Decreasing emissions and increasing sink capacity to support China in achieving carbon neutrality before 2060

Pengfei Han\textsuperscript{1,5*}, Ning Zeng\textsuperscript{2,5*}, Wen Zhang\textsuperscript{3}, Qixiang Cai\textsuperscript{1}, Ruqi Yang\textsuperscript{1}, Bo Yao\textsuperscript{4}, Xiaohui Lin\textsuperscript{3}, Guocheng Wang\textsuperscript{2}, Di Liu\textsuperscript{1}, Yongqiang Yu\textsuperscript{3}

\textsuperscript{1}Carbon Neutrality Research Center, Institute of Atmospheric Physics, Chinese Academy of Sciences, Beijing, China
\textsuperscript{2}Department of Atmospheric and Oceanic Science, and Earth System Science Interdisciplinary Center, University of Maryland, College Park, Maryland, USA
\textsuperscript{3}State Key Laboratory of Atmospheric Boundary Layer Physics and Atmospheric Chemistry, Institute of Atmospheric Physics, Chinese Academy of Sciences, Beijing, China
\textsuperscript{4}Meteorological Observation Centre, China Meteorological Administration, Beijing, China
\textsuperscript{5}State Key Laboratory of Numerical Modeling for Atmospheric Sciences and Geophysical Fluid Dynamics, Institute of Atmospheric Physics, Chinese Academy of Sciences, Beijing, China

\textit{*Correspondence: pfhan@mail.iap.ac.cn; zeng@umd.edu}
Abstract

In September 2020, President Xi Jinping announced that China strives to achieve carbon neutrality before 2060. This ambitious and bold commitment was well received by the global community. However, the technology and pathway are not so clear. Here, we conducted an extensive review covering more than 200 published papers and summarized the key technologies to achieve carbon neutrality. We projected sectoral CO₂ emissions for 2020-2050 based on our previous studies and published scenarios. We applied a medium sink scenario for terrestrial sinks due to the potential resource competition and included an ocean sink, which has generally not been included in previous estimates. We analyzed and revisited China’s historical terrestrial carbon sink capacity from 1980-2020 based on multiple models and a literature review. To achieve neutrality, it is necessary to increase sink capacity and decrease emissions from many sources. On the one hand, critical measures to reduce emissions include decreasing the use of fossil fuels; substantially increasing the proportion of the renewable energy and nuclear energy. On the other hand, the capacity of future carbon sinks is projected to decrease due to the natural evolution of terrestrial ecosystems, and anthropogenic management practices are needed to increase sink capacity, including increasing the forest sinks through national ecological restoration projects and large-scale land greening campaigns; increasing wood harvesting and storage; and developing CCUS. This paper provides basic source and sink data, and established and promising new technologies for decreasing emissions and increasing sinks for use by the scientific community and policy makers.

Keywords: Carbon neutrality; anthropogenic greenhouse gas emissions; terrestrial and ocean carbon sinks; negative emissions technology; renewable energy
1 Introduction

The Paris Agreement, adopted in 2015, aimed to limit global warming to much less than 2°C, and preferably less than 1.5°C, above preindustrial levels (UNFCCC, 2015). Governments agreed about the need to reach the global greenhouse gas (GHG) emissions peak as soon as possible and to achieve net zero emissions in the second half of this century. On 22 September 2020, President Xi Jinping announced on the virtual stage of the United Nations General Assembly that China is striving to become carbon neutral before 2060 (Xi, 2020b). At the Global Climate Ambition Summit in December 2020, he further announced four specific initiatives to support carbon neutrality from the perspective of reducing emissions and increasing sink capacity (Xi, 2020a). Achieving carbon neutrality before 2060 for China will require greater efforts than for developed countries because developed countries have generally had a transition period of 50-70 years from peak emissions to carbon neutrality, while China will have only 30 years. After 2030, China's annual emission reduction rate will have to reach 8-10%, which is much higher than the reduction rate for developed countries (He et al., 2020).

