

See discussions, stats, and author profiles for this publication at: <https://www.researchgate.net/publication/51029014>

Environmental Impacts of Products in China

ARTICLE in ENVIRONMENTAL SCIENCE & TECHNOLOGY · APRIL 2011

Impact Factor: 5.33 · DOI: 10.1021/es103206g · Source: PubMed

CITATIONS

13

READS

90

2 AUTHORS:



Yi Yang

13 PUBLICATIONS 113 CITATIONS

SEE PROFILE



Sangwon Suh

University of California, Santa Barbara

130 PUBLICATIONS 4,820 CITATIONS

SEE PROFILE

Environmental Impacts of Products in China

Yi Yang[†] and Sangwon Suh^{‡,*}

[†]Bioproducts and Biosystems Engineering Department, University of Minnesota, St. Paul, Minnesota 55108, United States

[‡]Bren School of Environmental Science and Management, University of California, 3422 Bren Hall, Santa Barbara, California 93106-5131, United States

S Supporting Information

ABSTRACT: As the Chinese economy has become an integral part of the global supply chain, quantifying the environmental impacts by Chinese industry is indispensable to understanding the environmental performance of products in general. Comprehensive and consistent environmental data infrastructure, however, is lacking in China, hindering such an understanding. In this paper, we demonstrate a simplified method for assembling and harmonizing various data sources to develop a sectoral environmental database for input-output life cycle assessment (IO-LCA). We first identified key substances by analyzing previous normalization studies and other countries' sectoral environmental databases. Data for priority substances were compiled and then adjusted and validated. The database created from this process was then used to analyze the direct and indirect environmental impacts generated by Chinese rural and urban consumptions. Expenditures on food and other basic household needs like heating and cooking were found to play a dominant role in generating environmental impacts in China, while previous studies of industrialized countries also highlighted the importance of transportation. This database provides background information for LCA through, for example, the hybrid approach, and is also conducive to ongoing efforts to develop generic life cycle inventory databases in China.



1. INTRODUCTION

Since the initiation of economic reform in the late 1970s, the past three decades have witnessed China's remarkable economic development. China's Gross Domestic Product (GDP) has grown at an average annual rate of 9–10% from 1979 to 2004.¹ Yet, underlying the halo of this so-called economic “miracle”² are growing environmental problems facing Chinese society.^{3–6}

While China's environmental issues have been frequently cited in research literature, previous studies focus either on anecdotal cases or a single category of environmental impact.^{6–8} A comprehensive analysis of a broad range of life-cycle environmental impacts would be indispensable not only for policy-makers when prioritizing policy options but also for Life Cycle Assessment (LCA) practitioners seeking to quantify the environmental implications of products manufactured in China. Such an analysis requires addressing the following three issues:⁹ (1) quantifying the environmental emissions generated directly from industries, (2) modeling the structure of the supply chain through which products are exchanged, and (3) quantifying the environmental impacts generated by various environmental emissions.

In previous studies in Europe and North America, these three issues were addressed by (1) sectoral environmental databases, (2) input–output accounts, and (3) life cycle impact assessment (LCIA).^{9–13} The Environmental Impacts of Product (EIPRO) study,¹⁰ for example, modified the Comprehensive Environmental Data Archive (CEDA) of the U.S. to identify the products

with major life-cycle environmental impacts in the EU25. The Vision 2020 study by the U.S. Environmental Protection Agency (EPA)¹¹ used the CEDA U.S. database, water consumption data and material flow data to prioritize the products in the U.S. of greatest environmental concern.

In the case of China, however, lack of sectoral environmental data is a challenge when undertaking such an analysis. For instance, the Chinese Environmental Statistics Yearbook, published by the National Bureau of Statistics (NBS), covers only a handful of monitored environmental interventions by sector.¹⁴ While there are many individual studies addressing various environmental emissions, they are highly fragmented, with each employing different industry sector classification.^{7,15–17}

This paper demonstrates a method to efficiently construct sectoral environmental database for China that covers a wide range of environmental impacts under conditions of limited data availability. The compiled database is then used to analyze life-cycle environmental impacts of products consumed by urban and rural households in the Chinese economy using the method implemented in the EIPRO study.¹⁰

Received: September 20, 2010

Accepted: March 24, 2011

Revised: March 11, 2011

Published: April 06, 2011

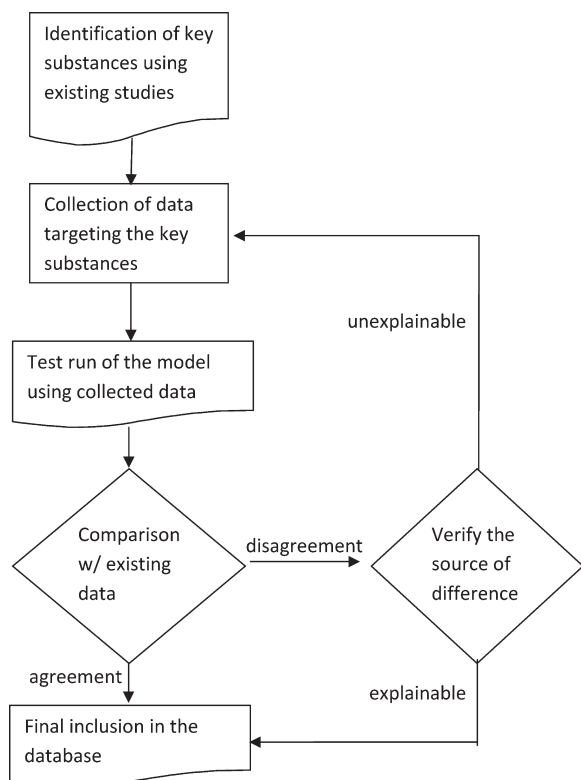


Figure 1. Overall process flow for compiling sectoral environmental data for China.

