by 2030 and carbon neutrality by 2060 (PRI, 2021; CAT, 2020; Zhou et al., 2019). According to our study, compared with non-IPBS, the IPBS had great potential in reducing GHG emissions in milk production system (Table 7), with a ratio of 14%, and dairy farms with a high self-sufficiency rate of maize silage tended to have lower GHG emissions (Table 8). In 2018, GHG emissions from livestock production systems in China were about 298 Tg $CO_2$ eq., accounting for 47% of total agricultural GHG emissions (FAOSTAT, 2019) and 3% of total domestic GHG emissions (PRI, 2021). It was estimated that GHG emissions from milk production system in China were 44.9 Tg $CO_2$ eq. in 2019, accounting for 15% of total livestock GHG emissions (Fig. 7). In the future, approximately 81% of dairy farms can adopt IPBS when the self-sufficiency rate of maize silage is 100% in this system, and GHG emissions will reduce by 21% compared with current situation, which means that total livestock GHG emissions will reduce by 3%. Only by reducing GHG emissions from all industries with all possible means, can China achieve the goal of carbon neutrality before 2060.

IPBS substantially increases the recycling rate of manure, reduces the consumption of chemical fertilizer and improves soil quality, which is meaningful to mitigating water eutrophication and soil acidification. In China, livestock manure is one of the crucial sources causing water eutrophication (Du et al., 2018; Huang et al., 2017), while applicated manure to cropland is an effective measure to reduce N and P losses (Zhang et al., 2019). However, crop farmers are accustomed to using chemical fertilizer, and are not willing to use organic fertilizer (Zhang et al., 2019, 2020). Recycling manure used in cropland is an essential challenge for China (Jin et al., 2020). At present, only about 40% of manure return to cropland in China (Gu et al., 2017), while in Europe, this share was over 65% (Bai et al., 2016). Based on our survey, currently, only orchards and vegetable field owners were willing to adopt organic fertilizer. Crop farmers reluctant to utilize organic fertilizer due to the inconvenience and high cost compared to chemical fertilizer. On the other hand, so far, China is the largest chemical fertilizer consumer worldwide, accounting for approximately 1/3 of the total global chemical fertilizer (N) consumption, resulting in water eutrophication, soil acidification, global warming (Jin et al., 2020). The protection and improvement of cultivated land quality should be considered in China (Wang et al., 2019). Long-term organic fertilizer application can improve soil quality and increase crop yield (Lin et al., 2015; Zhang et al., 2009). Obviously, incorporating cropland to produce maize silage in dairy farms is meaningful to recycling manure, as well as reduces the chemical fertilizer consumption, which is substantive in promoting green and cleaner agriculture production.

Besides mitigation of environmental impacts, IPBS improved the efficiency of non-renewable energy, water and land use. Milk production system consumed considerable amounts of non-renewable energy, water and land, resulting in resources scarcity (Sultana et al., 2014; Eshel et al., 2014). According to Delivering carbon neutrality in China (PRI, 2021), a plan for zero or near-zero carbon electricity will be implemented, all unabated coal power generation will be phase out in the future, and fossil fuel cars and vans will be not for sale by 2040. This means that the cost of electricity and diesel in farms, even the cost of transportation for fodder will increase in the future. Based on our study, maize silage production by dairy farm's own cropland can decrease the energy consumption in transportation, with a reduction ratio of 26% (Fig. 6). Livestock production system consumed about 32% of global agricultural water and 33% of global cropland to produce animal fodder (Herrero et al., 2016; Wei et al., 2018). It is predicted that available water for agriculture will reduce by 18% by 2050 (Strzepek et al., 2010). Along with the growth of population, milk production system will face severe challenges, such as water scarcity, land scarcity, competition between food and fodder, and limitation of GHG emissions. IPBS is a valid option for an improvement of resource use efficiency.

