Life cycle assessment of High Speed Rail in China

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ABSTRACT

China has built the world's largest High Speed Rail (HSR) network. Its environmental impacts have been examined by the means of life cycle assessment (LCA) which describes the whole picture of the HSR system instead of single stages, with a case study for the high-speed railway that links Beijing and Shanghai. The research employs the China-specific life cycle inventory database Chinese Core Life Cycle Database (CLCD). Vehicle operation dominates most impact categories, while vehicle manufacturing/maintenance/disposal and infrastructure construction contribute mostly to mineral consumption (43% and 38%) and organic compounds in water (54% for infrastructure construction). Several scenarios are developed to explore effects of changes in HSR development, utilization, electricity mix, and infrastructure planning and construction. Suggestions are provided for improving the life cycle environmental performance of China's HSR systems.

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Introduction

China has built the world's largest High Speed Rail (HSR) network in less than five years (Lu, 2012). In addition to prompting economic integration and providing safer and faster transportation service, HSR powered by electricity also brings potential environmental benefits compared to conventional trains that are mostly powered by coal or diesel oil. In particular, HSR avoids consuming oil during the operation, leading to reduced air emissions from vehicle and air travels. However, the operation is only one piece of the whole life cycle of an HSR system which includes also infrastructure development, vehicle manufacturing, and electricity generation. All these pieces from the HSR life cycle have different environmental implications. To date, there is no study specifically examining the life cycle environmental impacts of China's HSR system. These questions remain to be answered: What are the life cycle environmental impacts of China's HSR system? How does China's HSR system perform in comparison to other transportation modes? In this paper, we construct a life cycle inventory of environmental impacts of China's HSR system. We also compare our results with other transportation modes.

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Previous LCA studies on rail systems primarily focused on conventional rail systems. For example, it was found that the US conventional rail system had relatively low energy consumption, greenhouse gas (GHG) emissions, and air pollutant emissions (including CO_2, NO_x, PM_{10}, CO, and SO_2) compared to air and road freight systems, but relatively high SO_2 emissions in passenger transportation due to coal-fired power for vehicle operation and removal of sulfur from gasoline and diesel fuels (Chester and Horvath, 2009; Facanha and Horvath, 2007; Facanha and Horvath, 2006). Spielmann and Scholz (2005) conducted a life cycle inventory analysis of freight transport systems and found that rail systems have the lowest environmental impacts compared to barge and lorry systems, except for PM_{10} due to abrasion processes. Using attributional LCA, Spielmann et al. (2005) found that rail system is the best alternative for Switzerland's future regional transport in terms of environmental impacts.

The limited previous studies on life cycle environmental impacts of HSR systems mainly focused on HSR systems in the developed countries or regions including the US, Japan and Europe. Chester (2008) compiled a Life Cycle Inventory (LCI) of energy use, GHG emissions, and criteria air pollutant emissions for diverse passenger transportation modes in the US including the proposed California High Speed Railway (CAHSR). Chester and Horvath (2010) studied the life cycle environmental performance of CAHSR in comparison with alternative transportation modes. The results showed that CAHSR had lower GHG emissions and end-use energy consumption at high capacity utilization, and higher SO_2 emissions at low capacity utilization as it is mainly powered by fossil fuel-based electricity. In addition, CO, NO_x, VOC, and PM_{10} emissions mostly came from infrastructure construction instead of vehicle operation. Chester and Horvath (2012) also found that HSR could achieve considerable life cycle environmental benefits over other current transportation modes with state-of-the-art vehicles, renewable energy, and high ridership. Chang and Kendall (2011) examined GHG emission in the construction of CAHSR infrastructure with several specific infrastructure types depending on terrains. It was found that 80% of the infrastructure emissions were from material production. Tunneling and aerial structures, covering only 15% of the route's length, contributed 60% of the total emissions from the life cycle. Von Rozycki et al. (2003) found that energy use of a German HSR system mainly came from traction, tunnel construction, and rail point heating during winter. Åkerman (2011) examined a proposed Swedish HSR track and found significant GHG emission reduction potential due to transportation modes shifting to HSR, even though new railway construction and maintenance generate GHG emissions. Moreover, Lee et al. (2008), Miyauchi et al. (1999), and Ueda et al. (1999) studied environmental impacts of individual HSR components including infrastructure, vehicle, and materials, respectively. Although HSR offers multiple environmental benefits from replacing oil consumption during the operation, the life cycle environmental impacts of China's HSR may not be desirable, and they may be very different from HSR systems in the developed countries as well as conventional rail systems. First, China's HSR system uses considerable amount of bridges to cross diverse terrains, leading to massive material and energy consumption for infrastructure development. Second, the fuel mix for electricity generation in China is dominated by coal, which undermines the benefits of emission reduction from the operation. Therefore, life cycle assessment is needed to understand the environmental impacts of China's HSR across the entire life cycle. The results can potentially help design future HSR systems for better environmental performance.

