Environmental footprints of soybean production in China

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Abstract
As a significant protein source for humans and animals, soybean (Glycine max) has experienced a fast growth with the rapid development of population and economy. Despite broad interest in energy consumption and CO2 emissions generated by soybean production, there are few impact-oriented water footprint assessments of soybean production. This study evaluates the fossil energy, carbon, and water footprints of China’s soybean production so that key environmental impacts can be identified. To provide reliable results for decision-making, uncertainty analysis is conducted based on the Monte Carlo model. Results show that the impact on climate change, ecosystem quality, human health, and resources is $3.33 \times 10^3$ kg CO2 eq (GSD2 = 1.87), $6.18 \times 10^{-5}$ Species·yr (GSD2 = 1.81), $3.26 \times 10^{-3}$ Disability-adjusted Life Years (GSD2 = 1.81), and $81.51$ $ (GSD2 = 2.28)$, respectively. Freshwater ecotoxicity is the dominant contributor (77.69%) to the ecosystem quality category, while climate change (85.22%) is the dominant contributor to the human health category. Key factors analysis results show that diammonium phosphate and diesel, and on-site emissions, are the major contributors to the overall environmental burden of soybean production. Several policy recommendations are proposed, focusing on trade structure optimization, efficient resource use, and technological improvements. Such policy recommendations provide valuable insights to those decision-makers so that they can prepare appropriate mitigation policies.

Keywords Water footprint · Fossil energy footprint · Carbon footprint · Life cycle assessment · Soybeans · China

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1 Introduction

Soybean (*Glycine max*) is a significant protein source for humans and animals (Fried et al., 2018). The global soybean production increased to 336.6 million tons in 2019 from 220.8 million tons in 2005, with increasing demand for food and animal feed. China was the world’s fourth largest soybean producer in 2019, with a yield of 17.1 million tons and accounting for 5.1% of the global production (SOPA, 2020). China is also the largest soybean importer in order to satisfy its soaring demand (Fuchs et al., 2019; Wu et al., 2020). The Chinese government takes active measures to increase its domestic soybean cultivation, with China’s soybean production increasing to 17.1 million tons in 2019 from 11.95 million tons in 2013. However, such large production is based on a large amount of energy and water consumption and induced many environmental issues, such as soil contamination and greenhouse gases (GHGs) emission (Maciel et al., 2016; Taherzadeh & Caro, 2019). In addition, the outbreak of COVID-19 further influenced international trade globally. Consequently, it is critical to evaluate environmental footprints generated from soybean production and increase the self-sufficiency rate of soybean in a sustainable manner.

Academically, environmental footprint is one effective method to measure environmental impacts caused by human activities (Gu et al., 2016; Hoekstra & Wiedmann, 2014). It can quantify different environmental burdens on global sustainability caused by anthropogenic activities so that key problems can be identified (Čuček et al., 2012; Power, 2009). Several carbon and energy footprints studies on soybean production have been conducted. For example, Maciel et al. (2016) and Raucci et al. (2015) measured the GHGs generated from soybean production in Brazil. Arrieta et al. (2018) accounted carbon and energy footprints of Argentina’s soybean production. But few studies have been conducted to calculate water footprint (WF) generated from soybean production.

The WF of one product is a measure of the water consumed and polluted per unit of the product produced (Haghighi et al., 2018; Hoekstra et al., 2011). The International Standardization Organization (ISO) issued ISO 14046 (2014), in which principles, requirements, and guidelines for impact-oriented WF assessments under the framework of life cycle assessment (LCA) can be referred for stakeholders. Different from volume-based WF, LCA-based WF method focuses on the water-related impacts of the studied system rather than the water volume consumed and polluted (Berger & Finkbeiner, 2013). Since the publication of the ISO 14046, LCA-based WF studies on various aspects have been conducted worldwide (Chen et al., 2021), including several in the agricultural sector. For example, Zhai et al. (2019a, b) used the LCA-based WF method to evaluate the WF of wheat production. Bai et al. (2021) evaluated the water-related environmental impacts generated by China’s maize production. Zhang et al. (2021) performed an LCA-based WF analysis of China’s cotton production and found that the impact on human health and ecosystem quality generated by one ton of cotton production was $1.69 \times 10^{-3}$ Disability-adjusted Life Years (DALY) and 152.83 Species·yr, respectively. Up to now, no LCA-based WF studies on soybean production have been documented. A life cycle inventory (LCI) is essential for LCA studies of down-stream products, such as soybean oil, biodiesel, and soybean meal. However, few LCA studies on China’s soybean production have been published, except the one reported by Knudslen et al. (2010), in which the environmental impacts of soybean production in Jilin province of China based on the 2006 data were evaluated.