Interestingly, this study also found that the environmental performance of milk production system was affected by self-sufficiency rate of maize silage, and also restricted by the yield of maize silage and milk. Based on the regression analysis, dairy farms with a high self-sufficiency rate of maize silage, a high maize silage yield and milk yield tended to have lower environmental impacts on milk production system (Table 8). IPBS was adopted by 60.3% of the 189 interviewed farms, with a self-sufficiency rate of only 56.6% for maize silage in this system (Table 2), but a rate of over 70% in Europe (Matteo et al., 2013). A high fodder self-sufficiency is not only of benefits to GHG emissions, acidification and energy use (Matteo et al., 2013), and also for the reduction of fodder cost (Liu et al., 2018). The milk yield was 8173–8781 kg/head in different systems in our study region. Even in China, the average dairy cow milk yield was only 5647 kg/head in 2019, which was lower than that of Europe or New Zealand (FAOSTAT, 2019). The maize yield was 6317 kg/ha in China in 2019, which was lower than that of U.S. (FAOSTAT, 2019). China still has considerable potential for improving milk yield and maize silage yield. Along with the increasing milk and maize silage yield, environmental impacts of milk production system will be lower than the current estimation in the future. Thus, mitigation of environmental impacts from milk production system should combine a short-term strategy of increasing the planting area of maize silage in dairy farms and a long-term plan for technological improvement in the yield of milk and maize silage.

## 5. Conclusions

This study demonstrated a comprehensive comparison of environmental performance between IPBS and non-IPBS in Shandong and Heilongjiang provinces. For the interviewed dairy farms, 60% had already adopted IPBS, however, the self-sufficiency rate of maize silage was only 57% in this system. Environmental impacts for production of 1 kg of FPCM in non-IPBS were 1.37 kg $CO_2$ eq., 18.22 g $SO_2$ eq., 8.81 g $PO_4^{3-}$ eq., 4.59 MJ, 414 L and 1.53 $m^2$ for GWP, AP, EP, NREU, BWU and LU, respectively. Compared with non-IPBS, IPBS had the apparent potential to reduce environmental impacts and improve resource utilization efficiency (Table 7), with the reduction ratio ranging from 8% to 18% in various categories. Currently, GHG emissions from milk production system was 44.9 Tg $CO_2$ eq. in China. It is estimated that 81% of dairy farms could adopt IPBS with a 100% of self-sufficiency rate of maize silage, resulting in a reduction of approximately 21% in GHG emissions compared to current situation, but the premise is that 2.0 million ha cropland should be used for maize silage cultivation. Besides mitigation of environmental impacts, IPBS can substantially increased the recycling rate of manure, reduced the consumption of chemical fertilizer and improved soil quality, which all contribute to the sustainability of agricultural development. With IPBS, environmental performance of milk production was affected by the self-sufficiency rate of maize silage along with maize silage yield. It is also restricted by the milk yield, cow scale and proportion of lactating cow. Mitigation of the environmental impacts from milk production system should combine a short-term strategy of increasing planting area of maize silage in dairy farms and a long-term plan for technological improvements in milk and crop yield.

However, there are several limitations of this study to note. Firstly, the survey data used in this study were collected from only two provinces of China. Although they are the main milk production provinces in China, the estimated environmental impacts for the IPBS and non-IPBS may not be enough for application in other provinces. Thus, more surveys are suggested to strengthen the reliability and applicability of these findings. Additionally, this study focused purely on environmental impacts of milk production systems, cost-benefit analysis should be incorporated. Therefore, a more comprehensive analysis that evaluates both the environmental and economic disparities between non-IPBS and IPBS should be conducted in future studies.

## CRediT authorship contribution statement

**Xianlei Huang:** Conceptualization, Investigation, Methodology, Software, Data curation, Visualization, Writing – original draft. **Boyang Shi:** Investigation, Data curation, Visualization, Writing – review & editing. **Shu Wang:** Conceptualization, Methodology, Software, Writing – review & editing. **Changbin Yin:** Funding acquisition, Conceptualization, Methodology, Supervision, Validation, Writing – review & editing. **Linna Fang:** Conceptualization, Methodology, Supervision, Validation, Writing – review & editing.

## Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

## Acknowledgments

This research was funded by Major Program of National Philosophy and Social Science Foundation of China (18ZDA048) and China Agriculture Research System-Green Manure (CARS-22-G-25). We would like to thank the editor and the anonymous reviewers for their helpful comments and suggestions.

