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China's ecological footprint via biomass import and consumption is increasing

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As China's economic and population demands grow, the need for biomass imports rises, presenting challenges for sustainable development and global ecosystem protection. Here we examine China's increasing influence on global ecosystems through its biomass product consumption and trade using the Human Appropriation of Net Primary Productivity framework, analyzing data from 2004 to 2017 with projections until 2050. In 2017, China was the world's largest biomass consumer, consuming approximately 1.75 petagrams of carbon per year, projected to rise by 317.6% by 2050, highlighting China's major role in global ecosystem stress. Our findings also show an increase in biomass imports from lower-middle-income and low-income countries between 2004 and 2017, and these imports are expected to continue increasing by 402.9% in 2050. The analysis reveals that domestic product demand and changes in foreign production efficiency are key drivers of this trend, suggesting the need for China to shift towards more efficient trade practices and support cleaner production methods internationally.

As the largest developing nation in the world, China stands at the epicenter of global environmental challenges. Characterized by rapid economic growth, with its GDP growing from approximately \$1.21 trillion in 2000 to over \$17.96 trillion in 2022 (making it the second-largest economy globally), and an enormous population, which surpassed 1.4 billion in 2022 (the most populous country in the world), the scale of China's demand on natural resources is unparalleled. The country's rising middle class, coupled with urbanization and lifestyle changes, is driving an exponential increase in demand for goods and services. This heightened demand is expected to become a key determinant of future global environmental governance. As global interconnectedness has accelerated, international trade has become a pivotal bridge that links China's burgeoning demands with worldwide production activities. This global economic integration means that China's consumption patterns can exert far-reaching effects on ecosystems far removed from its geographical borders^{1–4}. A striking example is the environmental impact of China's soybean imports: 43% of deforestation emissions caused by soybean cultivation in Brazil in 2017 can be attributed to China's soybean import⁵. Another study shows that China's imports of ruminant products and livestock feed were responsible for transferring 12 Tg CO₂-equivalent greenhouse gas emissions and 42.8 Gg of NH₃ emissions to the exporting nations⁶. Understanding the environmental impact of

China's consumption, therefore, becomes crucial to a holistic approach to global ecological conservation.

Previous research regarding the environmental implications of China's demands has provided valuable insights^{7–9}. For instance, studies estimate indicating a substantial share of global emissions is attributable to China's economic activities^{10–12}. In addition, research has shown that PM_{2.5} pollution produced in China in 2007 is linked to more than 64,800 premature deaths in regions other than China¹³. A study found that consumption in China is responsible for 20% of global nitrogen pollution¹⁴. Feed demand in China causes 65% of groundwater depletion for soybeans in the USA¹⁵. China appropriates large amounts of forestland from Russia (64 Mha or 15% of Russian industrial forests), Africa (20 Mha or 10% of African industrial forests) and Southeast Asia (8 Mha or 8% of Southeast Asian industrial forests) for its consumption needs of timber, particularly for construction of infrastructure¹⁶. Nevertheless, despite these substantial contributions to our understanding, notable gaps remain. The impact of China's demands on global ecosystems has not been extensively explored. This leaves substantial room for further research to deepen our understanding of these ecological interactions on a global scale.

Human-induced alterations of terrestrial ecosystems are primarily driven by land use and its associated impact on net primary production (NPP).

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One notable metric used to evaluate this human influence on ecosystems is the human appropriation of net primary production (HANPP). HANPP quantifies the extent to which human activities alter the availability of biomass by measuring the difference between potential natural vegetation NPP (NPP_{pot}, the theoretical amount of biomass production in the absence of human intervention) and the NPP remaining in the ecosystem after human occupation (denoted as NPP_{eco}). It provides a unique perspective on human interference with ecosystems by considering both direct human-induced changes, such as agricultural practices, and indirect ones, like changes in land use and management. Despite its focus primarily on biomass appropriation and reliance on variable land use and NPP data, HANPP remains a critical indicator for understanding and managing the ecological impacts of human activities on terrestrial ecosystems. And it is an essential tool for comprehending impacts on biodiversity and ecosystem functioning. This indicator has garnered considerable attention^{17,18} following a pivotal study by Vitousek et al.¹⁹, who found that HANPP globally accounted for 31% of the total NPP generated on land. Estimates for global HANPP range from 6.9 to 29.5 petagrams of carbon per year (Pg C/yr)²⁰, corresponding to 20–40% of potential vegetation NPP^{21,22}. It provides a unique perspective on human interference with ecosystems, making it an essential tool for comprehending impacts on biodiversity and ecosystem function.

To sum up, previous research on the environmental impact of China's demand has provided valuable insights. However, considerable gaps remain despite these contributions to our understanding. First, the impact of China's demand on global ecosystems, particularly regarding HANPP, has not been widely explored. This is an oversight given China's status as a major global economic and ecological player. Our study focuses on this critical area, examining China's unique influence on global ecosystems. Second, while there is literature on territorial HANPP^{17,18,23} or HANPP flows embodied in agricultural trade (often represented as “eHANPP”)^{20,24,25}, the consideration of indirect HANPP consumption in non-agricultural production processes has been lacking. This is particularly pertinent in the case of China, where such indirect effects are substantial due to its vast and complex economic activities. In addition, the increasing divergence between territorial HANPP and consumption-based HANPP (cHANPP) due to global trade linkages is a crucial aspect that our study addresses. A comprehensive focus on a nation's cHANPP is crucial for identifying demand-side solutions for HANPP governance, rather than focusing solely on territorial HANPP. Third, there is limited understanding of the demand-side drivers of cHANPP. Fourth, to our knowledge, most previous studies have concentrated on the historical patterns of HANPP, with insufficient attention to how HANPP might evolve under future socioeconomic scenarios. This forward-looking analysis is crucial for understanding the potential trajectories of China's HANPP in the context of global ecological sustainability.

To fill these research gaps, our study combines a dynamic global vegetation model with an environmentally extended multi-regional input-output model. This innovative approach allows us to assess the comprehensive impact of China's demand on the global ecosystem. We thoroughly examine China's production-based HANPP, consumption-based HANPP, and embodied HANPP from international trade with 140 other countries/regions. Furthermore, we employ structural decomposition analysis (SDA) to evaluate the contributions of eight drivers to overall changes HANPP changes. Alongside this, leveraging Shared Socioeconomic Pathways (SSPs), we also projected future trends in HANPP and HANPP trade. Details of our methodology and data sources are in the “Methods” section.

Results

Temporal dynamics of HANPP in China

Figure 1 displays the top ten regions with the largest production-based HANPP, and embodied HANPP from exports, imports, and the HANPP trade balance or net trade. We can see that China is one of the most important HANPP-utilizing countries in the world. China has the world's third-highest production-based HANPP (1.18 Pg C/yr) and the highest consumption-based HANPP (1.75 Pg C/yr) in 2017 (Fig. 1a, b). And the higher consumption-based HANPP than production-based HANPP in

China indicates that China's ecological footprint extends far beyond its borders, influencing global ecosystems through its trade activities. The proportion of consumption-based HANPP in China to global HANPP has increased, from 8.4% in 2004 to 11.8% in 2017 (Fig. 1g). In addition, China is also the largest importer and net importer of HANPP in 2017 (Fig. 1d, f). Notably, China's HANPP imports were approximately twice that of the United States, which ranked second, and its net HANPP imports were around three times that of Japan, the second-largest net importer.

We also computed a low-estimate and a high-estimate HANPP and cHANPP datasets based on different parameter settings for robustness (detailed in the “Methods” section). Supplementary Fig. 1 illustrates that the global patterns of HANPP utilization remain largely consistent under both high-estimate and low-estimate. In all three estimates—standard, high, and low—China consistently ranks within the top five globally for production-based HANPP. Moreover, it holds the position of the highest consumer in consumption-based HANPP across all three estimates. In addition, China emerges as the world's largest net importer of HANPP, with its net import volume being approximately three times that of Japan, the second-largest net importer. This pattern demonstrates China's consistent and considerable impact on global HANPP dynamics under varying estimations.

The evolution of production-based and consumption-based HANPP in China between 2004 and 2014 was characterized by two periods: a decline from 2004 to 2007 and a growth period thereafter (Fig. 1g). Compared to consumption-based HANPP, the change in HANPP was relatively flat. While consumption-based HANPP increased rapidly, from 1.2 Pg C/yr in 2004 to 1.7 Pg C/yr in 2017, an increase of 41.7%. Since 2011, consumption-based HANPP has also overtaken production-based HANPP due to the rapid increase in HANPP inflows into China through trade, which exceeded HANPP outflows from China (Fig. 1g). These changes suggest that the ecological pressure on China's ecosystems remains largely stable. However, the impact of China's demand is growing rapidly, indicating that the environmental stress is mainly derived from imports. This growth in consumption-based HANPP underscores China's increasing reliance on global resources to meet its burgeoning demands, and the corresponding implications for the ecosystems of exporting countries.