This study aims to fill these research gaps by evaluating the environmental burden of China’s soybean production based on LCA-based water-, energy-, and carbon footprints analysis. In addition, key factors contributing to these impacts are identified and quantified.
so that potential environmental performance improvements can be explored. Also, uncertainty analysis is performed so that more reliable results for decision-making could be offered. The whole paper is organized as below: After this introduction section, Sect. 2 details research methods and data sources. Then, Sect. 3 presents research results, and Sect. 4 discusses policy recommendations. Finally, research conclusions are drawn in Sect. 5.

2 Methods and data

2.1 Functional unit and system boundary

A functional unit is usually used as a reference unit because it enables all inputs and outputs of one investigated system to be normalized (ISO 14040, 2006; Chen et al., 2015). In this study, the functional unit is defined as one ton of soybean production. The system boundary is set by using the “cradle-to-gate” approach, which means that the transportation and utilization of the soybean is not included in this study. The inputs and outputs associated with soybean production such as the seeds, chemical fertilizer, pesticides, irrigation water, energy, and direct emissions discharged to water are considered. In addition, the transport distances of various raw materials (e.g., seeds, chemical fertilizer, and pesticides) are assumed as 100 km.

2.2 Methods

Impact-oriented WF analysis of soybean production is performed in this study on the base of ISO 14046 (2014). In this study, the impact on water degradation (impact related to water quality) and impact on water availability (impact related to water quantity) are both considered, involving six midpoint categories (i.e., aquatic eutrophication, acidification, freshwater ecotoxicity, carcinogens, non-carcinogens, and water scarcity). Notably, all substances associated with water are considered, while substances that will finally be emitted to other media through medium exchange are not considered.

In terms of water degradation (impacts related to water quality), aquatic eutrophication, acidification, freshwater ecotoxicity, carcinogens, and non-carcinogens are considered. The characterization factor for individual category is evaluated based on Eq. (1). Detailed information of these characterization factors is available in our previous study (Chen et al., 2022).

\[ CF_i = FF_i \times RF_i \]  

where \( CF_i \), \( FF_i \), and \( RF_i \) represents the characterization factor of individual substance \( i \) (all substances associated with water are considered, while substances emitted to air/soil/sediment by medium exchanges are excluded), fate factor of \( i \) in water (calculated by using the Mackay’s multi-media fugacity model), and result factor of \( i \) (calculated by using the exposure model and the dose–response model), respectively.

For the impact on water availability (impact related to water quantity), water scarcity is evaluated based on WAVE+ model (Berger et al., 2018) and regional information (e.g., growing cycle and planting time).
In this study, fossil energy footprint is assessed by adopting ReCiPe model (Huijbregts et al., 2017) and carbon footprint is evaluated based on LCA by applying the impact category of climate change.

For endpoint categories, the categories of human health, ecosystem quality, and resources are considered. Detailed data on the coefficient of midpoint categories to endpoint categories are referred to Huijbregts et al. (2017). In addition, the Monte Carlo simulation is performed for uncertainty analysis (with 1000 trials) so that results obtained from this study can be more reliable.

2.3 Data sources

Table 1 lists the LCI of China’s soybean production. The inputs and outputs data related to soybean production (such as seeds, fertilizers, and diesel) are collected based on the functional unit and system boundary setting in this study. Data on per unit area yield and gross output of soybean are obtained from the China Rural Statistical Yearbooks compiled by the National Bureau of Statistics (NBS, 2019). Data on annual inputs for soybean planting are obtained from public sources, such as the Compile of Cost–Benefit Data of Agricultural Products published by the National Development and Reform Commission (NDRC, 2019) and the China Rural Statistical Yearbooks published by the National Bureau of Statistics (NBS, 2019). Meteorological data (including wind velocity, precipitation, temperature, relative humidity, and sunshine hours) are obtained from the China Meteorological Data Service Center (CMDC, 2016). The pollutants discharged to water from fertilizer and pesticide use are calculated on the basis of the national pollution survey (CAEP, 2003; CFPC, 2009a; CFPC, 2009b).