## Appendix A

**Table A1**
Emission factors for calculating environmental impacts in produced fodder

| | Emission factors | | | | | | |
|---|---|---|---|---|---|---|---|
| | $CO_2$ | CO | $NO_x$ | $N_2O$ | $NH_3$ | $NO_3$ | $SO_2$ |
| Production of fertilizer(N) (kg/kg) (Hu et al., 2006) | 10.37 | 0.0043 | 0.036 | - | - | - | 0.032 |
| Production of fertilizer ($P_2O_5$) (kg/kg) (Hu et al., 2006) | 1.59 | 0.0008 | 0.005 | - | - | - | 0.003 |
| Production of fertilizer ($K_2O$) (kg/kg) (Hu et al., 2006) | 0.66 | 0.0004 | 0.002 | - | - | - | 0.001 |
| Application of fertilizer(N) (kg/kg) (Wang et al., 2018) | - | - | - | 0.0105 | 0.048[a]<br>0.036[b] | 0.03[a]<br>0.035[b] | - |
| Application of organic fertilizer(N) (kg/kg) (Wang et al., 2012) | - | - | - | 0.0105 | 0.2 | 0.25 | - |
| Production of electricity (kg/kWh) (Hu et al., 2006) | 1.01 | 0.0005 | 0.003 | - | - | - | 0.002 |
| Production and Application of diesel(kg/L) (Hu et al., 2006) | 3.09 | 0.0011 | 0.005 | - | - | - | 0.010 |

Note: “a” maize silage; “b” wheat.

**Table A2**
Emission factors for calculating environmental impacts in purchased fodder (per $kg^{-1}$ DM)

| | Kg $CO_2$eq. | Kg $SO_2$eq. | Kg $PO_4^{3-}$eq. | MJ | $M^3$ water | $M^2$ land |
|---|---|---|---|---|---|---|
| Maize grain (Liu, 2017) | 0.7368 | 0.0058 | 0.0015 | 5.943 | 0.534 | 1.604 |
| Wheat bran (Liu, 2017) | 0.2748 | 0.0018 | 0.0005 | 2.311 | 0.158 | 0.580 |
| Soybean meal (Liu, 2017) | 0.5716 | 0.0094 | 0.0019 | 7.299 | 0.765 | 2.904 |
| Maize silage (Liu, 2017) | 0.2248 | 0.0018 | 0.0005 | 1.757 | 0.181 | 0.550 |
| Leymus chinensis (Liu et al., 2020) | 0.2000 | 0.0016 | 0.0006 | 0.350 | 0.167 | 0.423 |
| Oats (Sara et al., 2016) | 0.2820 | 0.0013 | 0.0010 | 2.145 | 0.309 | 0.515 |
| Alfalfa (Liu, 2017) | 0.2480 | 0.0049 | 0.0011 | 2.156 | 0.703 | 1.091 |
| Cotton seed meal (Liu, 2017) | 0.8146 | 0.0047 | 0.0009 | 7.552 | 0.726 | 0.773 |
| Wheat straw (Liu, 2017) | 0.0104 | 0.0007 | 0.0002 | 0.898 | 0.062 | 0.229 |

**Table A3**
Emission factors for calculating environmental impacts in milk production

| | Emission factors (kg $CH_4$/head/year) (Liu et al., 2020) | | Emission factors (kg $NH_3$/head/year) (Battini et al., 2014) |
|---|---|---|---|
| | emissions from animal enteric | emissions from cowshed manure | emissions from animal enteric |
| Lactating cow | 61.00 | 9.00 | 17.3 |
| Non lactating cow | 47.00 | 1.00 | |
| Finishing cow | 36.15 | 1.00 | |
| Calf | 10.40 | 1.00 | |

**Table A4**
Excreted parameters of manure and nutrient for different cow

| | Solid fraction (kg/head/day) | | | | Liquid fraction (kg/head/day) | | | | | VS(kg/head/day) |
|---|---|---|---|---|---|---|---|---|---|---|
| | Production | TN(%) | TP(%) | TOC(%) | production | TN(%) | TP(%) | COD(%) | TOC(%) | |
| Lactating cow (Duan, 2018) | 32.84 | 0.56 | 0.07 | 5.03 | 13.24 | 0.50 | 0.02 | 2.79 | 0.70 | 2.8 |
| Finishing cow (Duan, 2018) | 18.57 | | | | 2.62 | | | | | |

Note: Excreted manure in non-lactating cow was same as finishing cow, excreted manure in calf was half that of fattening cow.