Internationally, U.N. Secretary-General Antonio Guterres called on all countries to enter into a state of climate emergency and to establish a global carbon neutrality alliance (UN, 2020). In early 2021, at least 85 countries responsible for 65% of the total global CO₂ emissions, including the European Union, China, the United Kingdom, Japan, South Korea, and the U.S. (Deng et al., 2020), announced carbon neutrality commitments, strategies or plans. In March 2020, the EU submitted a "Long-Term Low Greenhouse Gas Emissions Development Strategy" to the UN Framework Convention on Climate Change; this strategy is guided by the “European Green Deal”, and Europe is striving to become the first climate-neutral continent by 2050 (European Commission, 2019). The European Green Deal is essentially a new economic growth strategy that features revolutionizing energy technology; vigorously developing renewable energy, nuclear energy, and green hydrogen energy; and
gradually abandoning the use of fossil fuels. These aims are supported by medium- and long-term GHG reduction targets that are deeply integrated with energy production as well as dynamic monitoring, assessment and reporting systems. Achieving carbon neutrality will require key improvements in energy efficiency; significant increases in the terminal electrification rate; the development and widespread adoption of renewable energy (e.g., hydropower, wind, solar) and nuclear power; carbon capture, usage and storage (CCUS) strategies; and new, low-carbon transportation technologies (Cao et al., 2016; He et al., 2020; Jiang et al., 2020; Li, X.-z. et al., 2018; Yang et al., 2017; Yao et al., 2016; Zhang et al., 2017). Currently, China’s total CO$_2$ emissions are about 10 Pg CO$_2$ yr$^{-1}$, and the power (40%) and industry (44%) sectors contribute most to the total emissions, while the transportation sector contributes 9%, and the residential and commercial sector contributes 7% (Han et al., 2020; Shan et al., 2020). As for the sinks, the terrestrial sink is generally estimated as 0.2-0.3 Pg C yr$^{-1}$ (Fig.1), which offset ~10% of total emissions (Fang et al., 2018; Piao et al., 2009). The sink is mostly contributed by forest through national ecological restoration projects (Lu et al., 2018) and cropland soils through enhanced straw return (Yu et al., 2012; Zhao et al., 2018), and the uptake from CCUS is currently only 1 Mt CO$_2$, and negligible compared with total emissions (Cai et al., 2020). There is a large gap between the present sources and sinks to achieve C neutrality in 30 years after CO$_2$ emissions peak.
Figure 1. CO₂ sources and sinks in a practical scenario of carbon neutrality in China before 2060. Data were derived from (He et al., 2020); (Han et al., 2020); (Fang et al., 2018); (Jiao, 2021; Jiao et al., 2018); (Hu et al., 2015); (Yu et al., 2012; Yu et al., 2013); and (Piao et al., 2009). Note that the sinks are selected as medium scenarios due to the resources competition and ocean sinks were generally not included in previous studies, and CC(U)S need to be expanded by astonishing hundreds of folds more than the 2020 stored amount.

However, to date, there is no comprehensive source of information about how to achieve carbon neutrality in terms of reducing emissions and increasing sink capacity for both CO₂ and non-CO₂. Here, we conducted an extensive review covering more than 200 published papers in order to present a state-of-the-art picture of GHG emissions and terrestrial and ocean carbon sinks. We then proposed suggestions for decreasing emissions and increasing sink capacity. This study provides both basic source and sink data and descriptions of developed and promising new technologies for decreasing emissions and increasing sink capacity for use by the scientific
community and policy makers.