Table 1 Life cycle inventory of China’s soybean production (functional unit: one ton of soybeans)

| Unit Amount           | Unit     | Amount       |
|-----------------------|----------|--------------|
| Raw materials consumption |          |              |
| Irrigation water kg   |          | 9.66×10⁵     |
| Effective rainfall kg |          | 1.76×10⁶     |
| Urea kg               |          | 17.71        |
| Ammonia kg            |          | 13.79        |
| Diammonium phosphate kg|          | 43.24        |
| Potash kg             |          | 17.68        |
| Seed kg               |          | 41.28        |
| Diesel kg             |          | 35.44        |
| Electricity kWh       |          | 24.43        |
| Pesticides kg         |          | 1.56         |
| Emissions to water    |          |              |
| Ammonia, as N kg      |          | 2.74×10⁻²    |
| Total phosphorus kg   |          | 3.77×10⁻³    |
| Total nitrogen kg     |          | 8.05×10⁻²    |
| Chemical Oxygen Demand |          | 79.34        |
| Chlorpyrifos kg       |          | 2.32×10⁻⁴    |
| Acetochlor kg         |          | 1.18×10⁻³    |
Background data (such as electricity generation and potash fertilizer production) for the LCI of soybean production (Table 1) are taken from the China process-based LCI database (CPLCID), except for the data of pesticides. European database which are available from the Ecoinvent centre (2015) is adopted for pesticide production due to the lack of Chinese data. During this process, the Chinese data such as electricity generation are used to replace those corresponding European data so that impacts from regional disparity can be minimized.

3 Results

3.1 Midpoint results

Table 2 lists environmental impacts generated by one ton of China’s soybean production at the midpoint level. This table also lists the results of uncertainty analysis so that more reliable results can be provided to different stakeholders. For example, the impact on aquatic eutrophication generated by one ton of soybean production is 1.21 kg PO$_4^{3-}$ eq, and the corresponding squared geometric standard deviation (GSD$^2$) is 1.71. These results suggest that the impact on aquatic eutrophication varies from 0.71 PO$_4^{3-}$ eq to 2.07 PO$_4^{3-}$ eq, at the 95% confidence interval. Similarly, results for other categories are obtained from Table 2.

3.2 Endpoint results

The environmental impacts of soybean production at the endpoint level are summarized in Table 3. Results show that the impact on ecosystem quality, human health, and resources is 6.18 × 10$^{-5}$ Species·yr, 3.26 × 10$^{-3}$ DALY, and 81.51 $, respectively. The uncertainty analysis on the endpoint results indicates that the impact on ecosystem quality ranges from 3.41 × 10$^{-5}$ Species·yr to 1.12 × 10$^{-4}$ Species·yr, the impact on human health ranges from 1.80 × 10$^{-3}$ DALY to 5.90 × 10$^{-3}$ DALY, and the impact on resources ranges from 35.75 $ to 185.84 $, at the 95% confidence interval.

| Impact categories         | Unit          | Amount       | GSD$^2$ |
|---------------------------|---------------|--------------|---------|
| Aquatic eutrophication    | kg PO$_4^{3-}$ eq | 1.21         | 1.71    |
| Acidification             | kg SO$_2$ eq  | 2.62         | 1.75    |
| Carcinogens               | Case          | 7.71 × 10$^{-6}$ | 2.66    |
| Non-carcinogens           | Case          | 3.48 × 10$^{-5}$ | 3.91    |
| Freshwater ecotoxicity    | PAF·m$^3$·d   | 1.68 × 10$^4$ | 1.88    |
| Water scarcity            | m$^3$ deprived | 456.23       | 1.22    |
| Climate change            | kg CO$_2$ eq  | 3.33 × 10$^3$ | 1.87    |
| Fossil depletion          | kg oil eq     | 343.37       | 2.46    |

GSD$^2$: squared geometric standard deviation
Table 3  Endpoint results of China’s soybean production (functional unit: one ton of soybeans)

| Impact categories      | Unit              | Amount      | $GSD^2$ |
|------------------------|-------------------|-------------|---------|
| Ecosystem quality      | Species·yr        | $6.18 \times 10^{-5}$ | 1.81    |
| Human health           | DALY              | $3.26 \times 10^{-3}$ | 1.81    |
| Resources              | $                 | 81.51       | 2.28    |