2. DATABASE DEVELOPMENT AND VALIDATION

Given that collecting and transforming individual studies into a harmonized and consistent database can be a time-consuming endeavor, we first examined existing reference databases that cover various environmental emissions. We then identified the key substances for each impact category for which cumulative contributions constitute at least 90% of the total impact.

Previous studies on normalization references and national environmental input-output databases list over a thousand Life Cycle Inventory (LCI) items or environmental interventions.^{9–11,18,19} LCI results are rarely used alone; however, a characterization step is required to translate the long list of LCI items into category results. Priority substances can therefore be identified by analyzing the characterized results of previous studies. We used characterization factors developed by the Institute of Environmental Sciences (CML) to identify priority substances for each impact category.^{20,21} For instance, CEDA 3.0 covers 44 greenhouse gases, which can be aggregated based on their Global Warming Potential (GWP). If expressed in GWP, only three of them, carbon dioxide, methane and nitrous oxide, make up over 90% of the total climate change impact by the GHG emissions compiled in the database. Likewise, there are generally less than 6 substances that constitute over 90% of each impact (Table S1 in Supporting Information (SI)).

Once preliminary data for the priority substances identified were compiled, the resulting data sets were then connected to the Chinese input-output table to calculate the life-cycle environmental impacts of products in China. These impact scores were subsequently compared with those of existing databases for validation. The results for the Chinese sectoral database,

however, were not expected to be exactly the same as those for other countries given the differences in efficiency, technology mix, product mix and regulations. The comparison was used rather as a litmus test to suggest the need for further investigation into the sources of discrepancy. If the differences could be reasonably explained as a result of such investigation, the data sets at hand were absorbed into the final database. If the difference could not be explained, effort was made to collect alternative data. The entire process of data compilation, which is by nature iterative, is shown in Figure 1.

2.1. Input–Output Accounts. We chose Chinese 2002 input–output tables (IOTs) as the basis for supply chain data, in which 122 sectors are distinguished.²² Chinese IOTs are in industry-by-industry format, whereas the United Nations Systems of National Accounts (SNA) recommends the use of supply and use framework that is more suitable for IO-based LCA studies focusing on functionality.²³ While Chinese IOTs from the year 2007 are currently available, we decided to use the 2002 tables due to limited environmental data for the year 2007.

2.2. CO₂, SO₂, and NO_x. These emissions were inventoried in Peters et al.,¹⁶ and used in several critical analyses.^{8,24,25} The Chinese Energy Statistics Yearbook provides the basic energy data for the inventory; however, it only distinguishes between 44 sectors, corresponding to the Chinese NBS classification system. This represents a much higher level of aggregation than that which exists in the 2002 Chinese IOT.

Therefore, we disaggregated these air emissions from 44 into 122 sectors based on the volume of fossil fuel consumption data from Chinese 2002 IOT. Underlying this approach is the assumption of homogeneity; fuel products aggregated into each energy sector have similar market prices and emission factors. While this assumption may provide a reasonable approximation, it introduces uncertainties and inaccuracies, particularly for energy sectors with a high degree of heterogeneity such as the petroleum sector.

These data went through a series of consistency checks with balancing operations, and various other adjustments were made as necessary. One of the major adjustments made was for the data on coal consumption and its associated air emissions. Historical data sources for coal usage in China have been found to be inconsistent, a fact which has been widely questioned and investigated in previous studies.^{26–29} Coal usage was adjusted based on its empirical correlation with Chinese electricity generation (for details, see SI). Another similar adjustment was made on the initial SO₂ emissions, where no abatement technology was assumed in Peters et al.¹⁶ In reality the flue gas desulfurization (FGD) technologies have been applied in China. Abatement data of SO₂ emissions by sector were compiled from the Chinese environmental statistics yearbook¹⁴ to reflect actual SO₂ emissions.

2.3. CH₄, N₂O, and Particulate Matters (PM). CH₄, N₂O, and particulate matter (PM) were derived from the Greenhouse Gas and Air Pollution Interaction and Synergies (GAINS) model.³⁰ The sector classification in GAINS is not entirely consistent with that used in the Chinese IOT; however, for most of them, particularly the major polluters, they were observed to be consistent or even at a higher sectoral resolution. Further modifications were also required as the GAINS model only displays its results in 5-year intervals for the time period 1990–2030, leaving the data of 2002 not directly accessible. Given no further information, emissions for 2002 were estimated by interpolation of 2000 and 2005 data.

2.4. Pollutants Released to Water. The Chinese NBS issues environmental statistics yearly. This annual publication reports monitoring data by sector for a number of toxic water pollutants including: mercury, cadmium, chrome, lead, arsenic, volatile hydroxybenzene, and cyanide.¹⁴ Sectoral classification in the environmental yearbook follows the NBS standard—meaning that the sectors are more coarsely aggregated than in the Chinese IOT. As a result, allocation was conducted in consultation with CEDA database (see SI for a specific example).

2.5. Atmospheric Hg, As, HF, and CO. Mercury (Hg), hydrogen fluoride (HF), and arsenic (As) are important sources of human- and eco-toxicity, while Carbon Monoxide (CO) contributes significantly to photochemical oxidization (Supporting Information Table S1). Estimation of atmospheric Hg emissions was drawn upon an inventory developed by Wu et al.¹⁵ Estimation of both atmospheric As and HF was based on coal combustion, with emission factors taken from literature.^{17,31} Data for CO emissions were derived from Streets et al.,⁷ which estimated sectoral CO emissions based also on emission factors and fuel consumption levels.