**Table A5**
Emission factors for calculating environmental impacts in manure management

| | Unit | Manure passive composting |
|---|---|---|
| Maximum $CH_4$ producing capacity (IPCC, 2006) | m3 $CH_4$ $kg^{-1}$ VS | 0.13 |
| $CH_4$ emission conversion factor (IPCC, 2006) | % | 0.5 |
| $N_2O$ direct emission factor (IPCC, 2006) | % | 1 |
| Manure N volatilization (IPCC, 2006) | % | 40 |
| $N_2O$ indirect emission factor from volatilization (IPCC, 2006) | % | 1 |
| $NH_3$ emission factor (Zhang et al., 2010) | kg $t^{-1}$ manure | 0.961 |
| $NH_3$–N (Zhang et al., 2010) | kg $t^{-1}$ manure | 0.128 |
| $CO_2$ emission factor (Zhang et al., 2010) | kg $t^{-1}$ manure | 36.201 |
| $NO_3$ emission factor (Zhang et al., 2010) | kg $t^{-1}$ manure | 0.948 |
| TP (Zhang et al., 2010) | kg $t^{-1}$ manure | 0.001 |
| COD (Zhang et al., 2010) | kg $t^{-1}$ manure | 0.640 |

**Table A6**
Characterization factors for calculating the environmental impacts

| Impact category | Unit | Contributing substance | Characterization factor |
|---|---|---|---|
| Global warming potential (GWP) (IPCC, 2013) | kg $CO_2$eq. | $CO_2$ | 1 |
| | | $CH_4$ | 30 |
| | | $N_2O$ | 265 |
| | | CO | 2 |
| Eutrophication potential (EP) (Guinee, 2002) | kg $PO_4^{3-}$eq. | $PO_4^{3-}$ | 1 |
| | | P | 3.06 |
| | | N | 0.42 |
| | | $NH_3$ | 0.35 |
| | | $NO_x$ | 0.13 |
| | | COD | 0.022 |
| | | $NO_3$ | 0.13 |
| Acidification potential (AP) (Wang et al., 2018) | kg $SO_2$eq. | $SO_2$ | 1.2 |
| | | $NH_3$ | 1.6 |
| | | $NO_x$ | 0.5 |

## References

Ardente, F., Cellura, M., 2012. Economic allocation in life cycle assessment. J. Ind. Ecol. 16 (3) https://doi.org/10.1111/j.1530–9290.2011.00434.x.

Arsenault, N., Tyedmers, P., Fredeen, A., 2009. Comparing the environmental impacts of pasture-based and confinement-based dairy systems in Nova Scotia (Canada) using life cycle assessment. Int. J. Agric. Sustain. 7 (1), 19–41. https://doi.org/10.3763/ijas.2009.0356.

Awasthi, M.K., Sarsaiya, S., Wainaina, S., Rajendran, K., Kumar, S., Quan, W., Duan, Y., Awasthi, S.K., Chen, H., Pandey, A., Zhang, Z., Jain, A., Taherzadeh, M.J., 2019. A critical review of organic manure biorefinery models toward sustainable circular bioeconomy: technological challenges, advancements, innovations, and future perspectives. Renew. Sust. Energy Rev. 111, 115–131. https://doi.org/10.1016/j.rser.2019.05.017.

Bai, Z., Ma, L., Jin, S., Ma, W., Velthof, G.L., Oenema, O., Liu, L., Chadwick, D., Zhang, F., 2016. Nitrogen, phosphorus, and potassium flows through the manure management chain in China. Environ. Sci. Technol. 50 (24), 13409–13418. https://doi.org/10.1021/acs.est.6b03348.