In this study, we conduct an LCA for the 1318-km long HSR between Beijing and Shanghai, which is one of the flagship projects of China's HSR development and the world's second longest HSR line (Li, 2007), as a case study to evaluate life cycle environmental impacts of China's HSR system. Our study focuses on life cycle inventory of the Beijing–Shanghai HSR line and multiple environmental impacts from different life cycle stages under various scenarios concerning factors such as infrastructure composition, electricity mix, and capacity utilization rate.

Methods and data

Life cycle stages and data sources

We divide the life cycle of the Beijing–Shanghai HSR system into three stages: (1) vehicle, including vehicle manufacture, maintenance, and disposal; (2) infrastructure, including infrastructure construction; and (3) operation, including vehicle operation. As China just built HSR systems in recent years, we exclude infrastructure operation, maintenance, and disposal due to data unavailability as well as their limited life cycle environmental impacts according to other HSR LCA studies (e.g., Chang and Kendall, 2011; Chester and Horvath, 2010). The system boundary also excludes facilities used and material, energy and equipment transportation due to lack of data and intention to be consistent with other previous studies (e.g., Chester and Horvath, 2010; Yang et al., 2013) for easier comparison. The function of the analyzed system is to transport passengers between Beijing and Shanghai. The functional unit is per seat per kilometer traveled (SKM). In the analysis of different capacity utilization rate scenarios, the functional unit is converted into per passenger per kilometer traveled (PKM).

There are currently no data for China's HSR vehicles (including vehicle manufacturing, maintenance, and disposal stages, but excluding the operation stage for which we obtain China-specific data). We estimate the LCI of China's HSR vehicles by adjusting LCI of Germany's Inter-City Express (ICE) HSR vehicle from the Ecoinvent 3.0 database, since China's HSR vehicle CRH3 uses Siemens's Velaro, or Germany's ICE-3, as a prototype with mostly the same structure and vehicle material (Li and Jin, 2011; Siemens, 2005). In particular, the LCI for China's CRH3 vehicle is estimated by multiplying the LCI of Germany's ICE-3 vehicle with the weight ratio of China's CRH3 vehicle to ICE-3 vehicle (details shown in the SI). Due to data unavailability, we choose not to make arbitrary assumptions on the actual efficiency differences between China and Germany in

various life cycle stages including HSR vehicle manufacturing, maintenance, and disposal. Moreover, we group three stages for vehicles (excluding vehicle operation) together, because they use the same non-China specific data and require the same adjustment methods. Assumptions made for the vehicle stages are the same as (Li and Jin, 2011) based on which we estimated our data for vehicle stages in this study. Comprehensive LCI data collection and modeling on China's HSR vehicles are needed for future research to improve the accuracy of this research.

The sources of data for infrastructure construction and vehicle operation are documented in detail in Wang et al. (in press), including data directly collected from experts in Chinese Ministry of Railways and secondary data from further analysis on material stocks and flows of infrastructure components. We also use Chinese Core Life Cycle Database (CLCD) (IKE, 2012) for other LCI data to ensure we analyze all life cycle inventory data under China's material and energy context. The CLCD database is a publicly available life cycle database designed for China, consisting of over 600 LCI datasets for key materials, chemicals, energy carriers, transport, and waste management representing the combination of various technologies in Chinese market (IKE, 2012; Greenhouse Gas Protocol, 2012). The CLCD is a network model obtained through matrix algorithm accounting for aggregated processes (AP) of product/material life cycle. CLCD has joined the Life Cycle Data Network (LCDN) which provides quality-assured life cycle data (Recchioni et al., 2015). Similar to the Ecoinvent database, CLCD is in the matrix form allowing the consideration of upstream impacts. The Supporting Information provides a dataset of gasoline from the CLCD to illustrate the set of inventory flows. Table 1 shows the inputs, processes and emissions of each stage for the CLCD to conduct analysis.