$GSD^2$: squared geometric standard deviation

Figure 1 shows that the impact of freshwater ecotoxicity is the dominant contributor (77.69%) to the endpoint category of ecosystem quality, followed by the impact of climate change (15.08%) and water scarcity (5.68%). For human health, the impact of climate change (85.22%) is the dominant contributor, followed by the impact of water scarcity (9.17%). For resources category, fossil energy footprint is considered, and it contributes to an impact of 81.51 $.
3.3 Dominant contributors

3.3.1 Dominant contributors to midpoint results

Figure 2 illustrates the contributions of key substances and processes to the midpoint categories. Irrigation water, mainly used in the direct planting process and production of soybean seeds, is the dominant contributor to the water scarcity category. For acidification category, sulfur dioxide to air mainly generated from diesel and diammonium phosphate production is dominant contributor, while it is Chemical Oxygen Demand (COD) to water for aquatic eutrophication category. For freshwater ecotoxicity category, direct chlorpyrifos emission and iron emitted to soil are dominant contributors, with copper emitted to air...
playing an additional contribution. The iron emitted to soil is mainly generated from the production of diesel and the copper to air is mainly generated from the production of diammonium phosphate and diesel. For climate change category, the contribution of carbon dioxide is mainly generated from the production of diesel and diammonium phosphate. For non-carcinogens, arsenic to air mainly caused by the production of diesel is the dominant contributor, while chromium to water and chromium to soil, mainly generated from the production of diesel and diammonium phosphate, are dominant contributors for carcinogens. For fossil depletion category, the dominant contributors are coal in the ground mainly caused by diammonium phosphate and diesel production, as well as oil in the ground mainly caused by diesel production.

3.3.2 Dominant contributors to endpoint results

Figure 3 illustrates the dominant contributors to the endpoint results of the soybean production. The results indicate that the carbon dioxide to air is the dominant contributor to human health, with irrigation water and the methane to air making additional contributions. These results also indicate that the production of diesel and diammonium phosphate are the dominant sources of carbon dioxide to air and methane to air. For resources category, the dominant contributor is oil in the ground, with coal in the ground making an additional contribution. These substances are mainly caused by the production of diesel and diammonium phosphate. For ecosystem quality, iron to soil, carbon dioxide to air, chlorpyrifos to water, and copper to air are the dominant contributors, with irrigation water, copper to soil, and strontium to water making additional contributions. These substances are mainly from the production of diammonium phosphate and diesel, as well as on-site emissions generated by soybean production.

3.4 Temporal and spatial analysis

Figure 4 illustrates the environmental impacts (midpoint level) of China’s soybean production from 2009 to 2018. It is clear that the impacts on aquatic eutrophication, acidification, carcinogens, non-carcinogens, freshwater ecotoxicity, climate change, and fossil depletion in 2016 are higher than those in other years. The main reason is that the diesel consumption for per ton of soybean production in 2016 was much higher than other years, which is consistent with the results that diesel is a significant contributor to environmental footprints generated from soybean production (Fig. 2). The impact on water scarcity category fluctuated with the amount of irrigation water used for soybean production.

The spatial distribution of soybean production and corresponding productivity in 2018 are shown in Fig. 5. It is clear that Heilongjiang and Inner Mongolia were the top two soybean producing provinces in China in 2018 (Fig. 5a), with the yield of 6.58 million tons and 1.79 million tons, respectively. The total environmental impacts of soybean production in Heilongjiang and Inner Mongolia are also higher than other provinces due to their higher soybean yields. With regard to productivity, the top two provinces in China include Tibet and Hainan (Fig. 5b), with the productivity of 3.23 ton/ha and 3.18 ton/ha in 2018, respectively. The productivity of soybeans of Heilongjiang and Inner Mongolia is 1.84 ton/ha and 1.64 ton/ha, respectively, much lower than Tibet and Hainan. If their soybean productivity can reach 3.00 ton/ha, then the national soybean production would be increased by 35.16%.
Fig. 3 Dominant contributors to endpoint categories
3.5 Environmental impacts of soybean production