In both the high and low estimates, China's HANPP exhibits a consistent trend. From 2004 to 2017, the development of China's production-based and consumption-based HANPP underwent two distinct stages (Supplementary Fig. 2). Before 2011, production-based HANPP was predominant over consumption-based HANPP. Post-2011, a reversal occurred with consumption-based HANPP surpassing production-based HANPP, and the divergence between the two has been widening since. Furthermore, the share of China's consumption-based HANPP in the global context has also seen a marked increase. According to our baseline calculations, it escalated from 8.4% in 2004 to 11.8% in 2017 (Fig. 1g). In the high-value assessment, this proportion grew from 9.0% to 13.1%, and in the low-value assessment, it expanded from 6.8% to 9.5% (Supplementary Fig. 2).

HANPP embodied in Chinese imports and export across different agricultural sectors

During our study period, China shifted from a net exporter to a net importer of HANPP marking a notable shift in its role in the global ecological landscape. In 2004, China saw net HANPP outflows of 0.06 Pg C/yr, but by 2017, net inflows reached 0.57 Pg C/yr. This shift indicates a fundamental change in China's interaction with the global ecosystem, increasingly drawing on international resources to meet its domestic demands. The main consumers of Chinese HANPP included the USA, Western Europe, Japan, Korea, and Russia (Supplementary Fig. 3). However, from 2004 to 2017, China's HANPP exports to these countries gradually decreased, with exports to the USA declining by 70.7%.

Conversely, China's HANPP exports to neighboring countries, specifically South Asia, increased by 42.5%. China primarily imports HANPP from Brazil, the USA, Australia, Canada, and Argentina, accounting for 28.0% of its consumption in 2017. Of these, Brazil was the largest supplier with China's consumption increasing by 392.3% from 2004 to 2017.

Regionally, China's HANPP exports to North America, the EU, and East Asia & Pacific decreased by 70.3%, 59.3%, and 55.9%, respectively. On the other hand, China's imports from various regions have seen marked increases from 2004 to 2017. The smallest increase of 76.3% was in imports from the rest of Europe, while the largest increase was a substantial 331.5% from Central and Western Asia. Latin America and the Caribbean region remain China's largest import sources. The total import volume from these regions increased by 245.0%, and their share of total imports rose from 28.6% in 2004 to 38.5% in 2017. This dramatic

growth in trade with developing countries is a reflection of the expanding South-South trade, with China increasingly turning to other developing countries, particularly in Asia, for its imports. However, this shift towards sourcing from developing countries, which typically have higher HANPP intensities, suggests that the intensification of South-South trade could bring additional ecological pressures. As China continues to import more from these regions, the associated ecological impact due to higher HANPP intensities could exacerbate global environmental stress.

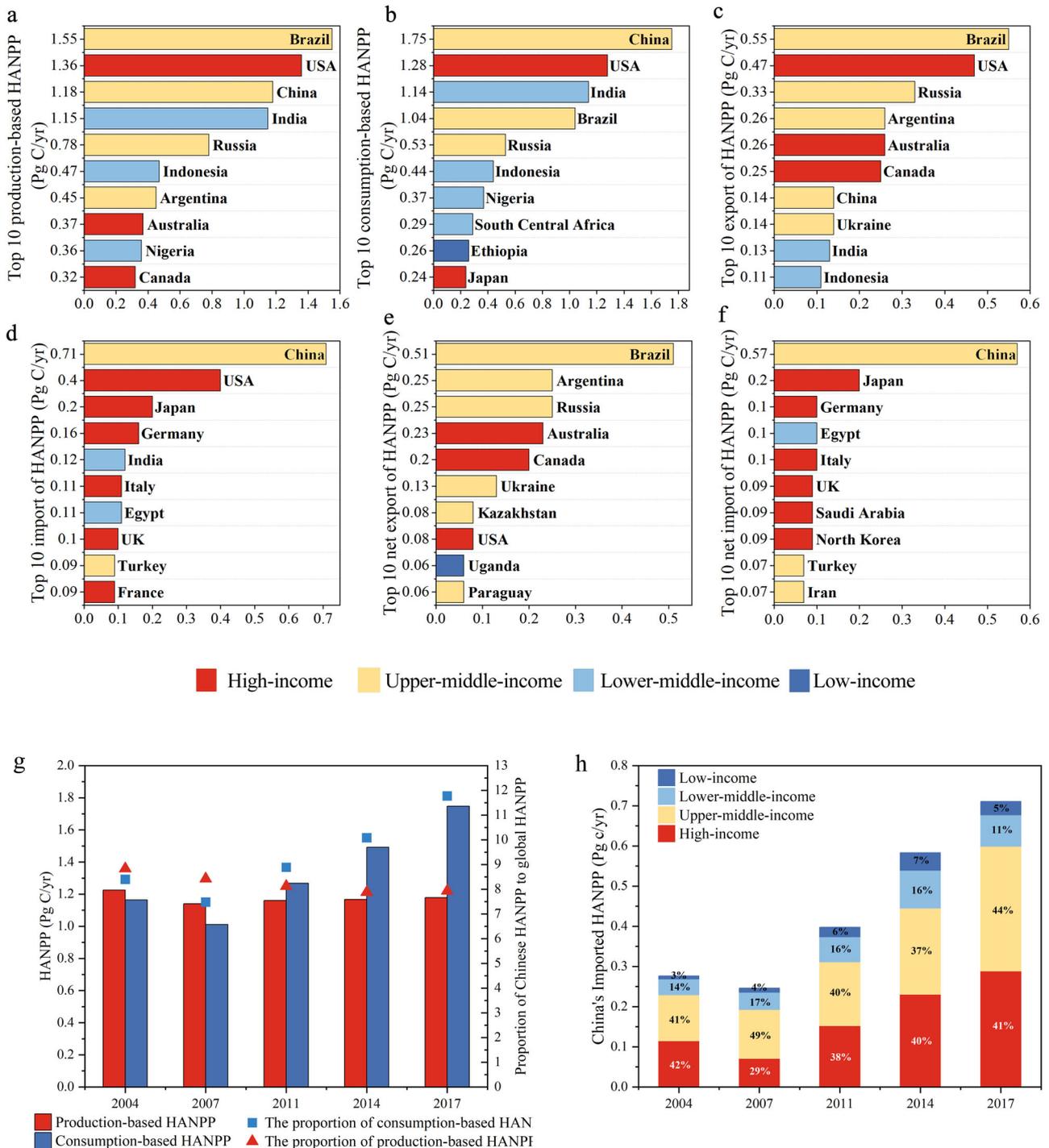


Fig. 1 | Global HANPP distribution and trends in 2017 and China's HANPP dynamics from 2004 to 2017. Top ten regions by production-based HANPP in 2017 (a), consumption-based HANPP in 2017 (b), export of HANPP in 2017 (c), import of

HANPP in 2017 (d), net export of HANPP in 2017 (e), and net import of HANPP in 2017 (f). **g** HANPP trends in China from 2004 to 2017. **h** Proportions of China's Imported HANPP from countries with different income levels from 2004 to 2017.