2 Reducing anthropogenic CO₂ emissions

2.1 Reducing fossil fuel and industrial process emissions

2.1.1 Reducing fossil fuel emissions

China contributes 28% of global CO₂ emissions (10 Pg CO₂ yr⁻¹) (Friedlingstein et al., 2020; Guan et al., 2018; Han et al., 2020). Coal is the primary energy source in China, accounting for 57.7% of energy sources in 2019, which represents a decrease from 70% in 2000. However, the total coal consumption in China has increased from 1410.9 Mt in 2000 to 3857.2 Mt in 2017. To achieve carbon neutrality, nonfossil fuels will have to make up 81%–85% of energy sources by 2060, with wind and solar contributing >50% and being supplemented by hydropower, nuclear power and other sources. Meanwhile, coal will have to decrease to making up less than 5% of energy sources (He et al., 2020; State Grid, 2020). There is strong pressure in China to abandon coal-fired power plants, which consume 46.6% of the total national coal consumption. Since most existing coal-fired power plants were constructed from 2005-2015 and have a life span of 40 years, they will need to be retired in ~2035. Future conditions will become more challenging for coal-fired power plants, as constructing these plants now represents an unfavorable investment, and they are likely to face higher costs with the implementation of CCUS. Moreover, the construction of coal-fired plants may be banned after a certain deadline determined by policy makers.

One potential scenario for CO₂ emissions and sinks that would achieve specific carbon neutrality is depicted in Fig. 1. In this scenario, the total emissions are 1720 Tg (He et al., 2020), which is ~17% of the present CO₂ emissions, and power generation and industry (process) are the largest contributors. The sinks will largely be terrestrial sinks (700 Tg) (Hu et al., 2015; Xu et al., 2010; Yu et al., 2013), ocean sinks (480 Tg) (Jiao, 2021; Jiao et al., 2018) and CC(U)S (880 Tg) (He et al., 2020); however, large uncertainties remain around CCUS, which has not been implemented at a large scale.
and is still attracting critiques from researchers (Fuss et al., 2014; McLaren and Markusson).

The total emissions from fossil fuel combustion and industrial processes were 10200 Tg CO₂ in 2019 and decreased slightly (by 0.5-2%) in 2020 due to restrictions related to the COVID-19 pandemic (Fig. 2) (Friedlingstein et al., 2020; Han et al., 2021). Since the emission reductions associated with the pandemic restrictions are only temporary, CO₂ emissions are likely to return to their previous levels in reviving economies (Forster et al., 2020; Le Quéré et al., 2020). Therefore, structural changes in the energy system and economy as well as green and low-carbon pathways to economic recovery are needed. Peak CO₂ emissions, at approximately 10600 Tg CO₂, are predicted to be reached in approximately 2025-2027 (Fig. 2) (Lin et al., 2019; Wang and Yan, 2021). Transitioning from coal to natural gas would decrease carbon emissions in the short term by producing higher net heat values (Farquharson et al., 2017). However, natural gas consumption still emits CO₂ and will need to be reduced to achieve carbon neutrality in the next 40 years. Oil consumption in the transportation sector could be decreased substantially through the promotion of electric vehicles that use green electricity generated from renewable energy; China is planning to increase the electric vehicle sales ratio to 40% in 2030 (MIIT, 2019) and is likely to set a deadline for ending sales of gasoline vehicles.

![Figure 2 Sectoral CO₂ emissions from 2000-2050. Data were derived from (He et al., 2020); (Han et al., 2020); (Han et al., 2021). Emissions from 2000-2020 are based on the ensemble means](image-url)
from China Emission Accounts and Datasets (CEADs) (Guan et al., 2018), the Multi-resolution Emission Inventory for China (MEIC) model (Zheng et al., 2018), the Peking University (PKU) emission inventories (Tao et al., 2018), the Emission Database for Global Atmospheric Research (EDGAR) (Janssens-Maenhout et al., 2019), and the International Energy Agency (IEA) (IEA, 2018), adjusted as in (MEE, 2018). Data from 2020-2030 were simulated with a 3% fluctuation (Han et al., 2020) and a peak of no more than 106 Mt in 2025, from (He et al., 2020; Lin et al., 2019; Wang and Yan, 2021). Emissions from 2030-2050 are assumed to decrease linearly at a rate of 8%, from (He et al., 2020; State Grid, 2020).