The total impact of China’s soybean production on aquatic eutrophication, acidification, carcinogens, non-carcinogens, freshwater ecotoxicity, water scarcity, climate change, fossil depletion, human health, ecosystem quality, and resources is $1.93 \times 10^7 \text{ kg PO}_4^{3-}$ eq, $4.18 \times 10^7 \text{ kg SO}_2$ eq, $123.13 \text{ Case}$, $555.76 \text{ Case}$, $2.68 \times 10^{11} \text{ PAF} \cdot \text{m}^3 \cdot \text{d}$, $7.29 \times 10^9 \text{ m}^3$ deprived, $5.32 \times 10^{10} \text{ kg CO}_2$ eq, $5.48 \times 10^9 \text{ kg oil eq}$, $5.21 \times 10^4 \text{ DALY}$, $986.95 \text{ Species-yr}$, and $1.30 \times 10^9 \text{ $ for the year 2018, respectively, with the total soybean yield of 15.97 million tons (1.898 ton/ha) in the same year.}
The environmental impacts of soybean production in the top three production provinces (i.e., Heilongjiang, Inner Mongolia, and Anhui) for year 2018 are listed in Table 4. It is clear that such impact in Anhui is lower than the national average, with the exception of the aquatic eutrophication category. However, such impact in Inner Mongolia is higher than the national average in all the categories except for the fossil depletion category and acidification category. For Heilongjiang, the impacts of acidification, climate change, fossil depletion, carcinogens, non-carcinogens, human health, and resources are higher than the national averages, while impacts of aquatic eutrophication, freshwater ecotoxicity, water scarcity, and ecosystem quality are lower than the national averages. These differences can be explained by region-specific soybean productivity, fertilization application, and diesel efficiency.

Carbon and fossil energy footprints of soybean production in various Chinese regions are listed in Table 5. Such impact in Shanxi is the highest, followed by Hebei. Wu et al. (2020) also found that the environmental impacts caused by soybean production in Shanxi and Hebei were higher than the national average. The carbon and fossil energy footprint of the production of one ton of soybeans in Jilin is $1.98 \times 10^3$ kg CO$_2$ eq and $178.55$ kg oil (7.50 GJ) in this study, higher than those in the previous published studies (Knudsen et al., 2010; Luo et al., 2011). By using data from questionnaires and interviews collected from 15 conventional farms in Jilin province of China for the year 2006, Knudsen et al. (2010) found that the GHG and non-renewable use from soybean production was $0.263$ ton CO$_2$ eq/ton and $1.71$ GJ/ton, respectively. Similarly, Luo et al. (2011) found that the GHG from soybean production in Jilin was $1.53$ ton CO$_2$ eq/ton, in which their data were collected for the year 2006–2007. These results differences can be explained by the increase in diesel consumption caused by the improvement of agricultural mechanization. It is worth noting that the carbon footprint ($0.38$ ton CO$_2$ eq/ton soybeans) and fossil energy footprint ($2.30$ GJ/ton soybeans) of soybean production in Chongqing is much lower than other Chinese regions (Table 5) due to its less diesel consumption for soybean production ($0.96$ kg diesel/ton soybean). Different from Knudsen et al. (2010) and Luo et al. (2011), statistical