Bai, Z., Ma, L., Oenema, O., Chen, Q., Zhang, F., 2013. Nitrogen and phosphorus use efficiencies in dairy production in China. J. Environ. Qual. 42 (4), 990–1001. https://doi.org/10.2134/jeq2012.0464.

Baldini, C., Gardoni, D., Guarino, M., 2017. A critical review of the recent evolution of Life Cycle Assessment applied to milk production. J. Clean. Prod. 140 (SI2), 421–435. https://doi.org/10.1016/j.jclepro.2016.06.078.

Basset-Mens, C., Ledgard, S., Boyes, M., 2009. Eco-efficiency of intensification scenarios for milk production in New Zealand. Ecol. Econ. Times 68 (6), 1615–1625. https://doi.org/10.1016/j.ecolecon.2007.11.017.

Battini, F., Agostini, A., Boulamanti, A.K., Giuntoli, J., Amaducci, S., 2014. Mitigating the environmental impacts of milk production via anaerobic digestion of manure: case study of a dairy farm in the Po Valley. Sci. Total Environ. 481, 196–208. https://doi.org/10.1016/j.scitotenv.2014.02.038.

Cao, Z., 2012. Life cycle assessment of dairy farm in Heilongjiang province, China. Northeast Agricultural University pp.1–51 (in Chinese).

CAT (Climate Action Tracker), 2020. China going carbon neutral before 2060 would lower warming projections by around 0.2 to 0.3 degrees C. https://climateactiontracker.org/press/china-carbon-neutral-before-2060-would-lower-warming-projections-by-around-2-to-3-tenths-of-a-degree accessed 1 December 2020.

DARH (Department of Agriculture and rural affairs of Heilongjiang), 2020. Plan for grain to feed in Heilongjiang province in 2020. http://nynct.hlj.gov.cn/ywgl/xmsy/wj/2020/4382.htm accessed 25 October 2020.

Del Prado, A., Mas, K., Pardo, G., Gallejones, P., 2013. Modelling the interactions between C. N farm balances and GHG emissions from confinement dairy farms in northern Spain. Sci. Total Environ 465 (SI), 156–165. https://doi.org/10.1016/j.scitotenv.2013.03.064.

Delgado, C., Narrod, C., Tiongco, M., 2008. Determinants and implications of the growing scale of livestock farms in four fast-growing developing countries. Soc. Sci. Electr. Publ. 157 https://doi.org/10.2139/ssrn.1275538.

Du, Y., Ge, Y., Ren, Y., Fan, X., Pan, K., Lin, L., Wu, X., Min, Y., Meyerson, L.A., Heino, M., Chang, S.X., Liu, X., Mao, F., Yang, G., Peng, C., Qu, Z., Chang, J., Didham, R.K., 2018. A global strategy to mitigate the environmental impact of China's ruminant consumption boom. Nat. Commun. 9 (1), 4133. https://doi.org/10.1038/s41467-018-06381-0.

Duan, X., 2018. Life cycle assessment of manure management systems in intensive dairy farms. Northwest A & F University pp.1–62 (in Chinese).

EC (Ecoinvent Centre), (2020). Ecoinvent Data v3.7. https://www.ecoinvent.org/database/ecoinvent-371/ecoinvent-371.html (accessed 25 October 2020).

Eshel, G., Shepon, A., Makov, T., Milo, R., 2014. Land, irrigation water, greenhouse gas, and reactive nitrogen burdens of meat, eggs, and dairy production in the United States. P. Natl. Acad. Sci. USA 111 (33), 11996–12001. https://doi.org/10.1073/pnas.1402183111.

Fan, X., Chang, J., Ren, Y., Wu, X., Du, Y., Xu, R., Liu, D., Chang, S.X., Meyerson, L.A., Peng, C., Ge, Y., 2018. Recoupling industrial dairy feedlots and industrial farmlands mitigates the environmental impacts of milk production in China. Environ. Sci. Technol. 52, 3917–3925. https://doi.org/10.1021/acs.est.7b04829.

FAOSTAT, 2019. FAO statistical databases. http://www.fao.org/faostat/en/#home accessed 25 October 2020.