In the baseline scenario, according to data from Chinese Ministry of Railways, we assume that the average travel distance of vehicles on the Beijing–Shanghai corridor—including vehicles that travel the entire corridor and vehicles that only travel part of the corridor (no matter on this corridor only or cross other adjacent corridors)—is 900 km (L), average capacity utilization rate is 70% (OR), total seats per vehicle with 16 carriages is 1000 (S), and vehicle utilization rate is twice per day (U). We also assume that there are 135 vehicles needed (V) to fulfill the demand based on current HSR travel length per year per capita (400 km/trip) and the proportion of HSR travel on the Beijing–Shanghai HSR line among the total HSR passenger travel (12%). Therefore, there are 13.14 billion SKM for each vehicle during its 20-year lifetime (Y_{vehicle}), assuming 365 days a year. Considering the number of vehicles on this line, there are 8.86 trillion SKM for the whole system during the 100-year lifetime ($Y_{\text{infrastructure}}$) of the infrastructure. We use Eqs. (1) and (2) below to calculate the amount of material and energy consumption allocated to every seat kilometer traveled (q) in each stage in the baseline scenario:

$$q_{\text{vehicle}} = Q_{\text{vehicle}}/(S \times L \times U \times 365 \times Y_{\text{vehicle}}) = Q_{\text{vehicle}}/1.314\text{E} + 10 \text{ SKM} \tag{1}$$

$$q_{\text{infrastructure}} = Q_{\text{infrastructure}}/(S \times L \times U \times 365 \times V \times Y_{\text{infrastructure}}) = Q_{\text{infrastructure}}/8.863\text{E} + 12 \text{ SKM} \tag{2}$$

where Q is the amount of material (kg) or energy (J) usage in each stage during the given lifetime of vehicle or infrastructure. For other infrastructure components, the total SKM will change according to their lifetime, thus leading to a change of material and energy consumption per SKM.

We examine seven categories of life cycle impacts, including Acidification Potential (AP, measured by kg SO_2 equivalent), Chinese Abiotic Depletion Potential (CADP, measured by kg coal resources equivalent, based on CML-ADP$_{\text{world}}$ and adjusted by China's domestic resource reserves and usages) (IKE, 2011), Primary Energy Demand (PED, measured by kg standard coal equivalent), Chemical Oxygen Demand (COD, measured by kg oxygen demanded), Eutrophication Potential (EP, measured by kg PO_4^{3-} equivalent), Global Warming Potential (GWP, measured by kg CO_2 equivalent), and Respiratory Inorganics (RI, measured by kg $PM_{2.5}$ equivalent). These seven categories are chosen because they are evaluated by previous LCA studies using CLCD (e.g., Yang et al., 2013). We use impact assessment methods including CML 2002 (for AP and EP), ISCP 2010 (for CADP),

Table 1
The inputs, processes and emissions of each stage for the CLCD (please refer to Supporting Information for details).

Stage	Sub-components or stage	Inputs, processes and emissions
Vehicle manufacturing, maintenance, and disposal	Manufacturing	Reinforcing steel; steel; aluminum; copper; polyethylene, HDPE; tempering, flat glass; flat glass; alkyd paint; transport, lorry 32t; transport, freight, rail; electricity, medium voltage; heat, light fuel oil; heat, natural gas; heat, waste
	Maintenance	Reinforcing steel; aluminum; copper; alkyd paint; sodium hydroxide; transport, lorry 32t; transport, freight, rail; heat, at cogen 1 MWe lean burn; electricity, medium voltage; tap water; paper; solvents, organic; lubricating oil; treatment, sewage; disposal, emulsion paint; heat, waste; NMVOC, non-methane volatile organic compounds
	Disposal	Transport, lorry 32t; disposal, emulsion paint; disposal, glass; disposal, plastics, mixture
Infrastructure construction	Ballasted track	Concrete C60; stone; steel
	Non-ballasted track	Concrete C60; concrete C30; steel; cement; sand
	Railway track	Steel
	Main project/infrastructure	Cement; concrete C30; stone; sand; steel; copper; lime; concrete C50; concrete C25; concrete C15; concrete CFG
	Others	Electricity; gasoline; diesel fuel; coke; coal; firewood; water
Vehicle operation		National average electric utility network mix and transmission

IPCC 2007 (for GWP), IMPACT 2002+ (for RI), and normalization method with normalization reference of China in 2010 from (IKE, 2012). Moreover, as currently there are no China-specific comprehensive local characterization factors, IKE (2012) uses characterization factors from existing characterization methods as a most feasible and applicable approximation. We use normalization factors from the IKE software (Table 2) which are calculated from statistical data of the National Bureau of Statistics of China (National Bureau of Statistics of China, 2015) and International Institute for Applied Systems Analysis (IIASA, 2012).

Scenarios

This study uses the baseline scenario described in the above section to present the baseline situation of China's HSR. We also develop several other scenarios on infrastructure composition, electricity mix, HSR system development stage, travel length of cross-line vehicles, vehicle utilization, capacity utilization rate, and use of fly ash in concrete for infrastructure construction. These scenarios are designed for two purposes. First, they are used to test the sensitivity of the LCA results to changes in various input factors. Second, results of the sensitivity analysis are used to identify major factors as potential levers to reduce life cycle environmental impacts. Forecasting future development of the HSR system should be based on additional analysis and is not within the scope of this study.