| Impact categories | Unit                   | Heilongjiang | Inner Mongolia | Anhui | Shandong | National average |
|-------------------|------------------------|--------------|----------------|-------|----------|------------------|
| Aquatic eutrophication | kg PO$_4^{3-}$ eq | 1.12         | 1.59           | 1.41  | 0.87     | 1.21             |
| Acidification | kg SO$_2$ eq             | 3.04         | 2.59           | 1.46  | 1.34     | 2.62             |
| Carcinogens | Case                    | 8.54×10$^{-6}$ | 9.97×10$^{-6}$ | 6.00×10$^{-6}$ | 2.94×10$^{-6}$ | 7.71×10$^{-6}$ |
| Non-carcinogens | Case                  | 3.74×10$^{-5}$ | 4.84×10$^{-5}$ | 3.02×10$^{-5}$ | 1.21×10$^{-5}$ | 3.48×10$^{-5}$ |
| Freshwater ecotoxicity | PAF·m$^3$·d           | 1.41×10$^4$ | 4.42×10$^4$    | 9.26×10$^3$ | 4.86×10$^3$ | 1.68×10$^4$ |
| Water scarcity | m$^3$ deprived           | 350.76       | 882.99         | 434.90 | 389.02   | 456.23           |
| Climate change | kg CO$_2$ eq             | 3.90×10$^3$  | 3.52×10$^3$    | 2.01×10$^3$ | 1.49×10$^3$ | 3.33×10$^3$   |
| Fossil depletion | kg oil eq               | 412.99       | 318.29         | 170.69 | 173.77   | 343.37           |
| Human health | DALY                    | 3.68×10$^{-3}$ | 3.76×10$^{-3}$ | 2.11×10$^{-3}$ | 1.57×10$^{-3}$ | 3.26×10$^{-3}$ |
| Ecosystem quality | Species·yr            | 5.49×10$^{-5}$ | 1.44×10$^{-4}$ | 3.60×10$^{-5}$ | 2.15×10$^{-5}$ | 6.18×10$^{-5}$ |
| Resources | $                      | 89.54        | 106.19         | 65.69 | 30.95    | 81.51            |
| Province        | Productivity (ton/ha) | Climate change (kg CO₂ eq/ton soybeans) | Climate change (kg oil eq/ton soybeans) |
|-----------------|-----------------------|-----------------------------------------|----------------------------------------|
| Chongqing       | 2.05                  | 381.23                                  | 54.69                                  |
| Shaanxi         | 1.58                  | 1.57                                    | 439.08                                 |
| Shanxi          | 1.57                  | 3.74×10³                               | 566.12                                 |
| Shaanxi         | 2.82                  | 5.03×10³                               | 566.12                                 |
| Shanxi          | 1.64                  | 1.49×10³                               | 566.12                                 |
| Henan           | 2.45                  | 3.52×10³                               | 566.12                                 |
| Henan           | 1.97                  | 3.04×10³                               | 566.12                                 |
| Heilongjiang    | 2.48                  | 1.98×10³                               | 566.12                                 |
| Heilongjiang    | 1.84                  | 3.90×10³                               | 566.12                                 |
| Hebei           | 1.50                  | 1.63×10³                               | 566.12                                 |
| Anhui           | 2.42                  | 4.01×10³                               | 566.12                                 |
| Henan           | 2.48                  | 1.01×10³                               | 566.12                                 |
| Henan           | 2.48                  | 2.01×10³                               | 566.12                                 |
| Heilongjiang    | 2.48                  | 2.01×10³                               | 566.12                                 |
| Heilongjiang    | 2.48                  | 2.01×10³                               | 566.12                                 |
Soybean productivity is influenced by various factors, such as solar radiation, local precipitation, and cultivation technology (Zhang et al., 2017). Soybean productivity in Heilongjiang (1.84 ton/ha), Inner Mongolia (1.64 ton/ha), and Anhui (1.50 ton/ha) is much lower than the national average level (NBS, 2019), while Tibet (3.23 ton/ha), Hainan (3.18 ton/ha), and Shandong (2.82 ton/ha) have the highest soybean productivity in 2018 (NBS, 2019). Also, the proportions of soybean planting area to the total agricultural area in the top three productivity regions (Tibet 1.66%, Hainan 0.84%, and Shandong Province 1.43%) are much lower than that of Heilongjiang (25.50%), Inner Mongolia (14.82%), and Anhui (7.41%). In addition, the impacts of soybean production in Shandong is much lower than the national average (Table 4). Such findings indicate that sustainable soybean production policies should be made from a holistic perspective rather than only focusing on one factor. This is especially true for regions where soybean yield is low and corresponding environmental impact is high. Consequently, such policies should be made by considering the local endowments and the holistic agricultural development.

4 Policy recommendations

The results from this study provide valuable insights to all the stakeholders related with soybean production. Based on the Chinese realities, the following policy recommendations are raised.

4.1 Optimizing soybean trade structure

Figure 6 illustrates China’s major soybean trade partners based on the 2016 data. Brazil, the USA, and Argentina are the major soybean exporters (thick arrow in Fig. 6). The eastern Chinese provinces are the major soybean importers, with Shandong being the largest one. For most Chinese provinces, the USA and Brazil are the major soybean trade partners, while it is Russia for Heilongjiang. China imported a total of 88.51 million tons of soybeans in 2019 (Yang et al., 2021), mainly from Brazil, the USA, and Argentina. However, the trade skirmishes between China and the USA has influenced such a soybean supply chain since 2017 (Geng & Sarkis, 2018), decreased from 34.17 million...
tons (40.72% of China’s total import) in 2016 to 16.94 million tons (19.1% of China’s total import) in 2019. This indicates the uncertainty of political factor and suggests China to diversify its soybean trade partners.