Gu, B., Ju, X., Chang, S.X., Ge, Y., Chang, J., 2017. Nitrogen use efficiencies in Chinese agricultural systems and implications for food security and environmental protection. Reg. Environ. Change 17 (4SI), 1217–1227. https://doi.org/10.1007/s10113-016-1101-5.

Guinee, J., 2002. Handbook on life cycle assessment - operational guide to the ISO standards. Int. J. Life Cycle Assess. 6 (5), 255. https://doi.org/10.1007/BF02978897.

Herrero, M., Henderson, B., Havlik, P., Thornton, P.K., Conant, R.T., Smith, P., Wirsenius, S., Hristov, A.N., Gerber, P., Gill, M., Butterbach-Bahl, K., Valin, H., Garnett, T., Stehfest, E., 2016. Greenhouse gas mitigation potentials in the livestock sector. Nat. Clim. Change 6 (5), 452–461. https://doi.org/10.1038/nclimate2925.

Hu, Z., Tan, P., Lou, D., Dong, Y., 2006. Assessment of life cycle energy consumption and emissions for several kinds of feedstock based biodiesel. Transactions of the CSAE (11) 141–146 (in Chinese).

Huang, J., Xu, C., Ridoutt, B.G., Wang, X., Ren, P., 2017. Nitrogen and phosphorus losses and eutrophication potential associated with fertilizer application to cropland in China. J. Clean. Prod. 159, 171–179. https://doi.org/10.1016/j.jclepro.2017.05.008.

IPCC (The Intergovernmental Panel on Climate Change), 2006. Guidelines for national greenhouse gas inventories. http://www.ipcc-nggip.iges.or.jp/public/2006gl/vol4.html. (Accessed 25 October 2020).

IPCC (The Intergovernmental Panel on Climate Change), 2013. Climate change 2013: the physical science basis. https://www.ipcc.ch/report/ar5/wg1/. (Accessed 25 October 2020).

ISO (International Standardisation Organisation), 2020. ISO 14040:2006/AMD 1:2020: environmental management - life cycle assessment - principles and framework - amendment 1. https://www.iso.org/standard/76121.html. (Accessed 19 April 2021).

Jin, S., Zhang, B., Wu, B., Han, D., Hu, Y., Ren, C., Zhang, C., Wei, X., Wu, Y., Mol, A.P.J., Reis, S., Gu, B., Chen, J., 2020. Decoupling livestock and crop production at the household level in China. Nat. Sustain. Times 1–8. https://doi.org/10.1038/s41893-020-00596-0.

Kimberly, R. Stackhouse, Sara, E. Place, Michelle, S. Calvo, Qian, Wang, Frank, M. Mitloehner, 2011. Greenhouse gas emission sources from beef and dairy production systems in the United States. Acs. Sym 1072, 407–417. https://doi.org/10.1021/bk-2011-1072.ch021.

Ledgard, S.F., Wei, S., Wang, X., Falconer, S., Zhang, N., Zhang, X., Ma, L., 2019. Nitrogen and carbon footprints of dairy farm systems in China and New Zealand, as influenced by productivity, feed sources and mitigations. Agric. Water Manag. 213, 155–163. https://doi.org/10.1016/j.agwat.2018.10.009.

Lin, Z., Chang, X., Wang, D., Zhao, G., Zhao, B., 2015. Long-term fertilization effects on processing quality of wheat grain in the North China Plain. Field Crop. Res. 174, 55–60. https://doi.org/10.1016/j.fcr.2015.01.008.

Liu, C., Han, L., Zhang, Y., 2018. International comparison and development suggestions of China's dairy industry competitiveness. Chinese rural economy 403 (7), 130–146 (in Chinese).

Liu, S., 2017. Life cycle assessment of dairy cattle feed crops in Guanzhon Plain. Northwest A & F University pp.1–69 (in Chinese).

Liu, X., Wang, L., Wu, R., Xin, X., Sun, H., Jiang, M., Li, X., Wang, M., Liu, Y., Shao, C., 2020. LCA-based assessment of Hulunber ecological grassland technology integration demonstration. Sci. Agric. Sin. 53 (13), 2703–2714 (in Chinese).