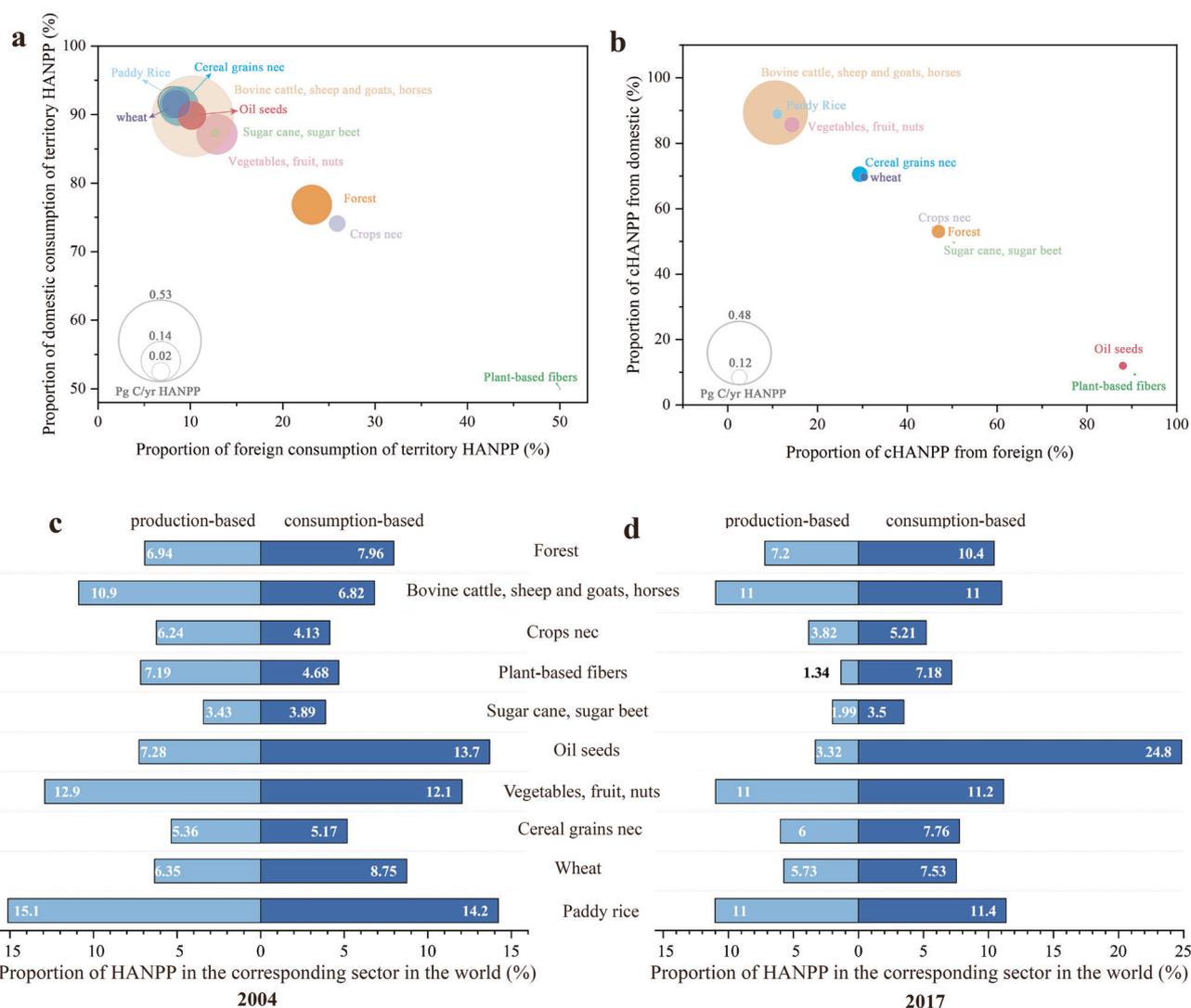


Fig. 2 | Utilization of production-based HANPP and sources of consumption-based HANPP in China by agricultural sector category in 2017. a Proportion of HANPP from China’s agricultural sector consumed domestically and globally. **b** Proportion of consumption-based HANPP in China impacting different agricultural sectors domestically and globally. The size of each circle indicates the

amount of production-based HANPP in that sector (a) and the amount of consumption-based HANPP (b). Production-based and consumption-based HANPP for different sectors in China accounts for the proportion of the global HANPP in that sector in 2004 (c) and 2017 (d).

It can be seen that the structure of production-based HANPP utilization in China varies widely across agricultural sectors (Fig. 2a). For example, almost all production-based HANPP from “Paddy rice” sector was used by China itself (91.8% of “Paddy rice” production-based HANPP), while “Plant-based fibers” were the sector with the most production-based HANPP used abroad (50.0% of “Plant-based fibers” production-based HANPP). The “Bovine cattle, sheep and goats, horses” sector in China has the largest production-based HANPP, of which 89.8% is consumed domestically and 10.2% is consumed foreign. China’s consumption-based HANPP mainly comes from the “Bovine cattle, sheep and goats, horses” sector (0.48 Pg C/yr), “Cereal grains nec” sector (0.12 Pg C/yr) and “Vegetables, fruit, nuts” sector (0.12 Pg C/yr). In terms of domestic and foreign source sectors for China’s consumption-based HANPP, we found that China has a high dependence on foreign countries for HANPP in the oil seeds and fibers sectors, with 88.0% of the HANPP demand in the oil seeds sector and 90.6% of the HANPP demand in the fibers sector being met by foreign countries (Fig. 2b).

From 2004 to 2017, we observed a decline in the contribution rates to the global HANPP total from seven out of China’s ten agricultural sectors. This phenomenon is primarily due to the relative stability in these sectors’ HANPP totals within China, coupled with an increase in the global HANPP

for these sectors during the same period. Among them, the plant-based fiber sector had the largest decline, from 7.2% in 2004 to 1.3% in 2017. In 2017, the “Paddy rice” in China produced the highest share of HANPP globally, producing 11.0% of the world’s HANPP in the “Paddy rice” sector (Fig. 2c).

In 2017, China consumed 24.8% of the world’s HANPP in the “Oil seeds” sector, 11.4% of the world’s HANPP in the “Paddy rice” sector, and 11.2% of the world’s HANPP in the “Vegetables, fruit, nuts” sector. Between 2004 and 2017, six of the ten agricultural sectors in China consumed an increased share of HANPP in the world’s HANPP in that sector. Among them, the share of HANPP consumed in the “Oil seeds” sector increased the most, from 13.7% in 2004 to 24.8% in 2017 (Fig. 2d).

Figure 3 depicts the sectoral distribution of HANPP in trade between China and other regions (for regional classifications, see Supplementary Table 1). In 2017, 39.3% of the 0.14 Pg C/yr HANPP China exported was from cattle, sheep, goats, and horse products, indicating their significance to markets including the USA, Japan, North Korea, and Europe. While overall HANPP exports declined, the “Cereal grains nec”, “Bovine cattle, sheep and goats, horses” and “Forest” sectors witnessed an increase in HANPP exports from 2004 to 2017, particularly to Asian countries such as Kyrgyzstan, Vietnam, and India.

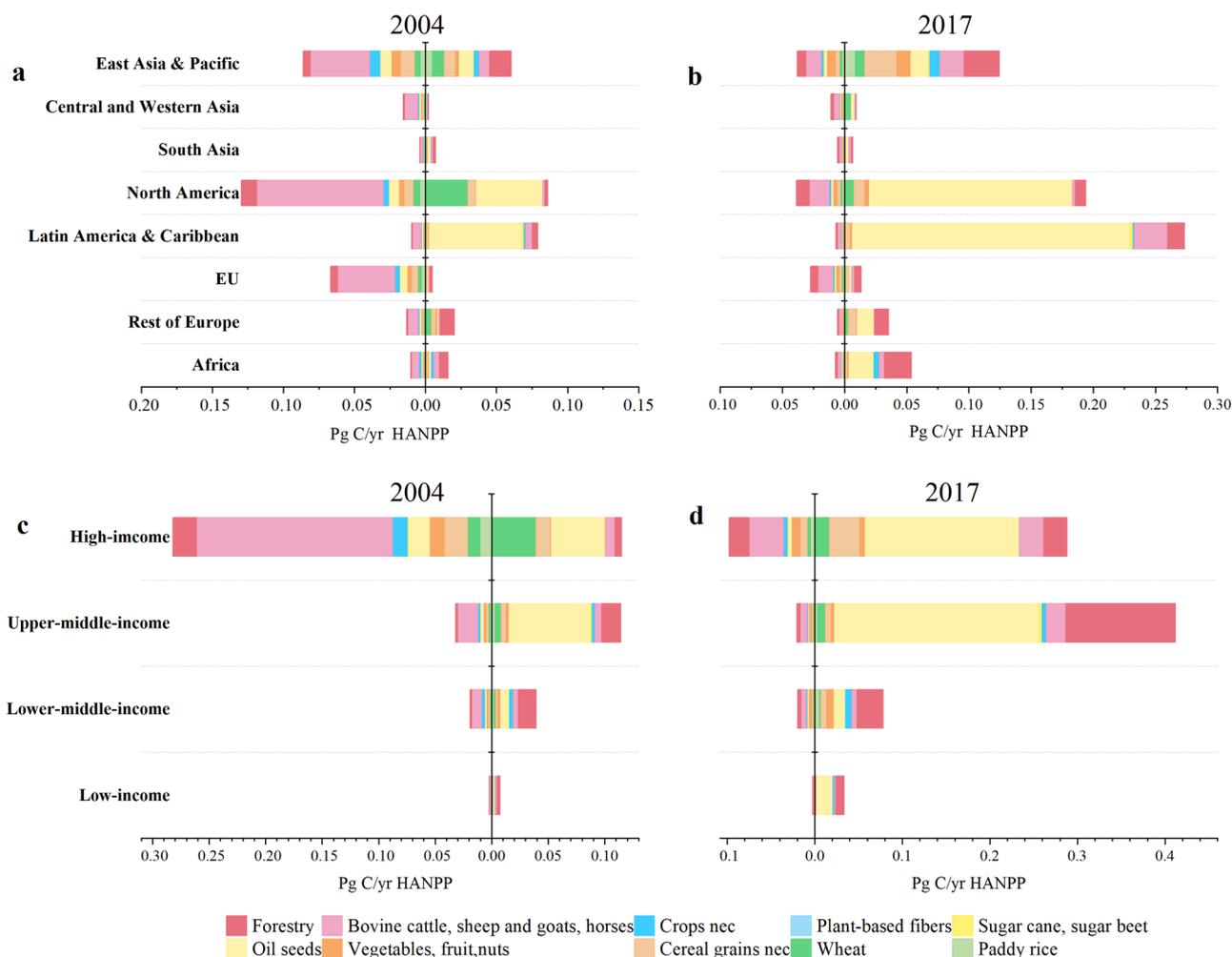


Fig. 3 | The magnitude and composition of HANPP embodied in trade between China and other global regions. The magnitude and composition of HANPP embodied in trade between China and other global regions in 2004 (a) and 2017 (b), and different income groups in 2004 (c) and 2017 (d). The left side of the axis indicates the amount of HANPP exported by Chinese agricultural sectors to each