In particular, we develop seven infrastructure composition scenarios (A1–A7) to reflect the usage of bridges, tunnels, and reinforced subgrade in HSR infrastructure development. There are eight electricity mix scenarios (B1–B8) varying the share of coal-fired power, hydropower, nuclear power, and wind power. Two HSR development stage scenarios (C1–C2) are developed to represent low and high level of HSR penetration comparing to the baseline scenario. Scenarios D1 and D2 represent different travel lengths of all cross-line vehicles between Beijing and Shanghai. Scenarios E1 and E2 represent different times that an HSR vehicle is used per day. Scenarios F1 and F2 reflect different capacity utilization rates and use PKM instead of SKM as the functional unit. Finally we develop a G1 scenario to use no fly ash in concrete, comparing to the baseline scenario that uses fly ash. Details of these scenarios are introduced when discussing scenario analysis results later and can also be found in the Supplementary Information.

Results

Baseline scenario

Fig. 1 shows life cycle environmental impacts of the Beijing–Shanghai HSR system by different impact categories. In particular, vehicle operation contributes to 72–91% of the life cycle environmental impacts for AP, PED, EP, GWP, and RI, which is similar to the fact that vehicle operation stage contributes to about 70–90% of the end-use energy consumption, GHG emissions, and SO_2 emissions in a LCA study of California HSR (Chester and Horvath, 2010). This is mainly because the electricity used for HSR vehicle operation largely comes from coal-fired power plants generating considerable amounts of SO_2, NO_x, CO_2, $PM_{2.5}$ emissions. HSR vehicle manufacturing/maintenance/disposal and infrastructure construction contribute to 43% and 38% of CADP, respectively, mainly because of the use of copper that has the largest impact on CADP comparing with other materials such as steel, aluminum, and concrete. Moreover, infrastructure construction dominates life cycle COD (54%) due to the use of concrete, steel, and copper which all contribute significantly to the COD impact.

We further investigate CADP and COD contributions by different materials as shown in Fig. 2. While the use of aluminum and steel in vehicle manufacturing/maintenance/disposal is approximately twice and ten times the amount of copper used, respectively, and the use of steel in infrastructure construction is about 200 times the amount of copper used, copper dominates the CADP impact at the vehicle manufacturing/maintenance/disposal stage (89%) and the infrastructure construction stage (62%), due to much higher CADP impact of one kilogram of copper (11,230 kg coal-r eq.) than those of steel (30.7 kg coal-r eq.) and aluminum (249 kg coal-r eq.). On the other hand, steel contributes to 91% of the COD impact in infrastructure construction, followed by diesel (only 5%).

We use a Chinese 2010 reference system (IKE, 2012) to normalize the seven categories of life cycle environmental impacts as shown in Fig. 3. In particular, CADP contributes to 32% of the normalized life cycle environmental impact, followed by PED (17%), AP (16%), EP (13%), GWP (11%), RI (10%), and COD (2%). Vehicle operation contributes to 63% of the overall life cycle

Table 2
Baseline normalization factors used in this study.

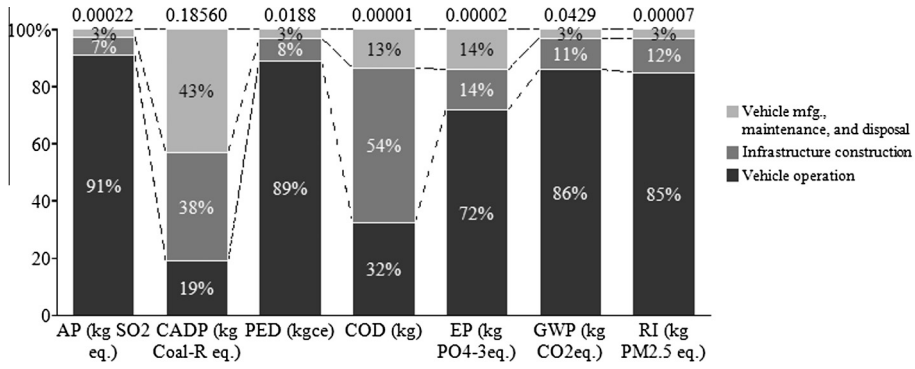| Indicators	Baseline normalization factors	Units
Nonrenewable resource demand	1.55E+13	kg coal-R eq.
Primary energy demand	1.06E+13	kg ce
Global warming	3.77E+09	kg CO_2 eq.
Acidification	3.64E+10	kg SO_2 eq.
Eutrophication	2.96E+12	kg PO_4^{3-} eq.
Respiratory inorganics	4.34E+09	kg $PM_{2.5}$ eq.