Initiated and promoted by the Chinese government, the Belt and Road initiative aims to promote broad cooperation among the countries locating in the Silk Road Economic Belt and the 21st Century Marine Silk Road (Chen et al., 2018). Many Belt and Road countries have large agricultural land and can plant more soybean to support their economy. However, they lack advanced cultivation technologies and feel reluctant to expand their soybean production. Therefore, the Chinese government should encourage technology transfer so that these countries can improve their soybean production efficiency and increase their soybean yields. Joint research activities and technical secondments are useful to facilitate such cooperation. For example, the maize–soybean strip intercropping technology has been successfully applied in Pakistan, which is an active response to the Sustainable Development Goals (SDG 2 food), with its benefits of achieving food security and improving nutrition. It is essential for China to further share its advanced technologies with those Belt and Road countries so that these countries can increase their economic revenue. In addition, it is necessary to establish an open data platform so that all the involved countries can share relevant data on soybean production. In particular, such a platform can help identify the key environmental challenges from soybean production and seek possible mitigation pathways.

4.2 Improving the efficiency of resource utilization

Results from this study (Figs. 2 and 3) indicate that fertilizer is a key contributor to environmental impact generated from soybean production. China is the largest fertilizer consumer in the world (Chen et al., 2018). The overuse of fertilizer has resulted in several environmental issues, such as soil contamination, groundwater contamination, soil infertility, and eutrophication (Zhai et al. 2019a). Statistical data show that 66.98 kg of fertilizer was used for producing one ton of soybean in 2018 (NBS, 2019). In Heilongjiang and Inner Mongolia, such figure is 76.31 kg and 83.82 kg, respectively, higher than the national average (NBS, 2019). In particular, as one province with severe water shortage and tough weather conditions, the low quality cultivated land accounts for 87.34% of its total cultivated land in Inner Mongolia (Zheng, 2018). Consequently, it is critical for such provinces to improve their fertilizer efficiency so that the overuse of fertilizer can be avoided or at least mitigated.

Returning straw to the fields is one effective way of reducing the use of fertilizer (Zhai et al. 2019a). Liu et al. (2019) found that returning straw to the fields increased the sustainable level of soybean production. If the consumption of diammonium phosphate is reduced by 10%, its environmental impact on acidification, climate change, fossil depletion, and human health can be reduced by 3.35%, 3.53%, 4.53%, and 3.05%, respectively. In addition, precision agriculture, such as testing the soil quality before the application of fertilizer, can not only improve the efficient use of fertilizer, but also increase economic and environmental benefits of soybean production (Huang et al., 2017; Zhai et al. 2019a). However, such application depends on financial support since the value of straw is so cheap and the collection and delivery of such straw is so difficult that most farmers do not like to engage in such efforts. It would be useful for the Chinese governments to provide financial subsidies to support the application of such innovation.
Another finding from this study is that diesel is a significant contributor to the environmental impact from soybean production. For instance, the impact generated by diesel production accounts for 77.76% and 90.76% of the impact on carcinogens category and non-carcinogens category, respectively. Also, the diesel consumption for soybean production in Heilongjiang and Inner Mongolia is higher than the national average. Therefore, it is crucial for such provinces to improve their diesel efficiency for soybean production. In this regard, biodiesel is one renewable alternative and can address climate change (Hasan & Rahman, 2017). For example, waste cooking oil can be used to generate biodiesel, which has been widely promoted (Goh et al., 2020). If 10% of diesel produced by the petroleum industry can be replaced by biodiesel produced from waste cooking oil, the environmental impacts on aquatic eutrophication, acidification, carcinogens, non-carcinogens, freshwater ecotoxicity, water scarcity, climate change, fossil depletion, human health, ecosystem quality, and resources would be reduced by 0.22%, 4.41%, 7.62%, 9.05%, 5.45%, 0.02%, 5.36%, 4.11%, 5.03%, 5.10%, and 8.02%, respectively. These findings indicate that promoting the use of biodiesel is beneficial to alleviating the stress on energy, water, and GHGs generated from the agriculture sector. However, the collection of waste cooking oil is difficult. Thus, innovative policies should be implemented so that the efficient collection of waste cooking oil can be promoted.