region of the world/ different income groups, and the right side indicates the amount of HANPP imported by Chinese agricultural sectors to each region of the world/ different income groups. The classification of regions and income groups is based on the World Bank’s classification.

On the import side, due to domestic soybean shortages, 61.9% of the 0.71 Pg C/yr HANPP imported to China in 2017 was in oilseed products, primarily from Brazil, the USA, Canada, Argentina, and Africa. China’s rising oilseed demand resulted in HANPP increases of 0.12 and 0.16 Pg C/yr from North America, Latin America & Caribbean between 2004 and 2017. The East Asia and Pacific region, particularly Australia, New Zealand, and Vietnam, remained a primary source of China’s HANPP imports across various agricultural sectors.

Our analysis of the origins of China’s HANPP imports uncovers that a substantial portion comes from high-income and upper-middle-income nations. As of 2017, the proportions of HANPP imports from these high-income and upper-middle-income territories stood at 41% and 44%, respectively. Nonetheless, an emerging trend of importance is the escalating share of HANPP imports sourced from low-income countries. The ratio of HANPP imports from these countries witnessed a remarkable rise from 3% in 2004 to 5% in 2017, marking a total import amount increase of 355%. This shift indicates a broadening of China’s ecological footprint, increasingly drawing on the resources of lower-income nations. This shift also raises concerns about the ecological impacts on these regions, which may be ill-equipped to handle the environmental stress associated with high HANPP extraction. It also underscores the need for sustainable and equitable trade practices to mitigate the growing ecological pressures on these countries. China’s

evolving HANPP trade dynamics thus have profound implications for global efforts to manage ecosystem health and resource sustainability.

Breaking it down sectorally, China’s imports in the sectors of “Paddy rice,” “Vegetables, fruit, nuts,” “Plant-based fibers,” and “Crops nec,” are notably attributed to lower-middle and low-income nations. In 2017, the portions of China’s import volumes in these sectors sourced from lower-middle and low-income countries stood at 62%, 48%, 56%, and 64%, respectively.

When evaluating these ratios over time, it is apparent that China’s reliance on HANPP imports from lower-middle and low-income countries is augmenting. Between 2004 and 2017, five out of the ten surveyed agricultural sectors experienced an uptick in the proportion of imports originating from these countries. Most strikingly, the “Paddy rice” sector saw a 20% increase in the proportion of imports sourced from lower-middle and low-income nations.

Projections of China’s future HANPP

Our projections indicate that by 2050, China’s consumption-based HANPP may surge to 7.3 Pg C/yr (5.5 Pg C/yr –9.1 Pg C/yr), a staggering 317.6% (214.7–420.6%) rise from 2017 (Supplementary Fig. 4). Despite a slight dip in global share by 2050 to 9.0% (8.8–9.2%), China remains the world’s top HANPP consumer (Fig. 4). This anticipated rise in HANPP consumption highlights the profound and expanding ecological footprint of China on a

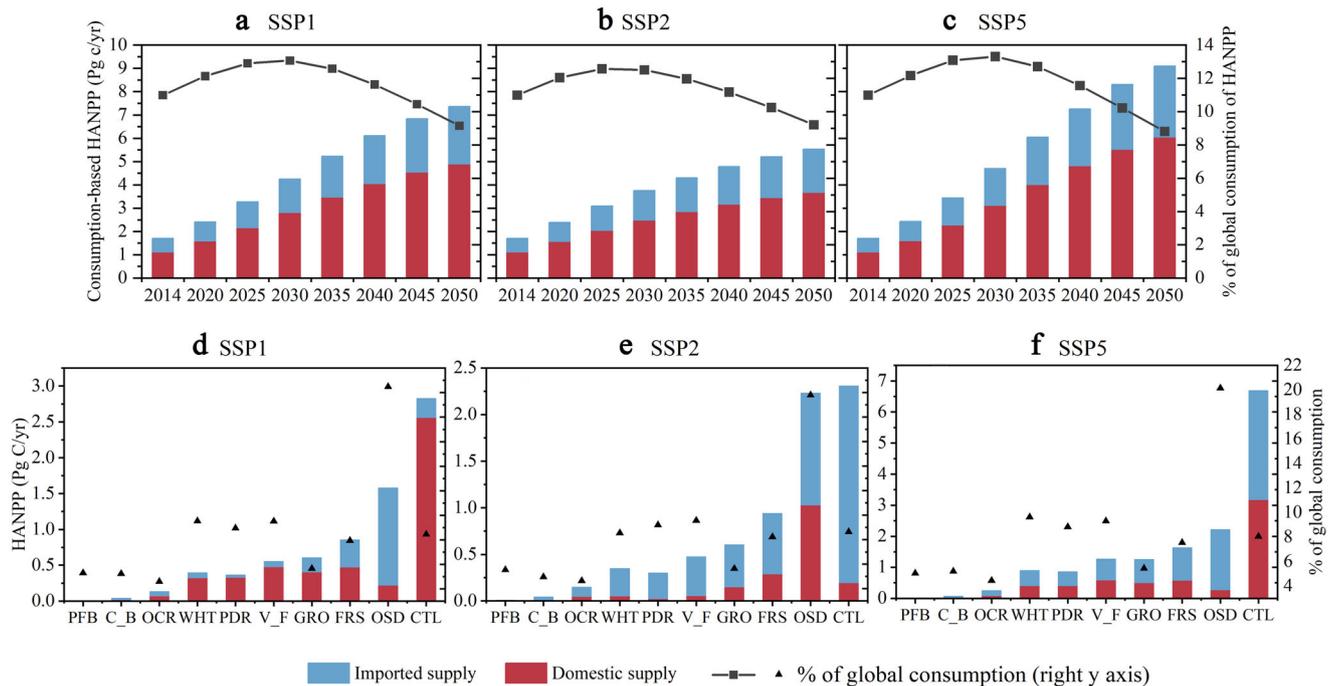


Fig. 4 | Projected HANPP in China. Projected HANPP in China under SSP1 (a), SSP2 (b), and SSP5 (c). Projected sources of consumption-based HANPP in China by agricultural sector category in 2050 under SSP1 (d), SSP2 (e), and SSP5 (f).

global scale. By 2050, the projected import proportion of HANPP consumed by China will reach 33.6% (33.5–33.7%), slightly lower than the 2017 ratio (40.7%), suggesting a minor shift towards reduced external biomass reliance possibly due to enhanced domestic production efficiency or sustainable policy implementation.

Assessing future HANPP inflow and outflow dynamics, China’s HANPP inflow globally surpassed the combined inflow of the two countries ranked next, emphasizing the vast biomass imports required to support China’s rapid population and economic growth, and potentially exposing China to vulnerabilities regarding external.

China’s import of HANPP from High-income, Upper-middle-income, Lower-middle-income, and Low-income countries in 2050 increased by 257.6%, 179.6%, 396.0%, and 418.7%, respectively, compared to 2017. Overall, there was a significant growth of 402.9% in HANPP import from Lower-middle-income and Low-income countries. This trend underscores China’s increasing reliance on biomass imports to meet its growing demands, particularly from economically disadvantaged regions.

Analyzing sectoral HANPP consumption, China is anticipated to consume 20% of the global Oil seeds sector’s HANPP by 2050, reflecting its burgeoning oil seeds demand, likely driven by its fast-growing population and dietary shift towards oil-rich foods. In addition, 9.5%, 9.2%, and 9% of HANPP from Vegetables, fruit, nuts, Wheat, and Paddy rice sectors, respectively, are projected to be consumed by China (Fig. 4), underlining its dependence on a diverse range of agricultural products.