Because of lack of data, however, estimation for HF was based exclusively on coal use, thus neglecting other routes of pollution including processing of fluoride-containing minerals, smelting reduction of metals using fluoride-containing fluxes, or the fabrication of brick or other ceramic materials.³² Consequently, the impact results for human- and eco-toxicity categories of the current study, in which HF is a key substance, likely underestimate the true impacts.

2.6. Agricultural Phosphorus Emissions. Emissions of phosphorus to water bodies make up a substantial contribution to the overall eutrophication impact. Data for phosphorus emissions in China were compiled from Chen et al.,³³ and directly assigned to two sectors in the Chinese IOT: *Crop cultivation* and *Livestock and livestock products*. As only agricultural activities were studied in their research, other industrial activities emitting phosphorus were unaccounted for, such as phosphorus mineral and phosphorus fertilizer processing and the like. Results from CDEA 3.0, however, demonstrates that over 95% of phosphorus emissions come directly from these two sectors in the U.S. economy.

2.7. Validation by Comparison with Existing Databases. To test the reliability of the Chinese sectoral database, CEDA U.S. was once again used as a standard for comparison. The intrinsic discrepancies between these two databases because of differences in technology, level of detail, comprehensiveness and regulatory standards may not permit direct one-to-one comparisons. Nor could their total impacts in an absolute sense be compared because of the differences in the composition and overall output levels of the two economies. To maximize comparability, we used the characterized impact per dollar, or characterized impact intensity, as the basis for comparison.

Characterized impact intensity denotes the total amount of environmental impacts directly and indirectly attributed to a given sector/commodity in delivering one dollar of its output to final demands. This value is derived by

$$q = CB(I - A)^{-1}y \quad (1)$$

where q indexes characterized impact, C is the characterization factor matrix, and B represents the environmental interventions matrix, derived from the sectoral database. The matrix, $(I - A)^{-1}$, is the Leontief inverse, through which both direct and indirect effects are traced.³⁴ Final demand is denoted by y , which

may include private and governmental consumption, export, and capital formation.

Comparing the results from the two databases, we found that similar patterns appear in the impacts for which environmental intervention data were relatively well covered in the Chinese sectoral database. For global warming, eutrophication, and acidification, all of the key substances identified from CEDA were included in the Chinese sectoral database developed. Thus, not only is there a similar pattern of priority in these impacts, but also the value of characterized impact intensity is relatively close between the Chinese sectors and corresponding U.S. commodities. By contrast, for other impacts like human toxicity for which the key substances from CEDA were only partially covered, the pattern could barely be seen and significant discrepancies exist in the intensities between the two databases. To illustrate the preceding discussion, a specific example is provided below.

Listed in Table 1 are the twenty sectors/commodities from each database with the highest embodied intensity. A similar pattern is observed in the rank of the sectors/commodities. For instance, two commodities, *Lime*, and *Cement, hydraulic* from CEDA and *Cement, lime and plaster products* from the Chinese IOT resulted in the largest amount of GHG emissions per dollar. The commodity, *Electric services (utilities)*, resembles the Chinese sector *Electricity and steam production and supply*, and both took the second place. The commodity *Sanitary services, steam supply, and irrigation systems* from CEDA matches two Chinese sectors, *Environmental resources and public infrastructure* and *Water production and supply*. The similarity is also mirrored in commodities like crops, livestock, coal, fertilizer, etc.

This pattern can also be used to illuminate some of the important differences between the environmental impacts associated with the various sectors within the two economies. Of interest are at least two observations. First, electricity generation in China showed much larger GHG emissions than its counterpart in the U.S. Second, the Chinese iron and steel industry (in bold in Table 1) stood out as one of the top GHGs emitters yet corresponding sector in the U.S. did not make up a significant portion of the U.S. GHG emissions. The first observation can be explained through a decomposition of the structure of the electric power grid in both economies. Over 80% of electricity generated in China is derived from coal, whereas only 50% of the electricity generation in the U.S. is from coal, with the balance being made up by 21% from nuclear and 16% from natural gas³⁵ (SI Table S2). It is therefore reasonable to expect that production of electricity in China directly generates more GHGs per kWh than that in U.S. Furthermore, energy conversion efficiency of Chinese power plants is generally considered lower than that of the U.S.

The energy input structure also helps explain the second. The energy used in the Chinese iron and steel industry was, on average, more dependent upon coal-fired electricity than in the U.S. The conspicuously large difference in Table 1, however, had more to do with the technology and value-added of this industry in each economy: the iron and steel industry of China produced fewer high value-added products than its counterpart in the U.S. Specifically, plate and strip products, considered high value-added, accounted for only 35% of the total output of the Chinese iron and steel industry, but for 80% in the U.S. case (SI Table S3). Accordingly, one dollar of output from the iron and steel industry in China has higher contents of material and energy inputs, generating more GHG emissions throughout the life-cycle.