Fig. 1. Life cycle environmental impacts (on a functional unit basis) of the Beijing–Shanghai HSR system by various impact categories as introduced in the Methodology section.
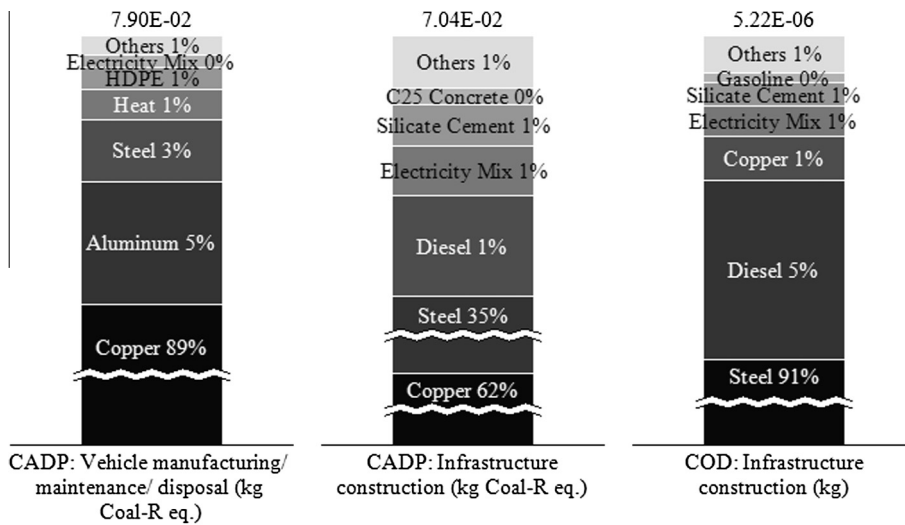

Fig. 2. Impacts by materials (on a functional unit basis): from left to right, CADP in vehicle manufacturing/maintenance/disposal, CADP in infrastructure construction, and COD in infrastructure construction.
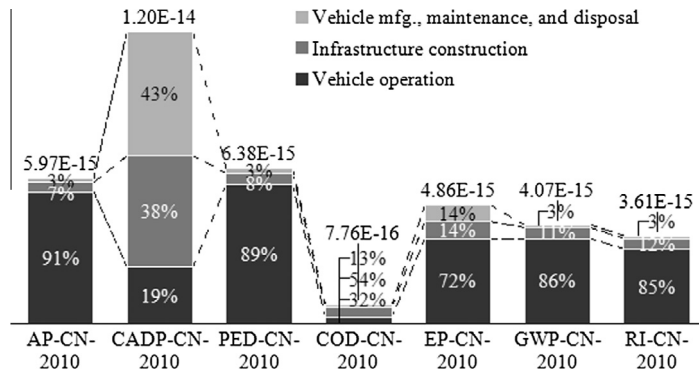

Fig. 3. Normalized life cycle environmental impact (on a functional unit basis) of the Shanghai–Beijing HSR system by different impact categories.
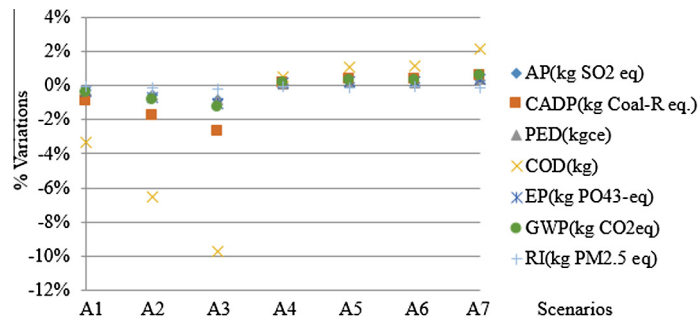
Fig. 4. Changes of life cycle environmental impacts (on a functional unit basis) under different infrastructure composition scenarios compared to the baseline.

environmental impact due to its dominance in five of the seven impact categories, while vehicle manufacturing/mainte nance/disposal and infrastructure construction take 20% and 17%, respectively, mainly due to their dominance in CADP.