Matteo, G., Marie, T.K., Luciana, B., Maddalena, Z., Philipp, S.N., Troels, K., 2013. Parameters affecting the environmental impact of a range of dairy farming systems in Denmark, Germany and Italy. J. Clean. Prod. 54 https://doi.org/10.1016/j.jclepro.2013.04.035.

Michael, Z. Hauschild, Mark, A.J. Huijbregts, 2016. Life Cycle Impact Assessment. Springer.

MOA (Ministry of Agriculture and Rural Affairs of the People's Republic of China), 2017. Plan for planting crops for feed instead of food. http://www.moa.gov.cn/nybgb/2017/dlq/201712/t20171231_6133718.htm (accessed 25 October 2020).

MOA (Ministry of Agriculture and Rural Affairs of the People's Republic of China), 2018. Opinions on further promoting the development of dairy industry. http://www.moa.gov.cn/nybgb/2019/201901/201905/t20190503_6288218.htm (accessed 25 October 2020).

NBSC (National Bureau of Statistics of China), 2020. China Rural Statistical Yearbook 2020. China Statistics Press, Beijing, China.

NDRC (National Development and Reform Commission of China), 2019. National Agricultural Product Cost-Benefit Data Compilation 2019. China Statistics Press, Beijing, China.

Nkoa, R., 2014. Agricultural benefits and environmental risks of soil fertilization with anaerobic digestates: a review. Agron. Sustain. Dev. 34 (2), 473–492. https://doi.org/10.1007/s13593-013-0196-z.

PRI (Principles for responsible investment), 2021. Delivering carbon neutrality in China. https://www.unpri.org/pri-blogs/delivering-carbon-neutrality-in-china/7000.article. (Accessed 19 January 2021).

Sara, G., Francesc, B., Gumersindo, F., Maria, T.M., 2016. Environmental performance of sorghum, barley and oat silage production for livestock feed using life cycle assessment. Resour. Conserv. Recycl. 111. https://doi.org/10.1016/j.resconrec.2016.04.002.

SC (the State Council of China), 2013. Regulations on prevention and control of pollution from large-scale livestock and poultry breeding. http://www.gov.cn/flfg/2013-11/26/content_2535095.htm. (Accessed 19 April 2021).

SPG (Shandong province People's Government), 2016. Plan for grain to feed in Shandong province. http://gb.shandong.gov.cn/art/2016/3/23/art_107851_89320.html. (Accessed 25 October 2020).

Strzepek, K., Boehlert, B., 2010. Competition for water for the food system. Phil. Trans. R. Soc. B 365 (1554), 2927–2940. https://doi.org/10.1098/rstb.2010.0152.

Sultana, M.N., Uddin, M.M., Ridoutt, B.G., Peters, K.J., 2014. Comparison of water use in global milk production for different typical farms. Agric. Syst. 129, 9–21. https://doi.org/10.1016/j.agsy.2014.05.002.

Thornton, P.K., 2010. Livestock production: recent trends, future prospects. Philos. T. R. Soc. Biol. 365 (1554), 2853–2867. https://doi.org/10.1098/rstb.2010.0134.

Wang, H., Qiao, J., 2019. Decoupling and predictive analysis of greenhouse gas emission from animal husbandry in China. Chin. J. Eco-Agric. 27 (5), 793–802 (in Chinese).

Wang, J., Li, P., Zhan, Y., Tian, S., 2019. Study on the protection and improvement of cultivated land quality in China. China population, resources and environment 29 (4), 87–93 (in Chinese).

Wang, X., Liang, D., Wang, X., Peng, S., Zheng, J., 2012. Estimation of greenhouse gas emissions from dairy farming systems based on LCA. Transactions of the CSAE 28 (13), 179–184 in Chinese).

Wang, X., Kristensen, T., Mogensen, L., Knudsen, M.T., Wang, X., 2016. Greenhouse gas emissions and land use from confinement dairy farms in the Guanzhong plain of China - using a life cycle assessment approach. J. Clean. Prod. 113, 577–586. https://doi.org/10.1016/j.jclepro.2015.11.099.