Table 1. Embodied Intensity of Global Warming (CO₂ equiv/dollar)^a

Chinese sectoral database 2002		CEDA 1998	
sector	intensity	commodity	intensity
cement, lime, and plaster products	24	lime	22
electricity and steam production and supply	23	electric services (utilities)	10
iron-smelting	17	cement, hydraulic	9.4
environmental resources and public infrastructure	15	sanitary services, steam supply, and irrigation systems	8.5
water production and supply	12	chemical and fertilizer minerals	8.4
coal mining and processing	12	miscellaneous livestock	5.8
chemical fertilizers	12	meat animals	5.6
steel-smelting	11	food grains	5.4
other nonmetallic mineral products	8.6	tobacco	5.0
coking	8.1	coal	4.9
alloy iron smelting	7.9	fruits	4.7
steel-processing	7.9	tree nuts	4.5
raw chemical materials	7.8	meat packing plants	4.4
gas production and supply	7.2	feed grains	4.2
crop cultivation	6.8	blast furnaces and steel mills	4.2
fireproof products	6.5	miscellaneous crops	4.2
glass and glass products	6.4	poultry and eggs	4.1
sugar refining	6.2	oil bearing crops	4.1
pipeline transport	6.1	grass seeds	4.1
livestock and livestock products	6.0	cotton	4.0

^a The price is adjusted to dollar according to the average exchange currency rate in 2002 between Chinese RMB and U.S. Dollar.

3. APPLICATION

Concomitant with a rapid economic development in China are enhancements in the standard of living and a series of societal, institutional, and cultural changes. One of such changes rapidly shaping China today is urbanization: the ratio of urban population to total, or urbanization rate, more than doubled over the last three decades.¹ Urban area in China is a home for roughly one out of every eleven people on earth, and its population, together with its importance as a production and consumption powerhouse, is rapidly growing. Nevertheless, one should not forget that rural areas in China still hold more people than its urban counterpart, and the magnitude of change that has yet to come, in the course of urbanization, is unprecedented.

The data compiled in this study can be used to illuminate the differences and similarities in rural and urban consumption patterns and associated environmental impacts in China. The results can help understand the environmental implications of urbanization in China.

Both consumption volume and composition in rural area differ significantly from those in urban regions due in large part to income disparities between rural and urban areas.³⁶ In this study, the implications of the differences in consumption pattern between the two areas on the induced environmental impact were analyzed.

The analysis was performed by the following equation:

$$m = C(B_1 B_2) \left(I - \begin{pmatrix} A_{11} & A_{12} \\ A_{21} & A_{22} \end{pmatrix} \right)^{-1} \begin{pmatrix} k_1 \\ k_2 \end{pmatrix} \quad (2)$$

Matrix A_{11} indexes the technology matrix from Chinese 2002 IOTs, and A_{12} the sales from intermediate sectors to households. Matrix k_2 denotes household expenditure per aggregate

consumption activity. For instance, expenditure on petroleum is combined with that on motor vehicles and vehicle parts to form an aggregate consumption activity, driving with auto vehicles (SI Tables S5 and S6). Matrix I is the identity matrix. B_1 is environmental interventions matrix for industrial sectors derived from the developed and validated sectoral database above, while B_2 is for household consumption (details on construction of A_{12} and B_2 matrices are in SI). Matrix C contains characterization factors. Matrices A_{21} , A_{22} , and k_1 are zero matrices. Note that the original method utilized by the EIPRO project also covers waste management stage,¹⁰ whereas we omitted it for lack of relevant data on the amount of recycling and its environmental emissions. We assume imports as being produced with identical technologies of domestic industries following the EIPRO study.

4. RESULTS AND DISCUSSIONS

4.1. Environmental Impacts Induced by Rural and Urban Expenditures. To provide a summary view of the results, we categorized different consumption activities based primarily on the classification of COIOP in EIPRO study (see SI Table S9 for detailed explanation). Then we normalized and weighed the impact induced by each consumption category. Normalization factors were derived from the total emissions compiled in the Chinese sectoral database. Weighting factors were derived from the EIPRO study for comparison only. Note that social values that affect weighting factors may be significantly different between Europe and China, as public perception in China may attach more importance to local environmental issues such as acid rain and human health than to global warming. Thus the application of European weighting factors is only for the purpose of comparison, not for domestic policy-decisions where reflecting such local values is important. The final results are

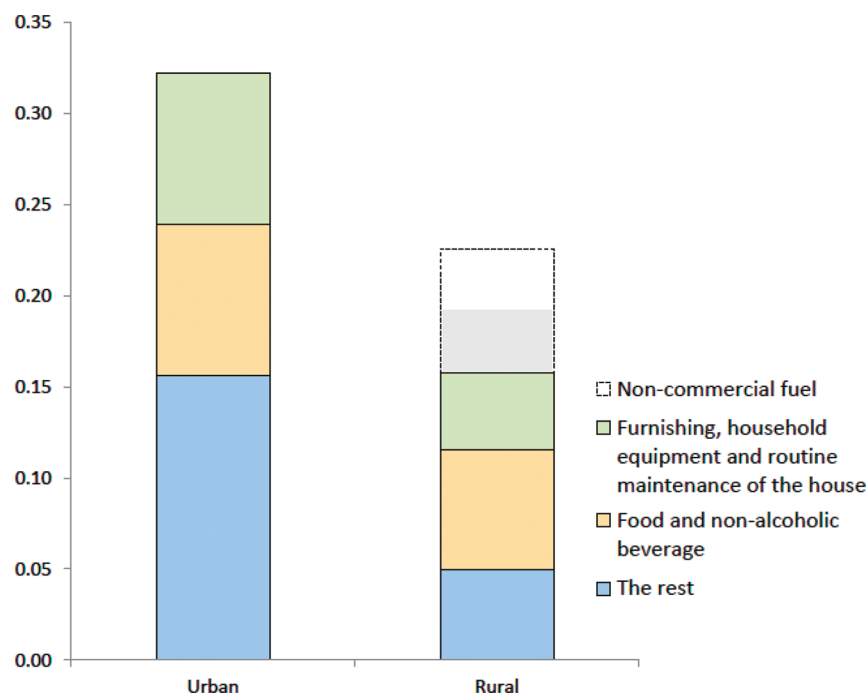


Figure 2. Total weighted environmental impact (unitless) by urban and rural areas in China and major consumption categories. The dotted line indicates the use of noncommercial fuel, mainly biomass, with the shade representing non-CO₂ emissions. The blank area within the dotted line shows CO₂ emission from biomass combustion, which is often considered carbon neutral.