Scenario analysis

Fig. 4 shows changes of life cycle environmental impacts in each category under different scenarios of infrastructure com-position (details shown in the SI). In scenarios A1, A2, and A3, we reduce the proportion of bridge to 70%, 60%, and 50% from 80.4% in the baseline, respectively. As a result, COD decreases rapidly and CADP drops as well but at a lower rate, indicating that life cycle COD and CADP impacts are sensitive to the number of bridges in infrastructure development. In scenarios A4 and A5, we increase the proportion of tunnels to 5% and 10% from 1.2% in the baseline, respectively, leading to a slight increase of COD. Scenarios A6 and A7 increase the proportion of reinforced subgrade to 40% and 60% from 18% in the base-line, respectively, leading to a slight increase of COD as well. In these scenarios, we compensate the change of the proportion of bridge, tunnel, and reinforced subgrade by accordingly decreasing or increasing the length of ordinary subgrade which has relatively less impact compared to other types of infrastructures. Overall, COD, CADP, and GWP are most sensitive to the por-tion of bridges on the HSR line, while COD is also sensitive to the portion of tunnels and reinforced subgrade. Changes of infrastructure composition have relatively less impact on other impact categories.

Fig. 5 shows the changes of life cycle environmental impacts under different electricity mix scenarios (details shown in the SI). We model these different electricity mix scenarios by adjusting the proportion of different power generation tech-nologies. We use national electricity mix due to data unavailability of regional electricity mix. In scenarios B1 and B2, we increase the proportion of hydropower from 15% in the baseline to 20% and 25%, respectively, resulting in significant decreases of GWP, RI, AP, PED, and EP, given the fact that hydro power generates much less GHG, NO_x, SO_x, and $PM_{2.5}$ emis-sions and consumes much less primary energy. In scenarios B3, B4, and B5, we increase the proportion of nuclear power from 2% in the baseline to 7%, 12%, and 17%, respectively, leading to considerable decreases of GWP, RI, AP, EP, and PED, as nuclear power has similar advantages as hydropower and uses less primary energy in its life cycle. In scenarios B6, B7 and B8, we increase the proportion of wind power from 2% in the baseline to 7%, 12%, and 17%, respectively, resulting in significant decreases of GWP, RI, AP, PED, and EP similar to scenarios B1 and B2. However, wind power can lead to a significant increase of CADP due to the use of metals and land in wind power development. Again, in these scenarios we make up the changes of the share of hydropower, nuclear power, and wind power by accordingly changing the share of coal-fired electricity. Due to lack of solar power data in CLCD, we do not examine the effect of solar power penetration on HSR's life cycle environmental impacts, which represents an interesting research avenue in future.

Fig. 6 shows the change of life cycle environmental impacts under different scenarios of HSR penetration, travel distance of cross-line vehicles, and vehicle utilization (details shown in the SI). Scenario C1 represents low level of HSR penetration, with relatively low number of HSR trips and travel length per year per capita and high share of travels using the Beijing–Shanghai HSR line among the total HSR passenger travels. As a result, the number of vehicles required and the total SKM for the Beijing–Shanghai HSR line are less than those in the baseline scenario, leading to approximately 8–15% increases of PED, AP, EP, GWP, and RI, and even higher COD (62%) and CADP (43%). On the other hand, scenario C2 represents high level of HSR penetration, with relatively high number of HSR trips and travel length per year per capita and low share of travels using the Beijing–Shanghai HSR line among the total HSR passenger travels. This results in the increases of the num-ber of vehicles required and the total SKM, leading to less than 5% decreases of PED, AP, EP, GWP, and RI, and 15% decrease for COD and 22% decrease for CADP.

In scenarios D1 and E1, shorter cross-line travel distance and lower level of vehicle utilization imply less SKM per vehicle, leading to increasing EP, COD, and CADP which are dominated by the vehicle manufacturing/maintenance/disposal stage. In contrast, scenarios D2 and E2 use longer cross-line travel distance and higher level of vehicle utilization, resulting in increased SKM per vehicle. This leads to deceasing EP, COD, and CADP for the same reason.
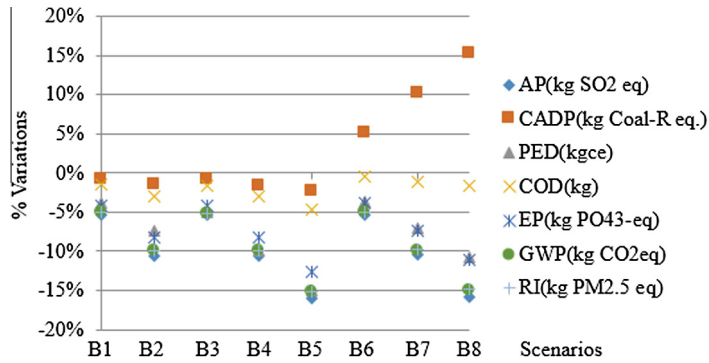
Fig. 5. Changes of life cycle environmental impacts (on a functional unit basis) under different electricity mix scenarios compared to the baseline.
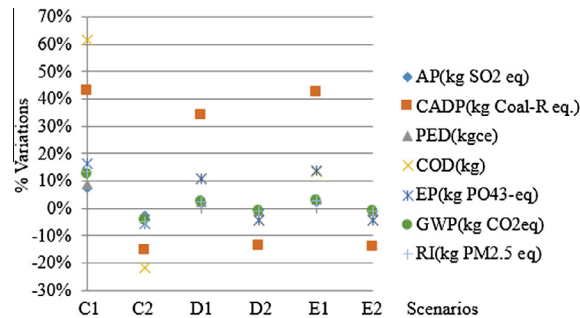