Wang, X., Ledgard, S., Luo, J., Guo, Y., Zhao, Z., Guo, L., Liu, S., Zhang, N., Duan, X., Ma, L., 2018. Environmental impacts and resource use of milk production on the North China Plain, based on life cycle assessment. Sci. Total Environ. 625, 486–495. https://doi.org/10.1016/j.scitotenv.2017.12.259.

Wei, S., Bai, Z.H., Chadwick, D., Hou, Y., Qin, W., Zhao, Z.Q., Jiang, R.F., Ma, L., 2018. Greenhouse gas and ammonia emissions and mitigation options from livestock production in peri-urban agriculture: beijing - A case study. J. Clean. Prod. 178, 515–525. https://doi.org/10.1016/j.jclepro.2017.12.257.

Wei, Y.S., Fan, Y.B., Wang, M.J., Wang, J.S., 2000. Composting and compost application in China. Resour. Conserv. Recycl. 30 (4), 277–300. https://doi.org/10.1016/S0921-3449(00)00066-5.

Winans, K., Kendall, A., Deng, H., 2017. The history and current applications of the circular economy concept. Renew. Sust. Energy Rev. 68 (1), 825–833. https://doi.org/10.1016/j.rser.2016.09.123.

Yin, C., Zhou, Y., Liu, L., 2013. Theory and practice of recycle agriculture in China. Chin. J. Eco-Agric. 21 (1), 47–53 (in Chinese).

Zhang, T., Hou, Y., Meng, T., Ma, Y., Tan, M., Zhang, F., Oenema, O., 2020. Replacing synthetic fertilizer by manure requires adjusted technology and incentives: a farm survey across China. Resour. Conserv. Recycl. 186, 105301. https://doi.org/10.1016/j.resconrec.2020.105301.

Zhang, Y., Xia, X., Li, Z., Wang, M., Yang, T., Xi, B., 2010. Life cycle assessment of manure treatment in scaled cattle farms. Journal of Agro-Environment Science 29 (7), 1423–1427 (in Chinese).

Zhang, C., Liu, S., Wu, S., Jin, S., Reis, S., Liu, H., Gu, B., 2019. Rebuilding the linkage between livestock and cropland to mitigate agricultural pollution in China. Resour. Conserv. Recycl. 144, 65–73. https://doi.org/10.1016/j.resconrec.2019.01.011.

Zhang, H., Xu, M., Zhang, F., 2009. Long-term effects of manure application on grain yield under different cropping systems and ecological conditions in China. J. Agr. Sci-Cambridge 147 (1), 31–42. https://doi.org/10.1017/S0021859608008265.

Zheng, Y.H., Wei, J.G., Li, J., Feng, S.F., Li, Z.F., Jiang, G.M., Lucas, M., Wu, G.L., Ning, T.Y., 2012. Anaerobic fermentation technology increases biomass energy use efficiency in crop residue utilization and biogas production. Renew. Sustain. Energy Rev. 16 (7), 4588–4596. https://doi.org/10.1016/j.rser.2012.03.061.

Zhou, J., 2017. Analysis on environmental impact of Livestock and poultry breeding based on life Cycle assessment. Qingdao University of Science & Technology pp.1–106 (in Chinese).

Zhou, N., Price, L., Dai, Y., Creyts, J., Khanna, N., Fridley, D., Lu, H., Feng, W., Liu, X., Hasanbeigi, A., Tian, Z., Yang, H., Bai, Q., Zhu, Y., Xiong, H., Zhang, J., Chrisman, K., Agenbroad, J., Ke, Y., McIntosh, R., Mullaney, D., Stranger, C., Wanless, E., Wetzel, D., Yee, C., Franconi, E., 2019. A roadmap for China to peak carbon dioxide emissions and achieve a 20% share of non-fossil fuels in primary energy by 2030. Appl. Energy 239, 793–819. https://doi.org/10.1016/j.apenergy.2019.01.154.

Zucali, M., Bacenetti, J., Tamburini, A., Nonini, L., Sandrucci, A., Bava, L., 2018. Environmental impact assessment of different cropping systems of home-grown feed for milk production. J. Clean. Prod. 172 (PT.4), 3734–3746. https://doi.org/10.1016/j.jclepro.2017.07.048.