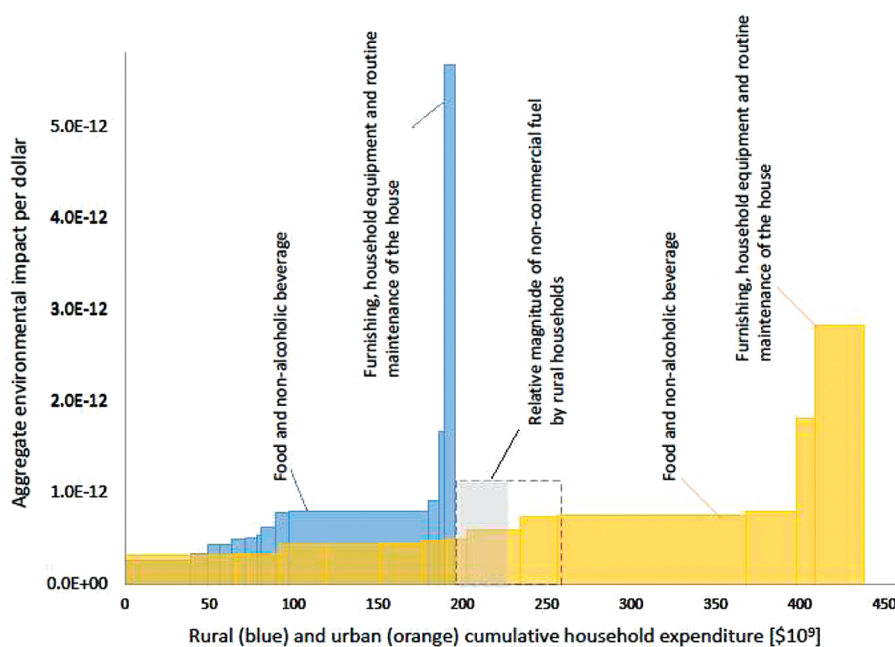


Figure 3. Environmental impact of urban (orange) and rural (blue) expenditures. For no monetary transaction, noncommercial fuel by rural households cannot be represented in this coordinate based on expenditure. The area within the dotted line illustrates the relative magnitude of impact by the use of noncommercial fuel, with the shade representing non-CO₂ emissions. The blank area within the dotted line shows CO₂ emission from biomass combustion, which is often considered carbon neutral.

summarized in Figures 2 and 3 (for detailed, characterized results at individual impact category level, see SI Figure S4).

In 2002, urban and rural households spent \$438.5 billion and \$196.6 billion, respectively. Given that urban and rural areas accounted for 39% and 61%, respectively, of the total population

in 2002, urban area's per capita expenditure was about 3.5 times of rural area's indicating significant urban-rural difference in per capita income in China. The difference is reduced when comparing environmental impacts by the two areas; urban and rural households contributed about 60% and 40%, respectively, to the

total aggregate environmental impact caused by private consumption in China (Figure 2). At the individual impact category level, human and ecotoxicity, acidification and eutrophication impacts induced by urban households were about twice as much as those by the rural households (SI Table S4). Exception, however, arises for the global warming and photochemical oxidation categories. Rural households induced comparable magnitude of global warming impact as did urban households, and even bigger impact of photochemical oxidation (SI Table S4). The exception is due primarily to the combustion of noncommercial fuels by rural households, which generates a significant amount of CO₂, CH₄, CO, SO₂, NO_x, and PM emissions. This aspect will be elaborated later in this section.

For both urban and rural households, food consumption and basic household needs such as cooking and heating were the largest sources of environmental impacts (Figure 2). To identify underlying contributors, we further decomposed the overall impact into the size of expenditure and impact intensity per dollar for each consumption category (Figure 3). For the category, *Food and nonalcoholic*, the overall size of expenditure was the decisive factor, as both urban and rural households spent most of their income on food and nonalcoholic beverage in 2002. In contrast, impact intensity accounts more for the salience of the category, *Furnishing, household equipment and routine maintenance of the house*, which was attributable to direct and indirect combustion of fuels needed to power the operation of various household appliances. These fuels include natural gas, liquefied petroleum gas (LPG), and, particularly, coal, which is still being widely and intensively used in the rural China for household cooking and heating as well as for generating electricity.^{37,38} According to energy statistics,³⁸ coal was the primary commercial fuel among rural households, whereas energy sources for urban households were much more diversified and cleaner in 2002 (dominated by LPG). For detailed information on other consumption categories, see SI Figure S2 and S3.