Fig. 6. Changes of life cycle environmental impacts (on a functional unit basis) under different scenarios of HSR penetration, cross-line travel distance, and vehicle utilization compared to the baseline.
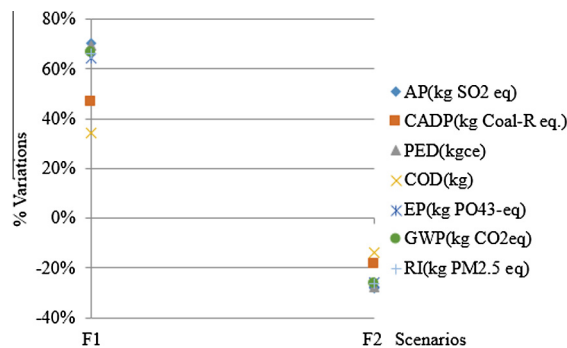

Fig. 7. Changes of life cycle environmental impacts (on a functional unit basis) under different capacity utilization scenarios compared to the baseline.

Fig. 7 shows the changes of life cycle environmental impacts under different capacity utilization scenarios. Scenario F1 reduces the capacity utilization rate to 40% from 70% in the baseline, leading to increasing number of HSR vehicles needed to fulfill the same total PKM and increasing environmental impacts by 34–70% across various categories. In particular, PED, GWP, AP, EP, and RI that are dominated by the vehicle operation phase increase by more than 60%. In scenario F2 with 100% capacity utilization rate, the number of HSR vehicles required decreases, resulting in decreasing environmental impacts by 19–28%.

Fig. 8 shows the changes of life cycle environmental impacts under scenario G1 eliminating fly ash use in concrete (details shown in the SI). The elimination of fly ash leads to over 5% increases of GWP, PED, and AP in the infrastructure construction stage, given the role of cement in infrastructure construction. For the entire life cycle of the Beijing–Shanghai HSR line, the elimination of fly ash can increase GWP by approximately 1%.

Due to limited information on life cycle environmental impacts of other transportation modes in China, it is difficult to compare HSR with other transportation modes on a functionally equivalent basis. Because the same impact assessment method is used for RI as SimaPro, we used SimaPro to conduct an LCA for Germany's ICE, Germany's conventional rail
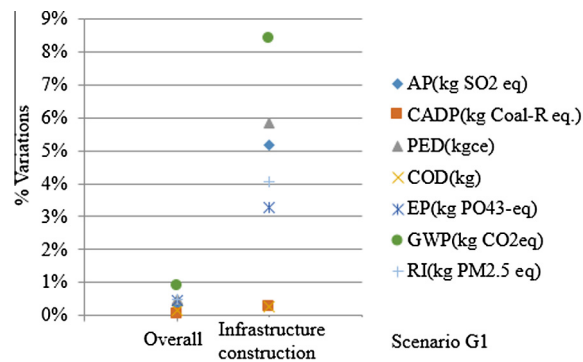
Fig. 8. Changes of life cycle environmental impacts (on a functional unit basis) in scenario G1 (for both all stages overall and infrastructure construction stage only) compared to the baseline.
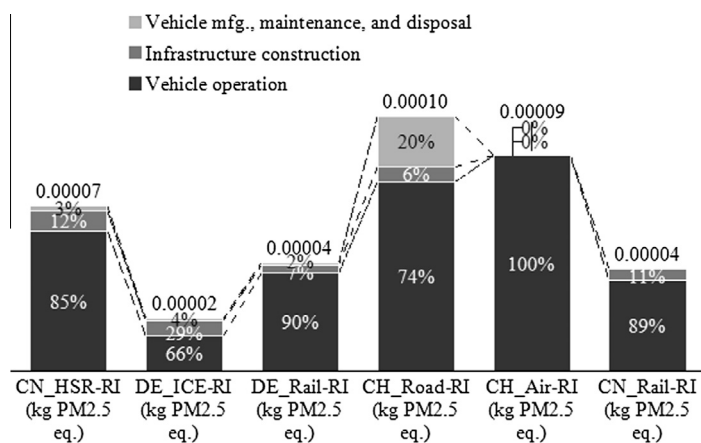