2.1.2 Industrial process emissions and green-hydrogen-based energy

Emissions from industrial processes accounted for 7.2% of total anthropogenic emissions, with cement production contributing more than 70% (Liu, Z. et al., 2015; Shan et al., 2018). Industrial process emissions from cement production were 643 Mt CO$_2$ in 2016 and peaked in 2014 (Liu, Z. et al., 2015; Shan et al., 2019). It should also be noted that cement carbonation during weathering resquesters CO$_2$, which offsets 43% of the previous emissions on a 100-year scale (Xi et al., 2016). Carbon sequestration can be increased by increasing the amount of exposed surface areas, increasing the size and number of pores in cement, and using other alkaline minerals, such as blast slag and lime (Johannesson and Utgenannt, 2001; Xi et al., 2016).

Hydrogen is the most abundant chemical element. It is a promising secondary energy source that can provide a clean source of fuel and heat for industry, transportation and buildings (IEA, 2019; UK government, 2020). Green hydrogen, which is generated through the electrolysis of water powered by solar or wind energy, can serve as a reducing agent for metal smelting (Homann, 2019), which is critical for industries that have difficulty reducing CO$_2$ emissions (van Renssen, 2020). The use of electric vehicles and hydrogen-based transportation would substantially decrease CO$_2$ (7-10% of the total China emissions, Fig. 2) and pollutant emissions. More than a dozen countries have set timetables for banning the sale of fossil fuel-powered vehicles; ambitious countries, such as Norway, the Netherlands and Sweden, have set deadlines
that fall within 2025-2030 (Netherland Government, 2019; Norway Government, 2017; Sweden Government, 2019). Moreover, hydrogen energy storage provides an important guarantee for the flexibility of power systems (Ould Amrouche et al., 2016; Yu, D. et al., 2019). Developing the hydrogen economy is named as a priority in the EU’s post-COVID-19 green recovery package (European Commission, 2020). However, safety must be considered a top priority, and there have several accidents related to hydrogen power; for example, an explosion at OneH2 Hydrogen and Gas Company in Longview, North Carolina, USA, caused significant damage to surrounding buildings (Fernsby, 2020).

China is the largest iron and steel producer in the world. In April 2019, the Ministry of Ecological Environment issued Opinions on Promoting the Implementation of Ultralow Emissions in the Iron and Steel Industry (MEE, 2019), which focused on atmospheric pollutants (PM, SO$_2$, NO$_x$). The control measures for these pollutants are also suitable for decreasing CO$_2$ emissions, including upgrading all production processes and controlling organized emissions, fugitive emissions and transportation emissions. In the future, China's steel industry will need to achieve breakthroughs in low-carbon smelting technology, such as hydrogen-based steel production, as well as control of carbon emissions throughout the entire steel production process (Homann, 2019; Li and Li, 2019).

### 2.2 Current status and future potential of renewable and nuclear energy

The implementation of renewable energy has increased substantially over the last two decades. Currently, renewable sources make up 42.0% of the installed capacity (800 GW in 2019, Fig. 3a) and contributed 32.7% of the power generated in China (CEC, 2020a; IEA, 2020; NBS, 2021a). Correspondingly, the annual reductions in CO$_2$ emissions due to renewable and nuclear energy reached 1500 Tg in 2019 (Fig. 3b) (CEC, 2020a). China continues to optimize the arrangement of wind and solar photovoltaic (PV) power generation facilities, promote the construction of wind and solar PV power bases in the "Three North" region, encourage the local use of
renewable energy and make full use of planned transmission channels to achieve cross-regional delivery. The National Energy Administration requires that newly constructed renewable energy projects be connected to the grid on schedule (NDRC and NEA, 2017). By 2030, the proportion of the total national power generation that is nonfossil energy power will reach 50%. Nonfossil fuel energy will account for more than 85% of the total primary energy consumption by the 2050s (Fig. 3c) (He et al., 2020; Jiang et al., 2018; State Grid, 2020).
2.2.1 Hydropower