Another major source of variation in the structure of environmental impacts between urban and rural households in China is the use of noncommercial fuel by rural households. According to energy statistics,³⁸ the major energy source for rural households was noncommercial fuel in forms of firewood, stalks, and biogas, which amounted to 25.8 million ton (Mt) of coal equivalent (around 7566 PJ) or about 80% of total heating value of direct combustion by rural households in 2002.³⁷ As a result, noncommercial fuel was the major source of assorted air emissions for rural households. For instance, combustion of noncommercial fuel generated about 702 Mt of CO₂ as opposed to 96 Mt from commercial fuel used by rural households, making direct CO₂ emissions from fuel combustion by rural households nearly eight times of those attributed to urban households. And it also contributed about one-fourth to the total CO emissions generated by the entire China.⁷ While the CO₂ part of noncommercial fuel could be offset considering CO₂ uptake through photosynthesis (distinguished in Figures 2 and 3), equally important were its other impacts due to significant release of criteria pollutants, particularly CO. In conclusion, combustion of firewood and stalks, coupled with coal use, by rural households for cooking and heating has been one of the major stressors on rural health, observed to be associated with a series of health issues including respiratory illnesses, lung cancer, and weakening of immune system.^{37,39,40}

4.2. Comparison with the Results of EIPRO Study. Above all, it is notable that the economic growth of China depends

substantially on investment and export. This contrasts with European countries where private consumption is the leading contributor to GDP. Also, the European economy is more balanced with almost equal exports and imports. Moreover, the EIPRO study has a more comprehensive coverage of environmental interventions, especially in such categories as human- and eco-toxicity.

Given these caveats, comparison with the EIPRO study focuses on the relative magnitude of environmental impacts of consumption categories between these two economies. The results of the comparison are dominated more by dissimilarities between the two economies (compare Figure 3 with Figure 3 of ref 10). Owing to the difference in economic structure and income-level, consumption patterns in these two economies are quite different. Private expenditures in China are dominated by the consumption of food and the related: one-fifth of the total expenditure for urban households and more than 40% for the rural. Yet the expenditure of EU households is spread relatively evenly over six to seven consumption categories. This result implies that achieving significant environmental impact reductions by reducing the level or composition of consumption in rural China would be more difficult than doing so in Europe under the current economic structure.

Among others, transportation accounts for 17.1% of the overall induced environmental impacts for the EU, whereas it barely constitutes 5% for both the rural and urban areas of China (see SI Tables S7 and S8). This large difference in transportation arises primarily from the distinct gap in vehicle ownership between these two economies. Although China has encountered the highest growth rate of vehicle ownership globally, vehicle ownership per thousand persons, or motorization rate, in China in 2002 was only 16, lagging far behind European countries where the motorization rate was about 500 vehicles per thousand.⁴¹ It is notable that, however, the role of transportation can be much more significant for some major cities in China. For instance, motorization rate of Beijing in 2006 was 180, over 4 times higher than the average urban motorization rate in China for the same year.⁴² Thus it is reasonable to expect transportation in China's most modernized cities such as Beijing and Shanghai has a much higher contribution in the total household environmental impacts.

4.3. Implications on LCA for China. Along with increasing global recognition and popularity, life cycle thinking and LCA are also gaining momentum in China's academic and industrial communities. Besides the increasing number of LCA case studies,^{43–46} generic LCA databases are currently under construction, as well as China's own LCA software.^{47,48} The sectoral database that has been developed can be used as background data for LCA studies through, for example, the hybrid approach,^{49–52} and is also helpful for China's ongoing LCI database projects⁴⁸ by providing a background, reference database. The whole process of development, from identification of key substances to validation using established database, also lends some experience for countries that have not built such sectoral database yet.

4.4. Limitations, Uncertainties and Future Work on the Database. The current study should be considered a first step in constructing a comprehensive environmental IO database for China in the absence of well-structured data infrastructure. Sustaining efforts will be needed to enhance the reliability and timeliness of the database. First, as the current database does not cover the impact category of ozone layer depletion due to data unavailability, further effort should be directed toward

incorporating this impact category once the relevant data become available. Second, uncertainties arise throughout the whole chain of sectoral database development, from the original acquisition of environmental data at their various sources to the process of harmonizing these data to Chinese IOTs. Therefore, quantitative uncertainty characterization for each data cell would be desirable. It is also worth noting that uncertainties reside in the Chinese 2002 IOTs. In the Chinese input-output table, a column named "Others/Error" is used as a balancing term for the discrepancies between input and output, and the magnitude of this term is not negligible. Third, the database needs regular updates with new and better data sources, especially where large uncertainties still exist. In addition, it will be desirable to build and maintain publicly accessible data infrastructure for environmental emissions, especially for toxic substances.

In the absence of Chinese characterization models, normalization references and weighting factors, completion of this study largely resorts to an EU-based LCIA method. It is thus clear that the next steps of LCA development in China include building consistent characterization models, normalization references and weighting factors. Due to China's data constraint, modifying existing characterization models with local geographic or topological data would be a first step.⁵³ The case study on urban and rural consumption ignores the difference in prices between the two areas. Prices of goods and services in rural area are generally lower than those of urban areas, which may lead to an underestimation of environmental impacts attributable to rural consumption.

■ ASSOCIATED CONTENT

S Supporting Information. Additional material as described in the text. This information is available free of charge via the Internet at <http://pubs.acs.org>.

■ AUTHOR INFORMATION

Corresponding Author

*E-mail: suh@bren.ucsb.edu.

■ ACKNOWLEDGMENT

We are grateful to the four anonymous reviewers for their constructive comments, which helped improve the quality of this paper. We also thank Eric Fournier and Joe Bergesen for their helpful comments.