Looking at China’s HANPP sectoral consumption, the Bovine cattle, sheep and goats, horses sector is the most prominent source under the SSP5 scenario, contributing 44% to China’s HANPP consumption. This shows a strong preference for livestock-based products in this socioeconomic pathway. The Oil seeds sector is the second-largest contributor, particularly under the SSP2 scenario, denoting its crucial role in China’s food and industrial sectors.

Under the SSP1 scenario, local supply satisfies a substantial portion of HANPP for most agricultural sectors, indicating a relatively sustainable and self-reliant agricultural sector. Conversely, under SSP2, most of China’s HANPP is imported across sectors, with import ratios exceeding 53%, suggesting a higher reliance on global markets. Lastly, the SSP5 scenario also

shows high import ratios, indicating a sustained reliance on external biomass supply, albeit moderated by more robust domestic production capabilities.

Under the 2050 SSP1 scenario, China mainly relies on HANPP imports from East Asia & Pacific, North America, and Latin America & Caribbean, with a balanced demand for diverse agricultural products from East Asia & Pacific, and oil seed products from North America and Latin America & Caribbean (Fig. 5).

Under SSP2, China’s HANPP imports decrease while exports increase. Exports under this scenario are dominated by the Bovine cattle, sheep and goats, horses sector, and the Forestry sector, suggesting China’s comparative advantage in these sectors is possibly due to improved livestock and forestry management.

In SSP5, trade between China and the regions of East Asia & Pacific, North America, and Latin America & Caribbean is highly imbalanced, with China’s imports of HANPP from these regions greatly surpassing its exports. This shows an increased dependence on these regions to meet China’s growing biomass demand in this high-growth, fossil-fueled socioeconomic pathway.

Driving forces of consumption-based HANPP

With the economy’s development and income growth, there is typically an upsurge in demand for domestic and foreign products within a country, leading to an increase in the country’s cHANPP. This is especially true in emerging nations such as China. Our findings corroborate this phenomenon, indicating that from 2004 to 2017, the main driver behind the growth of China’s overall HANPP was the marked increase in domestic consumption. This trend contributed to 104.3% of the total value of production-based HANPP and a notable 141.0% to the total value of consumption-based HANPP (Supplementary Figs. 5 and 6).

However, this growth was mitigated to some extent by several key factors, such as changes in domestic and foreign HANPP intensities. Over the period from 2004 to 2017, a decline in domestic HANPP intensity had a substantial impact in alleviating HANPP, resulting in a 117.9% decrease in production-based HANPP and a 104.8% decrease in consumption-based HANPP. Furthermore, between 2004–2007, 2007–2011, and 2011–2014, our results showed that in addition to a decline in domestic HANPP

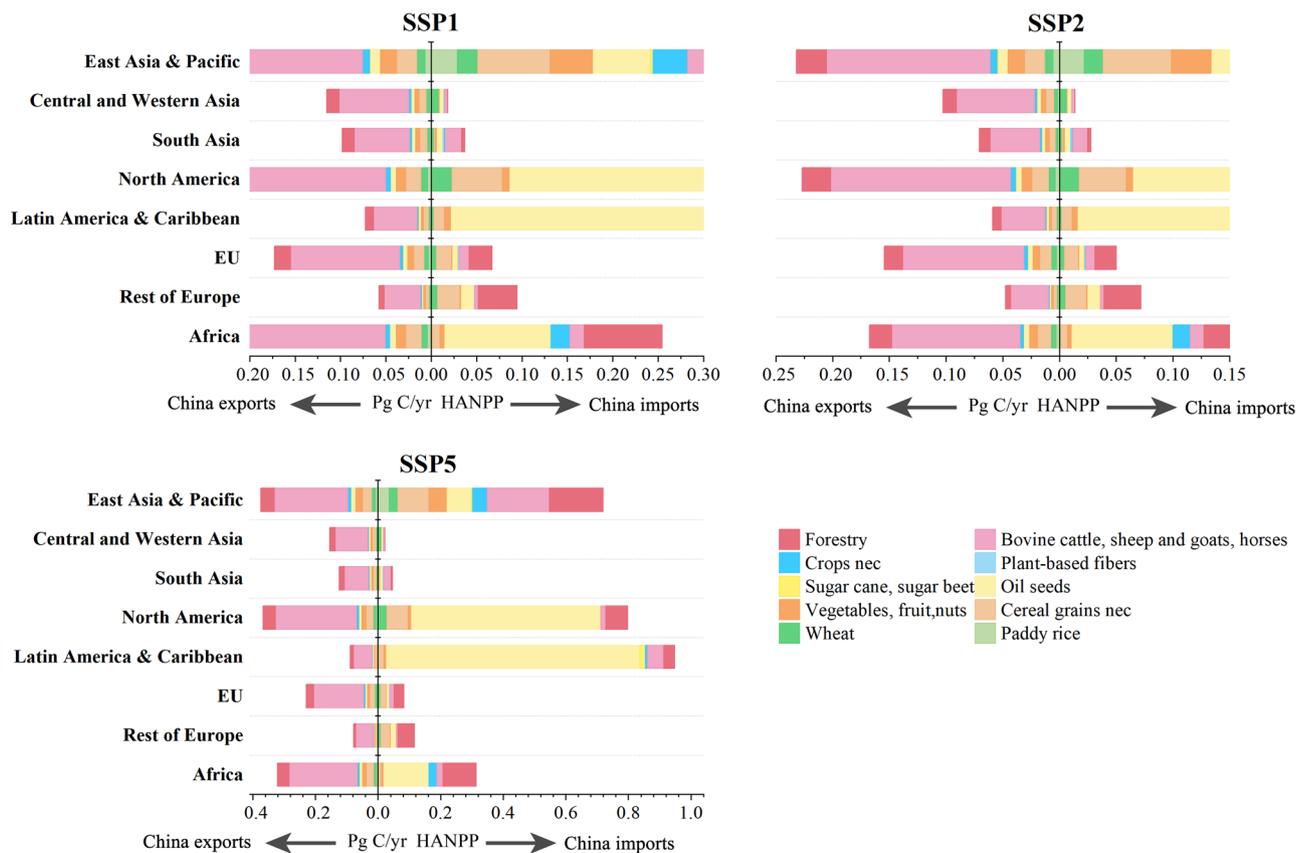


Fig. 5 | The magnitude and composition of HANPP embodied in trade between China and other global regions in 2050 under three SSPs. The left side of the axis indicates the amount of HANPP exported by Chinese agricultural sectors to each region of the world, and the right side indicates the amount of HANPP imported by Chinese agricultural sectors to each region of the world.

intensity, a decrease in foreign HANPP intensity also exerted a downward influence on consumption-based HANPP (Fig. 6).

The modifications in domestic consumption patterns and production structures played a crucial role in offsetting a majority of the HANPP growth driven by the increased consumption level throughout the period. We found that improvements in domestic consumption patterns and production structures between 2004 and 2007 reduced consumption-based HANPP by 5.2% and 21.7%, respectively. Even though the rise in consumption level is a potent driving force, these improvements contributed to the balanced growth of HANPP.

Importantly, foreign production and consumption structures began to have an increasingly substantial effect on China’s consumption-based HANPP over the period. From 2011 to 2014, changes in foreign production and consumption structures collectively led to an 8.9% increase in consumption-based HANPP, and from 2014 to 2017, alterations in foreign consumption structures contributed to an 8.6% growth in consumption-based HANPP, whereas shifts in foreign production structures resulted in a 13.4% decrease in consumption-based HANPP. This highlights the critical role of China’s trading partners’ production technology in reducing China’s consumption-based HANPP. Consequently, efforts to alleviate China’s ecosystem pressures through a reduction in consumption-based HANPP must consider both domestic initiatives and cooperation with trading partners in terms of technological upgrades and the sensitive zoning of production areas.