China has the world’s largest hydropower installed capacity. Its total installed capacity has increased from 130 million kilowatts (130 GW) in 2010 to 358 GW in 2019 (Fig. 3 a) (CEC, 2020b), with the corresponding CO₂ reductions increasing from 11 to 719 Mt (Fig. 3 b). The technically exploitable hydropower potential is 540 GW, and the economically exploitable hydropower potential is 402 GW; thus, there is still the potential for large-scale hydropower development (Huang and Yan, 2009; Zhang et al., 2017). At present, hydropower development in Tibet is minimal, making up only ~1% of the technically available resources. Proposals in the 14th Five-Year Plan and Vision 2035 recommend “the implementation of hydropower development in the lower reaches of the Yarlung Zangbo River” (Xinhua News Agency, 2020), which would provide nearly 300 billion kWh of near-zero carbon electricity per year. Moreover, overall hydropower development in the Yarlung Zangbo, Nujiang, Jinsha, Yalong, Dadu and Lancang Rivers is still limited (Li, X.-z. et al., 2018). Future hydropower
development efforts should balance political, economic and environmental challenges, including human resettlement, ecological environmental protection, earthquake resilience, and national security (Li, X.-z. et al., 2018; Zhang et al., 2017).

2.2.2 Wind

A wind turbine can generally function for 20 years, but it will offset its lifetime CO\(_2\) emissions within six months of operation, thus providing green electricity. The wind installed capacity in China increased from 8.4 GW in 2008 to 210.1 GW in 2019 (Fig. 3 a), making up 10.5% of the national total installed capacity in 2019. The CO\(_2\) emission reductions resulting from wind power increased from 11 to 311 Mt CO\(_2\) from 2006-2019 (Fig. 3 b) (CEC, 2020a). There is great potential for wind energy in China (610 GW), and the areas with the greatest potential are distributed mainly on the northwestern North China Plain and in the East China Sea (Chen, 2011; Zhang et al., 2017; Zhang et al., 2019). The wind power abandonment rate decreased from 17% in 2016 to 4% in 2019 and decreased most notably in Xinjiang and Gansu (from 40% to 10%) due to improvements in the spatial distribution of power and increased utilization within and across regions. Moreover, wind turbine technology has improved substantially in recent years, with substantial increases in single-machine capacity, and the predictability of wind power has also improved, achieving an average root mean squared error (RMSE) of 0.5-2 m/s (Liu, H. et al., 2018; Liu, H. et al., 2015; Zhao, J. et al., 2019). A new trend in recent years is the rapid development of offshore wind farms, which have three times the power generation potential of land-based wind farms (Chen, 2011; Liu et al., 2013; Yang et al., 2017; Zhang et al., 2019; Zhao and Ren, 2015). For example, pilot projects have been set up in Fujian, Shanghai and Jiangsu (Chen, 2011; Zhang et al., 2019). The implementation of offshore wind farms entails high risks due to the underdeveloped technology and high construction and maintenance costs. However, they have the advantages of more stable, higher wind speeds; less wind shedding; a higher single-machine installed capacity; shorter transport distances for the electrical load; and smaller land area
requirements. Thus, they show great promise and potential for the coming years (Chen, 2011; Yang et al., 2017; Zhang et al., 2019). The UK proposed a “Ten Point Plan for a Green Industrial Revolution” in November 2020, and advancing offshore wind was the first proposed measure; the proposal is to quadruple the offshore wind capacity by 2030 with the goal of producing 40 GW of offshore wind power (UK government, 2020). Future development will include improving the core components of wind turbines, continuously decreasing wind power costs, and making technological progress toward low-wind power generation (NDRC and NEA, 2017). Harnessing land-based wind power throughout the "Three North" region will require enhancing the flexibility of power systems and exploring cross-provincial power transmission. For eastern offshore wind power, it will be necessary to focus on research and development related to large-capacity wind turbines that are adapted to the offshore environment, and large-scale projects will be needed to decrease wind power costs.