■ REFERENCES

- (1) National Bureau of Statistics of China. *China Statistics Yearbook*, 2006; China Statistics Press: Beijing, China, 2007.
- (2) Lin, J.; Cai, F.; Li, Z. *the China Miracle: Development Strategy and Economic Reform*; Chinese University Press: Hong Kong, 2003.
- (3) Smil, V. *China's Environmental Crisis: An Inquiry into the Limits of National Development*; ME Sharpe Inc: New York, 1993.
- (4) Liu, J.; Diamond, J. China's environment in a globalizing world. *Nature* **2005**, 435 (7046), 1179–1186.
- (5) Managi, S.; Kaneko, S. Economic growth and the environment in China: an empirical analysis of productivity. *Int. J. Global Environ. Issues* **2006**, 6 (1), 89–133.
- (6) Fu, B.; Zhuang, X.; Jiang, G.; Shi, J.; Lü, Y. Environmental problems and challenges in China. *Environ. Sci. Technol.* **2007**, 41, 7597–7602.
- (7) Streets, D.; Zhang, Q.; Wang, L.; He, K.; Hao, J.; Wu, Y.; Tang, Y.; Carmichael, G. Revisiting China's CO emissions after the Transport and Chemical Evolution over the Pacific (TRACE-P) mission: Synthesis of inventories, atmospheric modeling, and observations. *J. Geophysical Res.* **2006**, 111 (D14), D14306.
- (8) Peters, G.; Weber, C.; Guan, D.; Hubacek, K. China's growing CO₂ emissions: A race between increasing consumption and efficiency gains. *Environ. Sci. Technol.* **2007**, 41 (17), 5939–5944.
- (9) Suh, S. Developing a sectoral environmental database for input–output analysis: The comprehensive environmental data archive of the US. *Econ. Syst. Res.* **2005**, 17 (4), 449–469.
- (10) Huppes, G.; Koning, A.; Suh, S.; Heijungs, R.; Oers, L.; Nielsen, P.; Guinée, J. Environmental impacts of consumption in the European Union: High-resolution input–output tables with detailed environmental extensions. *J. Ind. Ecol.* **2006**, 10 (3), 129–146.
- (11) U.S. Environmental Protection Agency. *Sustainable Materials Management: The Road Ahead*; EPA: Washington, DC, 2009.
- (12) Duchin, F.; Lange, G. *the Future of the Environment: Ecological Economics and Technological Change*; Oxford University Press: New York, 1994.
- (13) Hertwich, E.; van der Voet, E.; Suh, S.; Tukker, A.; Huijbregts M.; Kazmierczyk, P.; Lenzen, M.; McNeely, J.; origuchi, Y. *Assessing the Environmental Impacts of Consumption and Production: Priority Products and Materials*, A Report of the Working Group on the Environmental Impacts of Products and Materials to the International Panel for Sustainable Resource Management; United Nation Environmental Programme: Nairobi, Kenya, 2010.
- (14) National Bureau of Statistics of China. *Chinese Environmental Statistics Yearbook*, 2003; China Statistics Press: Beijing, China, 2003.
- (15) Wu, Y.; Wang, S.; Streets, D.; Hao, J.; Chan, M.; Jiang, J. Trends in anthropogenic mercury emissions in China from 1995 to 2003. *Environ. Sci. Technol.* **2006**, 40 (17), 5312–5318.
- (16) Peters, G. P.; Weber, C. L.; Liu, J. *Construction of Chinese Energy and Emissions Inventory*; Industrial Ecology Programme Reports and Working Papers; Norwegian University of Science and Technology: Trondheim, Norway, 2006; http://www.ntnu.no/eksternweb/multimedia/archive/00023/rapport4_06web_23578a.pdf
- (17) Tian, H.; Qu, Y. Inventory of atmospheric arsenic emissions from coal combustion in China, 2005. *Environ. Sci.* **2009**, 30 (4), 956–962 [In Chinese with English abstract].
- (18) Huijbregts, M.; Breedveld, L.; Huppes, G.; De Koning, A.; Van Oers, L.; Suh, S. Normalisation figures for environmental life-cycle assessment: The Netherlands (1997/1998), Western Europe (1995) and the world (1990 and 1995). *J. Cleaner Prod.* **2001**, 11 (7), 737–748.
- (19) Kim, J.; Yang, Y.; Bae, J.; Suh, S. The importance of normalization references in interpreting LCA results. *Environ. Sci. Technol.* **2010**, submitted.
- (20) Guinée, J. B.; Gorree, M.; Heijungs, R.; Huppes, G.; Kleijn, R.; de Koning, A.; van Oers, L.; Wegener Sleeswijk, A.; Suh, S.; Udo de Haes, H. A.; de Bruijn, H.; van Duin, R.; Huijbregts, M. A. J. *Handbook on Life Cycle Assessment: Operational Guide to the ISO Standards*; Kluwer Academic Publisher: Dordrecht, the Netherlands, 2002.
- (21) International Organization for Standardization. *ISO14042, Environmental Management-Life Cycle Assessment-Life Cycle Impact Assessment*; ISO: Geneva, Switzerland, 1999.
- (22) National Bureau of Statistics of China. *Chinese 2002 Input–Output Tables*; China Statistics Press: Beijing, 2006.
- (23) Suh, S.; Huppes, G. Methods for life cycle inventory of a product. *J. Cleaner Prod.* **2005**, 13 (7), 687–697.
- (24) Weber, C.; Peters, G.; Guan, D.; Hubacek, K. The contribution of Chinese exports to climate change. *Energy Policy* **2008**, 36 (9), 3572–3577.
- (25) Guan, D.; Hubacek, K.; Weber, C.; Peters, G.; Reiner, D. The drivers of Chinese CO₂ emissions from 1980 to 2030. *Global Environ. Change* **2008**, 18 (4), 626–634.
- (26) Sinton, J. Accuracy and reliability of China's energy statistics. *China Econ. Rev.* **2001**, 12 (4), 373–383.