4.3 Supporting related research and capacity-building activities

China’s soybean productivity (1.89 ton/ha for 2019) is much lower than that in the USA (3.40 ton/ha for 2019), Brazil (3.26 ton/ha for 2019), and Argentina (3.33 ton/ha for 2019) (SOPA, 2020). Consequently, it is crucial for China to improve its soybean production technologies. The total production of the maize-soybean intercropping system is higher than monoculture of either maize or soybean (Yang et al., 2015). Thus, the maize–soybean strip intercropping system is one promising, efficient and sustainable planting system and should be promoted (Iqbal et al., 2019). The Ministry of Agriculture and Rural Affairs (MARA) of China released the national guidelines for promoting maize–soybean strip intercropping technology based on the local climate and soybean production features in different Chinese provinces (MARA, 2020), in which the detailed parameters, such as the space of the neighboring rows, seeding density, and the cultivation time are all provided (Yang et al., 2015). Capacity-building efforts are necessary so that all the local stakeholders can quickly learn and apply such new technologies. In particular, those western provinces such as Gansu, Yunnan, and Guizhou do not have adequate financial resources to apply such technologies. Under such a circumstance, those eastern Chinese provinces should support their western counterparts by providing financial and technological support.

In addition, green consumption behaviors should be promoted in China. Soybean has been consumed mainly for edible oil refinery and fodder in China. If low-protein diet can be promoted across the whole country, then the total soybean consumption may be reduced. In fact, the technology of low-protein diet, which can reduce the consumption of soybean meal and mitigate the environmental impacts, has been promoted in China. However, because it is affected by traditional concepts of feeding, reducing the demand for soybean meal as animal feed in China remains difficult to achieve in the short term. Therefore, it is necessary for the Chinese government to promote nutritious diets across the whole country. In this regard, education is key so that more Chinese citizens can better understand how to prepare appropriate diets. Regular workshops, TV and Internet promotions, pamphlets are useful measures to promote low carbon diets.
5 Conclusions

Soybean production is essential for supporting modern societies, although it is based upon a large amount of energy and water consumption and causes serious environmental emissions. China is the largest soybean consumption country in the world. The aim of this study is to provide a combined analysis based on the assessment of different footprints from soybean production so that the corresponding environmental impacts could be quantified.

The results show that the impact of soybean production on aquatic eutrophication, acidification, carcinogens, non-carcinogens, freshwater ecotoxicity, water scarcity, climate change, and fossil depletion is 1.21 kg PO$_4^{3-}$ eq (GSD$^2$ = 1.71), 2.62 kg SO$_2$ eq (GSD$^2$ = 1.75), 7.71 × 10$^{-6}$ Case (GSD$^2$ = 2.66), 3.48 × 10$^{-5}$ Case (GSD$^2$ = 3.91), 1.68 × 10$^4$ PAF·m$^3$·d (GSD$^2$ = 1.88), 456.23 m$^3$ deprived (GSD$^2$ = 1.22), 3.33 × 10$^3$ kg CO$_2$ eq (GSD$^2$ = 1.87), and 343.37 kg oil eq (GSD$^2$ = 2.46), respectively. The key factors analysis results indicate that the production of diammonium phosphate and diesel, as well as the on-site emissions during soybean production are major contributors to the environmental impacts from soybean production. In addition, the impacts on acidification, carcinogens, non-carcinogens, freshwater ecotoxicity, human health, ecosystem quality, resources, climate change, and fossil depletion could be reduced by 4.41%, 7.62%, 9.05%, 5.45%, 5.03%, 5.10%, 8.02%, 5.36%, and 4.11%, respectively, if 10% diesel from the petroleum industry was replaced by biodiesel produced from waste cooking oil. Several policy recommendations are proposed by considering the Chinese realities, such as optimizing soybean trade structure, improving resource efficiency, and supporting technological development. These results can also expand the LCI database of China’s soybean production and provide useful insights for various stakeholders to improve the overall environmental performance of soybean production.

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Declarations

Conflict of interest The authors declare that they have no conflict of interest.

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