Discussion

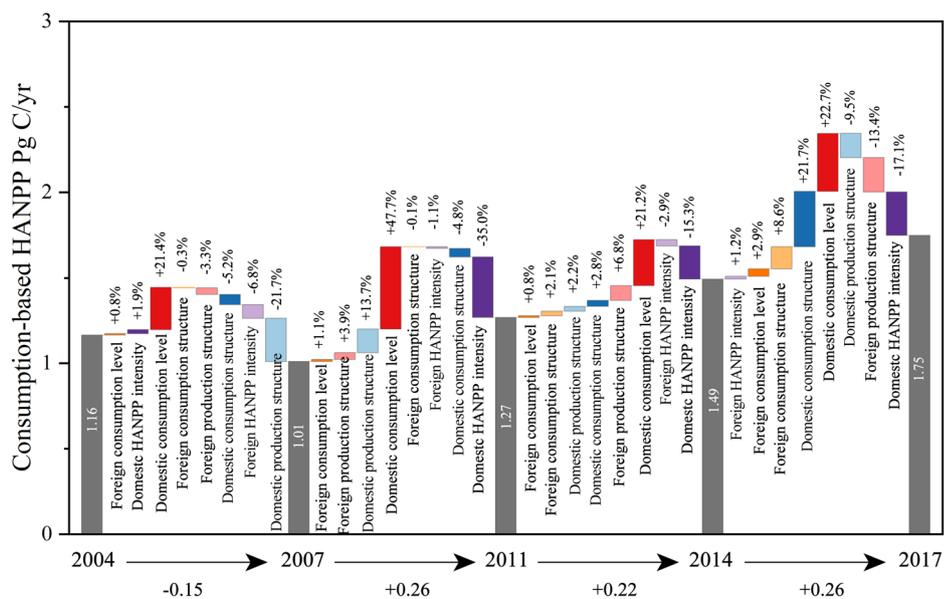
China has the third-highest production-based HANPP (1.18 Pg C/yr) and the highest consumption-based HANPP (1.75 Pg C/yr) in the world in 2017, which highlights its substantial role in the world’s ecological dynamics. China’s production and consumption activities have a profound impact not

only on its ecosystems but also on global HANPP through the complex network of international trade and global supply chains. During the period 2004–2017, China’s HANPP pattern has undergone important changes. There was a slight decrease in production-based HANPP and a marked increase in consumption-based HANPP, with the share of total global HANPP increasing from 8.4% in 2004 to 11.8% in 2017, underscores the escalating ecological footprint of its growing demands, impacting not only its ecosystems but also those globally through international trade. This signifies that, in any efforts to manage or regulate global HANPP, China stands as a highly important country whose involvement would be essential.

As the world’s second-largest economy, China has shifted from being a net exporter of HANPP to a net importer of HANPP to meet its growing demand for biomass for food, feed, fiber, and bioenergy. China’s HANPP imports are about five times as large as its exports. China consumed 24.8% of the world’s HANPP in the “Oil seeds” sector, 11.4% in the “Paddy rice” sector, 11.2% in the “Vegetables, fruit, nuts” sector, and 11% in the “Bovine cattle, sheep and goats, horses” sector. Between 2004 and 2017, six of the ten agricultural sectors in China consumed an increased share of HANPP in the world’s HANPP in that sector. The share of HANPP consumed in the “Oil seeds” sector increased the most, from 13.7% in 2004 to 24.8% in 2017.

Previously, China heavily relied on imports from high-income and upper-middle-income countries to meet its HANPP requirements. However, in recent years, we have observed a notable increase in China’s imports of HANPP from lower-middle and low-income countries, particularly in sectors such as “Paddy Rice”, “Vegetables, Fruit, Nuts”, “Plant-Based Fibers”, “Crops NEC”, and “Forestry”. For example, the “Paddy rice” sector saw a 20% increase in the proportion of imports sourced from lower-middle and low-income nations from 2004 to 2017. This shift could be driven by several factors. First, lower-middle and low-income countries often possess abundant natural resources, which align with China’s needs. Moreover,

Fig. 6 | Contributions of different factors to changes in consumption-based HANPP. The gray columns show the HANPP values for selected years; the colored columns show the contribution of the different drivers.



being in developmental stages, these countries could potentially offer goods and services at more competitive prices compared to their more developed counterparts. In addition, China has been actively fostering tighter trade relationships with these countries, contributing to the rapid growth of South-South trade²⁶.

However, this shift in dependency is not without its challenges. Compared to affluent countries, lower-middle and low-income countries tend to have a much higher HANPP intensity, meaning they allocate a greater proportion of their ecological capacity to human activities, often at the expense of local biodiversity and ecosystem health. As this trend continues, China’s expanding ecological impact could inadvertently incentivize these countries to further increase their HANPP intensity to meet China’s growing demands, potentially leading to a destructive cycle of escalating environmental impact^{27–29}. To mitigate this impact, China and its trade partners must collaborate and enforce sustainable environmental practices.

By 2050, China’s HANPP is projected to increase by 317.6% (214.7–420.6%) compared to 2017, further consolidating its position as the world’s leading HANPP consumer. This dominant role suggests that any changes in China’s HANPP consumption will have substantial implications for global ecological pressures.

Our study reveals that, depending on the SSP scenario, China’s HANPP consumption is largely influenced by specific agricultural sectors. Oil seeds and the livestock sector (bovine cattle, sheep and goats, horses) emerge as key contributors to China’s HANPP demand, highlighting the need for interventions focused on these sectors. The increasing demand for oil seeds, possibly driven by dietary changes and industrial uses, suggests the potential for enhancing efficiency and sustainability within this sector. Similarly, given the substantial contribution of the livestock sector to HANPP, strategies to optimize livestock management and promote more sustainable dietary choices could be instrumental.

From a regional perspective, East Asia & Pacific, North America, and Latin America & Caribbean stand out as primary suppliers of HANPP to China, indicating their important role in China’s food and biomass supply chains. Policy measures fostering regional cooperation and sustainable agriculture practices could help reduce the ecological pressures associated with these trade flows.

Our study offers a deeper understanding of the factors influencing China’s consumption-based HANPP from 2004 to 2017, which could help assess the potential impacts of different forces on reducing HANPP in the future. In China, the growth of domestic product demand has been the most crucial factor driving the increase of consumption-based HANPP. Over time, the influence of foreign HANPP intensity on China’s consumption-

based HANPP has first been negative and then positive, primarily due to the strengthening of South-South trade and increased dependence on lower-middle and low-income countries. This suggests that for China to decrease its consumption-based HANPP, it needs to direct its trade towards regions of higher efficiency or help trade partners achieve cleaner production through technological collaboration. In addition, domestic consumption patterns also play a important role in the growth of consumption-based HANPP. Solutions could include reducing food waste, shifting away from heavy meat diets.

The increase in China’s consumption-based HANPP highlights its growing ecological footprint, a consequence of rapid economic growth and a large population. As the world’s foremost HANPP consumer, China’s consumption habits notable impact global ecosystems. This externalization of ecological impacts means that the environmental consequences of China’s consumption are increasingly shouldered by exporting countries, particularly those in lower-middle and low-income categories, which often face higher HANPP intensities. As China sources more of its biomass from these countries, the strain on their ecosystems intensifies. This increased pressure can lead to a cascade of ecological issues. For instance, in countries with high HANPP intensities, the demands of export can accelerate deforestation, leading to habitat loss and a decline in biodiversity. Such environmental degradation not only undermines the ecological integrity of these regions but also affects global biodiversity and ecosystem services. The broader implications of this trend are profound. It underscores the need for a global perspective on resource consumption and ecosystem management. As the world’s ecosystems are interconnected, the environmental impacts in one region can have far-reaching effects.

To harmonize its socio-economic advancement with ecological sustainability, China must implement strategic interventions. This involves not only enhancing efficiency in production processes to reduce its domestic HANPP footprint but also adopting policies that foster sustainable consumption patterns. For example, investments in innovative agricultural technologies can yield higher efficiencies while reducing ecological impacts. In addition, policy measures encouraging the adoption of eco-friendly products and practices can alter consumption patterns within the country.

Further, in transitioning towards sustainable practices, it is crucial to recognize that China, as a developing country, needs technological assistance and support from developed nations with which it engages in trade. This collaboration is not merely about import and export but also encompasses knowledge exchange, financial investments, and joint research and development projects. Developed countries like the United States, Canada, and nations in Western Europe, which are key trade partners with China,

can play a notable role in providing this support. Their advanced technologies and established practices in sustainability can greatly benefit China in enhancing its ecological practices. This form of global cooperation, rooted in trade relationships, is vital for China to effectively transition towards more sustainable practices.

Simultaneously, China holds a crucial position in supporting the sustainable development of low-income countries, many of which are its trading partners in regions like Southeast Asia and Africa. Leveraging its own developmental experience and resources, China can offer technological support, capacity building, and financial aid to these nations. This support is especially pertinent in the context of trade, as it can lead to more sustainable production practices in these countries, which in turn, reduces the global ecological impact of HANPP. By assisting these low-income countries in developing sustainable practices, China not only contributes to reducing the ecological footprints linked to its imports but also fosters a more collaborative and supportive international trade environment for sustainable development.

Therefore, China's approach to reducing its HANPP impact requires a blend of internal reforms and external collaboration. With assistance from developed countries and by extending support to developing nations, China can contribute substantially to global ecological sustainability. This interconnected approach underscores the necessity of international cooperation and collective effort in addressing global ecological challenges.