Fig. 9. RI impact (on a functional unit basis) of China's HSR systems, Germany's ICE and rail systems, Switzerland's road and air systems, and China's rail systems (for freight transportation).

systems, and Switzerland's road and air transport systems to compare with China's HSR systems. Other impact categories are difficult to compare, since the two software tools use different impact assessment methods. We exclude the infrastructure maintenance/operation/disposal stage to keep the same scope with this research. Meanwhile, we use a recent LCA study of China's railway freight transportation using CLCD (Yang et al., 2013) as a reference, although it does not consider the vehicle manufacturing/maintenance/disposal stage. The functional unit for China's railway freight transportation is ton per kilometer traveled, making direct comparison infeasible. For other systems, the functional unit is passenger kilometer traveled assuming 100% capacity utilization rate which can be easily converted to seat kilometer traveled.

Fig. 9 shows that the RI impact of China's HSR systems (CN HSR-RI) is higher than Germany's ICE (DE HSR-RI) and rail systems (DE Rail-RI), but is lower than Switzerland's road (CH Road-RI) and air transportation systems (CH Air-RI). As electricity use in vehicle operation dominates the RI impact and vehicle operation contributes significantly to most impact categories, it is very likely that the comparison of other impact categories is similar to RI. Thus, China's HSR systems may have larger environmental impacts than conventional rail systems, but fewer impacts than road and air transportation systems.

Discussion and conclusions

This research assesses life cycle environmental impacts of China's HSR transportation between Beijing and Shanghai. Results show that the vehicle operation stage dominates impacts of AP, PED, EP, GWP, and RI. It is mainly because that the electricity mix in China heavily depends on coal-fired power generation, leading to significant SO_2, NO_x, CO_2, and $PM_{2.5}$ emissions. Infrastructure construction contributes mostly to COD and CADP, due to large amounts of steel and copper used. Vehicle manufacturing/maintenance/disposal generates significant effects of CADP, primarily due to copper use in vehicle manufacturing.

The scenario analysis shows that bridges contribute significantly to COD, because these aerial structures are highly steel-intensive. Electricity mix with less coal-fired power can reduce multiple environmental impacts in vehicle operation.

Hydropower, nuclear power, and wind power all have similar benefits in reducing GWP, RI, and AP, but wind power can actually increase CADP due to large use of metals and land. Higher level HSR penetration can curb life cycle environmental impacts on a per SKM basis. Longer travel distance of cross-line trains and more frequent vehicle utilization are preferable as well. Moreover, higher seat utilization rate saves electricity consumption and requires fewer vehicles, thus it can diminish all the impacts, notably with respect to PED, GWP, AP, RI, and EP. Using fly ash to replace cement in concrete can abate the consequences of GWP, PED, and AP.

To improve the life cycle environmental performance of HSR systems, several recommendations are proposed. First, the current electricity mix in China should gradually shift toward cleaner and renewable sources including nuclear, wind, hydropower, and solar energy. This would likely create the biggest environmental benefits in consideration of the dominant role the vehicle operation phase plays in the life cycle of HSR. Second, better operation strategies can potentially reduce life cycle environmental impacts of HSR systems; options include optimizing scheduling and marshalling, boosting capacity utilization rate, increasing travel length of cross-line vehicles, and improving HSR penetration in overall transportation. Third, if geographic conditions allow, avoid bridges, tunnels, and reinforced subgrade as far as safety and speed are not sacrificed. Last but not least, materials of smaller environmental footprints are recommended for rail, vehicles, and infrastructure, such as lightweight metals and composites in cars and fly ash in concrete.

As the first LCA study investigating China's HSR system, the research was constrained by data availability. For example, data for vehicle manufacturing, maintenance, and disposal of China's HSR vehicles are unavailable, and we have to estimate those data by adjusting the LCI of Germany's ICE HSR. Meanwhile, CLCD has not been as mature and comprehensive as the Ecoinvent database due to much shorter development history, which may render the analysis using the two databases not as fully comprehensive as other analyses using Ecoinvent database alone. Although CLCD aims to provide comprehensive China-specific inventory flows for Chinese life cycle studies, current Chinese data statistics do not support the construction of comprehensive China-specific inventory flows. Nonetheless, with growingly useful and comprehensive information incorporated into CLCD, more accurate and insightful assessment, including the assessment of other transportation modes, will be carried out in China in the future. This will enable comparisons between multiple modes, and hence significantly strengthen the usefulness of the relevant research.

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Appendix A. Supplementary material

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.trd.2015.10.005.

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