- (27) Akimoto, H.; Ohara, T.; Kurokawa, J.; Horii, N. Verification of energy consumption in China during 1996–2003 by using satellite observational data. *Atmos. Environ.* **2006**, *40* (40), 7663–7667.
- (28) Fisher-Vanden, K.; Jefferson, G.; Liu, H.; Tao, Q. What is driving China's decline in energy intensity? *Resour. Energy Econ.* **2004**, *26* (1), 77–97.
- (29) Wu, L.; Kaneko, S.; Matsuoka, S. Driving forces behind the stagnancy of China's energy-related CO₂ emissions from 1996 to 1999: the relative importance of structural change, intensity change and scale change. *Energy Policy* **2005**, *33* (3), 319–335.
- (30) International Institute for Applied System Analysis, *Greenhouse Gas and Air Pollution and Synergies (GAINS)*; IIASA: Laxenburg, Austria, 2009.
- (31) Luo, K.; Xu, L.; Li, R.; Xiang, L. Fluorine emission from combustion of steam coal of north China plate and northwest China. *Chin. Sci. Bull.* **2002**, *47* (16), 1346–1350.
- (32) Weinstein, L.; Davison, A. *Fluorides in the Environment: Effects on Plants and Animals*; CAB International: Oxford, U.K., 2004.
- (33) Chen, M.; Chen, J.; Sun, F. Agricultural phosphorus flow and its environmental impacts in China. *Sci. Total Environ.* **2008**, *405* (1–3), 140–152.
- (34) Leontief, W. Environmental repercussions and the economic structure: an input-output approach. *Rev. Econ. Stat.* **1970**, *52* (3), 262–271.
- (35) State Electricity Regulatory Commission, *Comparison of the Electric Power Industry in China and U.S.*; SERC: Beijing, 2009.
- (36) Sicular, T.; Ximing, Y.; Gustafsson, B.; Shi, L. The urban–rural income gap and inequality in China. *Rev. Income Wealth* **2007**, *53* (1), 93–126.
- (37) Zhang, J.; Smith, K. Household air pollution from coal and biomass fuels in China: Measurements, health impacts, and interventions. *Environ. Health Perspect.* **2007**, *115* (6), 848.
- (38) National Bureau of Statistics of China. *China Energy Statistic Yearbook 2005*; China Statistics Press: Beijing, China, 2005.
- (39) Liu, S. M.; Zhou, Y. M.; Wang, X. P.; Wang, D. L.; Liu, J. C.; Zheng, J. P.; Zhong, N. S.; Ran, P. X. Biomass fuels are the probable risk factor for chronic obstructive pulmonary disease in rural South China. *Thorax* **2007**, *62* (10), 889–897.
- (40) Kleinerman, R. A.; Wang, Z.; Wang, L.; et al. Lung cancer and indoor exposure to coal and biomass in rural China. *J. Occup. Environ. Med.* **2002**, *44* (4), 338–344.
- (41) Dargay, J.; Gately, D.; Sommer, M. Vehicle ownership and income growth, Worldwide: 1960–2030. *Energy J.* **2007**, *28* (4), 143.
- (42) Arvesen, A.; Liu, J.; Hertwich, E. G. Energy cost of living and associated pollution for Beijing residents. *J. Ind. Ecol.* **2010**, *14* (6), 890–901.
- (43) Duan, H.; Eugster, M.; Hischer, R.; Streicher-Porte, M.; Li, J. Life cycle assessment study of a Chinese desktop personal computer. *Sci. Total Environ.* **2009**, *407* (5), 1755–1764.
- (44) Dones, R.; Zhou, X.; Tian, C. Life Cycle Assessment (LCA) of Chinese energy chains for Shandong electricity scenarios. *Int. J. Global Energy Issues* **2004**, *22* (2), 199–224.
- (45) Di, X.; Nie, Z.; Yuan, B.; Zuo, T. Life cycle inventory for electricity generation in China. *Int. J. LCA* **2007**, *12* (4), 217–224.
- (46) Zhang, L.; Huang, Z. Life cycle study of coal-based dimethyl ether as vehicle fuel for urban bus in China. *Energy* **2007**, *32* (10), 1896–1904.
- (47) Gong, X.; Nie, Z.; Wang, Z.; Zuo, T. Research and development of Chinese LCA database and LCA software. *Rare Metals* **2006**, *25* (6), 101–104.
- (48) Wang, H. T. Purpose for accessing sectoral database. Personal communication, 2010.
- (49) Suh, S. Comprehensive Environmental Data Archive (CEDA). In *The Sustainability Practitioner's Guide to Input-Output Analysis*; Murray, J., Wood, R., Eds.; Common Ground Publishing LLC: Champaign, IL, 2010; pp 109–122.
- (50) Suh, S.; Lenzen, M.; Treloar, G.; Hondo, H.; Horvath, A.; Huppes, G.; Joliet, O.; Klann, U.; Krewitt, W.; Moriguchi, Y. System boundary selection in life-cycle inventories using hybrid approaches. *Environ. Sci. Technol.* **2004**, *38* (3), 657–664.
- (51) Strömman, A.; Peters, G.; Hertwich, E. Approaches to correct for double counting in tiered hybrid life cycle inventories. *J. Cleaner Prod.* **2009**, *17* (2), 248–254.
- (52) Reich-Weiser, C.; Ace, F.; Brooks, C.; Suh, S. An iterative hybrid carbon footprint. In *The Sustainability Practitioner's Guide to Input-Output Analysis*; Murray, J., Wood, R., Eds.; Common Ground Publishing LLC: Champaign, IL, 2010.
- (53) Toffoletto, L.; Bulle, C.; Godin, J.; Reid, C.; Deschênes, L. LUCAS—A new LCIA method used for a Canadian-specific context. *Int. J. LCA* **2007**, *12* (2), 93–102.