The limitations of this research are primarily centered around two key aspects: Firstly, our study relies on data from the Global Trade Analysis Project database, which was limited to the year 2017. The lack of more recent data means that our findings might not fully reflect the current or emerging trends in global HANPP dynamics. Secondly, while our research provides insights into HANPP's evolution, it is important to note that the future trajectory of HANPP will be influenced by a multitude of factors, including population and economic growth, dietary changes, shifts in energy structures (like the adoption of Bioenergy with Carbon Capture and Storage, BECCS), and changes in international trade patterns. Theoretically, an accurate prediction of HANPP would require considering all these variables. However, in practice, it is not feasible to cover every factor within the scope of a single study. As a result, our research primarily focuses on whether the current pace of technological advancement is sufficient to offset the increased demand stemming from population and economic growth, without taking into account other potential influencing factors.

Methods

Production-based HANPP approach

The calculation of HANPP involves the following parameters: the potential NPP of the ecosystem assuming no human activity (denoted as NPP_{pot}), the NPP of the actually distributed vegetation (denoted as NPP_{act}), the NPP remaining in the ecosystem after human occupation (denoted as NPP_{eco}), the NPP harvested by human activity (denoted as HANPP_{harv}), and the change in NPP due to land use and conversing (denoted as HANPP_{luc}). From a socioeconomic perspective, HANPP is the sum of HANPP_{luc} and HANPP_{harv}. From the ecological point of view, it is the difference between NPP_{pot} and NPP_{eco}^{20,30}.

$$\text{HANPP} = \text{NPP}_{\text{pot}} - \text{NPP}_{\text{eco}} \quad (1)$$

$$\text{HANPP} = \text{HANPP}_{\text{luc}} + \text{HANPP}_{\text{harv}} \quad (2)$$

$$\text{HANPP}_{\text{luc}} = \text{NPP}_{\text{pot}} - \text{NPP}_{\text{act}} \quad (3)$$

In this study, we calculated HANPP as the sum of HANPP_{luc} and HANPP_{harv}, where HANPP_{luc} was calculated as the difference between NPP_{pot} from NPP_{act}.

HANPP_{harv} comprises biomass harvested during the year and the biomass destroyed during harvest^{17,20}. The scope of the biomass resource was classified into two main categories, i.e., primary product harvest, and

used and unused harvest residues. Raw biomass demand and production information data were obtained from the Food and Agriculture Organization of the United Nations (FAO) and converted to dry-weight basis values based on the dry-weight conversion factor provided by Zhou et al.³¹. Used and unused harvest residues were derived from the primary product harvest utilizing the specific harvest coefficients. Below-ground residues on cropland were measured as 15% of above-ground¹⁸. Grazing biomass is the biomass consumed by ruminants directly through grazing or indirectly through harvesting hay or silage¹⁸. The data for the calculation of grazing biomass were obtained from FAO and the detailed methodology and correlation coefficients for the calculation can be found in Zhou et al.³¹. In this study, the biomass from forest harvesting consists of two components: biomass from wood material and biomass from energy. Relevant data on industrial roundwood production and fuelwood can be obtained from the FAO Forestry Production Database. The following coefficients were used to calculate dry matter biomass of forestry products: wood density of 0.725 t/m³, moisture content of wood of 20%, and the proportion of wood waste on site of 40%³¹. In addition, the average carbon content of dry matter biomass was assumed to be 50%. HANPP_{harv} in urban areas (e.g., lawn mowing and tree cutting) was assumed to be 50% of NPP_{act}^{17,18}.

We calculated NPP_{pot} using the Lund-Potsdam-Jena Dynamic Global Vegetation Model (LPJ-DGVM), which is now widely used in HANPP studies¹⁸. LPJ-DGVM was developed based on the BLOME family of models, which combines process-based large-scale representations of terrestrial vegetation dynamics and land-atmosphere carbon and water exchange in a modular framework³². In this study, LPJ-DGVM is extended to 1000 years of processes to equilibrate vegetation cover and carbon pool fluxes, with model-driven data including climate data (temperature, precipitation, clouds, and wet days), soil property data, and carbon data (Supplementary Table 3). And the estimation results of LPJ-DGVM of this study are compared with the NPP_{pot} measured by Haberl et al.¹⁷ to verify the reliability of the model. The sample area was chosen to be in a region with constant plant functional types from 2000 to 2014. The comparison results showed that the NPP_{pot} calculated in this paper was significantly correlated with previous studies (R = 0.868; N = 5475; P < 0.001). In addition, the land use type data were obtained from the Historical Database of the Global Environment (HYDE)³³, which is consistent with the statistical caliber of the Food and Agriculture Organization of the United Nations (FAO).

HANPP_{luc} is calculated as the difference between NPP_{pot} and NPP_{act}. We calculated HANPP_{luc} based on the method developed by Krausmann et al.¹⁸. NPP_{act} on cropland is the sum of HANPP_{harv} and pre-harvest losses caused. The loss expansion factor used to calculate pre-harvest losses was assigned based on the fertilizer use provided by FAO. The NPP_{pot} of cropland (calculated from HYDE data) can be divided into two parts according to the ratio of fallow to cropland area. As for the HANPP_{luc} on grassland, this paper calculates it in two parts, one being the NPP loss due to conversion from forest to artificial grassland (grazing land in HYDE 3.2). This conversion was assumed to result in a 20% reduction in NPP. The second is the NPP loss due to human-induced soil degradation on grazed land. Information on the extent of soil degradation was obtained from the Global Assessment of Human-Induced Soil Degradation (GLASOD). It is assumed that NPP is reduced by 25%, 45%, 65%, and 85% depending on the degree of land degradation^{17,18,34}. The HANPP_{luc} in urban areas was assumed to be 2/3 of the NPP_{pot}. HANPP_{luc} in forests and other nature was defined as zero.

We developed a HANPP satellite account matched with GTAP's MRIOs to obtain HANPPs for ten agricultural sectors (see Supplementary Table 2 for specific agricultural sectors). The matching steps were as follows: first, HANPP data for grasslands were classified into GTAP's animal husbandry and HANPP data for various forest lands were classified into GTAP's forestry. Second, the HANPP of cultivated land in the HANPP satellite account was classified to each agricultural sector of GTAP using national agricultural area data provided by the Food and Agriculture Organization of the United Nations (FAO).

Consumption-based HANPP approach

Existing studies calculated embodied HANPP mainly based on physical trade flow approaches, those methods do not, however, take into account the full supply chain flows from production to consumption^{20,35}. To address this gap, we utilize multi-regional input–output (MRIO) analysis, a method that allocates HANPP transfers occurring in production and the supply chain to the final consumers of goods³⁶. MRIO method is now widely used to account for the environmental impacts embodied in international trade^{37–39}.

Here, we coupled a Lund-Potsdam-Jena Dynamic Global Vegetation Model (LPJ-DGVM) and an environmentally extended MRIO model to account for consumption-based HANPP. The LPJ-DGVM is utilized to simulate the potential natural vegetation NPP (NPP_pot). This model provides a dynamic and comprehensive understanding of how natural vegetation biomass production could evolve under different environmental conditions. On the other hand, the environmentally extended MRIO model is employed to trace the flows of HANPP through global supply chains. This model allows us to map the transfer and distribution of HANPP across different regions and sectors, linking the consumption in one region to the ecological impacts in another.

The basic linear expressions in the MRIO framework can be expressed as:

$$X = AX + F$$

$$X = \begin{bmatrix} x^1 \\ x^2 \\ \dots \\ x^n \end{bmatrix}; A = \begin{bmatrix} A^{11} & A^{12} & \dots & A^{1n} \\ A^{21} & A^{22} & \dots & A^{2n} \\ \vdots & \vdots & \ddots & \vdots \\ A^{n1} & A^{n2} & \dots & A^{nn} \end{bmatrix}; F = \begin{bmatrix} f^{11} & f^{12} & \dots & f^{1n} \\ f^{21} & f^{22} & \dots & f^{2n} \\ \vdots & \vdots & \ddots & \vdots \\ f^{n1} & f^{n2} & \dots & f^{nn} \end{bmatrix}$$

where X represents the aggregate output vector and x^s is aggregate output of region s . A stands for the matrix of technical coefficient. And F denotes the final demand matrix.

The formula can be further expressed as:

$$X = (I - A)^{-1}F = LF$$

where I is the identity matrix and $L = (I - A)^{-1}$ called the Leontief inverse matrix, which comprehensively reveals the intricate economic relations among all sectors of the national economies.

The embodied HANPP of the final demand can be characterized by the environmentally extended MRIO model, as follows:

$$cHANPP = fLF$$

The LPJ-DGVM is used to compute the potential and actual NPP at a detailed spatial resolution (Detailed in the Production-based HANPP approach). Once we obtain the NPP values from the LPJ-DGVM, we then use them to determine the HANPP for each sector and region. The calculated HANPP values are then integrated into the environmentally extended MRIO model by Eq. (6). In this model, f is a row vector where each element represents the per-unit economic output of HANPP (HANPP intensity) for each sector in the economy. This integration allows us to map how HANPP is embodied in the products and services through global supply chains, attributing the environmental impacts back to the final consumer goods.

The advantage of this integrated approach lies in its ability to provide a comprehensive and spatially explicit assessment of the human impact on ecosystems. By combining the LPJ-DGVM's detailed ecological assessment with the environmentally extended MRIO model's economic and trade analysis, we can trace the global footprint of consumption patterns in a more nuanced and accurate way. However, it's important to note that the precision of our HANPP intensity estimates depends on the accuracy and resolution of the data from the LPJ-DGVM. The integration of these two models enables us to analyze both the potential ecological capacity (via LPJ-

DGVM) and the actual human appropriation of this capacity (via environmentally extended MRIO). This approach allows for a more nuanced understanding of the relationship between human activities and ecological changes on a global scale.

Structural decomposition analysis

In this paper, the SDA method is used to distinguish the contribution of different factors to the change in HANPP. The change in HANPP of a country within the base year 0 to the final year 1 can be expressed as:

$$HANPP = HANPP_1 - HANPP_0 = f_1L_1F_1 - f_0L_0F_0$$

F (final demand) in Eq. (7) can be further decomposed into consumption structure and consumption level. In addition, each factor can be decomposed into domestic and foreign components. Therefore, the change in HANPP can be decomposed into eight factors: domestic HANPP intensity (f^d), foreign HANPP intensity (f^f), domestic production structure (L^d), foreign production structure (L^f), domestic consumption structure (Fs^d), foreign consumption structure (Fs^f), domestic consumption level (Ff^d), and foreign consumption level (Ff^f).

$$\Delta HANPP = \Delta f^d f^f L^d L^f F_s^d F_s^f Ff^d Ff^f + f^d \Delta f^f L^d L^f F_s^d F_s^f Ff^d Ff^f + f^d f^f \Delta L^d L^f F_s^d F_s^f Ff^d Ff^f + f^d f^f L^d \Delta L^f F_s^d F_s^f Ff^d Ff^f + f^d f^f L^d L^f \Delta F_s^d F_s^f Ff^d Ff^f + f^d f^f L^d L^f F_s^d \Delta F_s^f Ff^d Ff^f + f^d f^f L^d L^f F_s^d F_s^f \Delta Ff^d Ff^f + f^d f^f L^d L^f F_s^d F_s^f Ff^d \Delta Ff^f$$

Δ represents the difference operator. All eight terms in Eq. (8) represent the contribution of the change factor to HANPP in the absence of changes in other drivers. In the SDA, there are several methods for performing decomposition in SDA. Following the previous studies, we use the average of two polar decompositions. According to the average of two polar decompositions, Eq. (8) can be rewritten as:

$$\Delta HANPP = \frac{1}{2} (\Delta f^d) (L_1 F_1 + L_0 F_0) + \frac{1}{2} (\Delta f^f) (L_1 F_1 + L_0 F_0) + \frac{1}{2} [f_1 \Delta L^d F_0 + f_0 \Delta L^d F_1] + \frac{1}{2} [f_1 \Delta L^f F_0 + f_0 \Delta L^f F_1] + \frac{1}{2} [f_1 L_1 \Delta F_s^d Ff_0^d + f_0 L_0 \Delta F_s^d Ff_1^d] + \frac{1}{2} [f_1 L_1 \Delta F_s^f Ff_0^f + f_0 L_0 \Delta F_s^f Ff_1^f] + \frac{1}{2} \Delta Ff^d (f_1 L_1 f_s^d + L_0 F_0 f_s^d) + \frac{1}{2} \Delta Ff^f (f_1 L_1 f_s^f + L_0 F_0 f_s^f)$$

Projecting HANPP under SSP scenarios

Our study utilizes Shared Socioeconomic Pathways (SSPs), a set of scenarios for integrated analysis of future climate impacts, vulnerabilities, adaptation, and mitigation characterized by socioeconomic trends including demographics, economic growth, education, urbanization, and technological change⁴⁰. We focus on SSP1 (“Sustainability—Taking the Green Road”), SSP2 (“Middle of the Road”), and SSP5 (“Fossil-fueled Development—Taking the Highway”), which respectively represent a shift towards sustainability, continuation of current trends, and economic growth-driven development.

In the succeeding part of our methodology, we leveraged the work of several scholars who have formulated predictions for population, economic growth, and urbanization for each SSP narrative on a national scale^{41,42}. These predictions have been integrated by the International Institute for Applied Systems Analysis (IIASA) into a unified SSP database. In this paper, we specifically utilized the predictions for national population and GDP from IIASA's SSP scenario database.

We model HANPP from 2020 to 2050, projecting under different SSP scenarios based on Eq. (6). According to Eq. (6), the future consumption-based HANPP is influenced by changes in sectoral consumption-based HANPP intensities (fB) and overall final demand. Utilizing a log-log model and data spanning from 2004 to 2017, we have estimated the average annual change rate in these sectoral consumption-based HANPP intensities. By projecting these rates of change forward, mirroring the 2004 to 2017 period, we aim to assess if the current pace of technological advancements can effectively mitigate the increasing demand driven by population and economic growth. Utilizing the population and GDP projections from the SSPs, we are able to project the final demand for each country. This projection incorporates the anticipated growth in demand due to changes in GDP and population, utilizing the income elasticity dataset from the Global Trade Analysis Project to estimate how this demand is distributed across different sectors.

Importantly, our study focuses specifically on exploring how HANPP might evolve if current production technology trends persist, under a range of future demographic and economic growth scenarios as depicted by the SSPs. In this context, our analysis does not account for other factors that could potentially influence HANPP, such as the adoption of new technology, shifts in dietary patterns, and changes in the global trade landscape.

Robustness and reliability

In our study, we assessed the robustness of our results by addressing key uncertainties and computing high and low estimates of HANPP. These uncertainties include variables from the LPJ-DGVM, the statistical data used, and the assumptions underlying our estimation methods.

To validate the LPJ-DGVM utilized in our study, we compared our estimated potential NPP in 2000 with the results obtained by Haberl et al.¹⁷ using the same model. We focused our comparison on areas where Plant Functional Types (PFTs) remained unchanged from 2000 to 2017. A significant correlation ($R = 0.868$; $N = 5475$; $P < 0.001$) was observed between our potential NPP and that of the previous study, indicating a high level of reliability in our results. In addition, to minimize the influence of extreme events, we used the average NPP_pot value from 2001 to 2018 for our high and low HANPP estimates.

Regarding the uncertainty in statistical data, especially in FAO's wood harvest data, we referred to the uncertainty range provided by Krausmann et al.⁴³. The uncertainty ranges for other statistical data and assumptions, particularly for crop-harvested biomass and grazed biomass, were derived from comparisons with previous studies^{18,43} and the variance in the high and low values of our parameter assumptions. Krausmann et al.⁴³ educated guesses on the likely uncertainty ranges for HANPP land use change (HANPP_lucc) were also incorporated.

By combining the average NPP_pot results with the high and low estimates for each item, we calculated the corresponding high and low estimates for HANPP, thereby ensuring a comprehensive and cautious approach to our estimations.

Data availability

The data used to calculate HANPP can be obtained at <http://www.fao.org/faostat/>. The History Database of the Global Environment (HYDE version 3.2) is available at <https://dataportal.pbl.nl/downloads/HYDE/>. The Global Assessment of Human-induced Soil Degradation (GLASOD) is available at <https://files.isric.org/public/other/GLASOD.zip>. GTAP data is available at <https://www.gtap.agecon.purdue.edu/databases/default.asp>. IIASA SSP Database is available at <https://tntcat.iiasa.ac.at/SspDb>. Source data required for reproducing the main figures is available at: <https://doi.org/10.6084/m9.figshare.25592049.v1>.

Code availability

All computer codes generated during this study are available from the corresponding authors on request.

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Author contributions

S.W. and C.F. designed the research; J.W. performed experiments and computational analysis; S.W., J.W. and C.F. drafted the paper; S.W., J.W., X.C., J.L., K.L., C.F., K.F. and K.H. contributed to the interpretation and the preparation of the manuscript, and all authors contributed to the final draft of the manuscript.

Competing interests

The authors declare no competing interests.

Additional information

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