2.2.3 Solar photovoltaic (PV) power

Solar PV power is one of the most abundant, widely available renewable energy sources and is projected to become the lowest-cost and largest power supply in China (He and Kammen, 2016; Zhang and He, 2013; Zhao et al., 2013). The installation of 1 kW solar PV can reduce emissions by ~2000 kg CO₂ yr⁻¹ compared to the emissions from fossil fuel power generation. The solar PV installed capacity in China increased from 0.03 GW in 2008 to 202.4 GW in 2019 (Fig. 3 a), accounting for 10.1% of the national total installed capacity. The CO₂ emission reductions resulting from PV power increased from 11 to 204 Mt CO₂ from 2006-2019 (Fig. 3 b) (CEC, 2020a). In the first three quarters of 2020, the newly installed PV capacity in China was 18.7 GW; of this, PV power plants generated 10.0 GW, and distributed PV panels generated 8.7 GW. In the short term, competitive processes have pushed developers to focus on larger solar PV projects to benefit from economies of scale and achieve lower bids. In the long run, compared with large power stations, distributed PV
provides lower single-unit capacity, positions the power source closer to the power user, and has flexible modes of power generation; thus, distributed PV has great potential for reducing carbon emissions. The International Energy Agency (IEA) has predicted that by 2025, solar energy will account for nearly 60% of clean energy production. During this period, PV utility costs will drop by 36% (IEA, 2020).

A simulation study showed that if large-scale PV panels and wind turbines were deployed in the Sahara, rainfall in the area would increase more than twofold due to increased surface friction and reduced albedo; the vegetation coverage would thus increase by 20%, benefiting both energy production and the environment (Li, Y. et al., 2018). However, plans to generate power in the desert should consider energy transmission methods, long-distance power loss, the high cost, and financing. The world's largest centralized PV power station in a desert was built in Dalate Banner, Inner Mongolia. It has a total capacity of 100 MW and generates up to 2 billion kWh per year. The National Energy Administration (NEA) proposed that solar PV power generation capacity should reach the equivalent of 5 billion tons of standard coal combustion heat by 2030. By 2050, nonfossil energy consumption is projected to account for more than 50% of the total energy consumption, and the PV installed capacity is projected to reach 1 billion kW (Chen and Zheng, 2018).

China’s electricity loads are concentrated in eastern and southern China, while the solar resource-rich regions are located in Northwest and North China (e.g., Inner Mongolia, Xinjiang and Gansu), far from the regions that consume the greatest electrical power load (He and Kammen, 2016; Zhang and He, 2013). This mismatch will require an efficient, flexible, and robust national electrical grid to be developed in the future.

2.2.4 Nuclear

Nuclear power is an important component of low-carbon energy (IEA and NEA, 2015; WNA, 2020). China's operating nuclear power installed capacity was 48.8 GW in 2019 (Fig. 3a), ranking third in the world (WNA, 2020), and its construction of
installed capacity ranked first in the world. It is expected that by approximately 2035, China's nuclear power installed capacity that is in operation will reach 200 GW. In the "14th Five-Year Plan" and in the medium- and long-term future development periods, 6-8 large-scale nuclear power facilities are projected to be built and put into operation each year to meet the average annual need for nuclear energy. The CO₂ emissions reductions due to nuclear power have increased to 247 Mt CO₂ (Fig. 3 b). Currently, nuclear power technology has progressed from the second generation to the third, and the third-generation “Hualong One” and “Guohe One” reactors, "high-temperature, air-cooled" reactors and other new technologies are in continuous development (Cao et al., 2016; Zhang, 2020). The European Union, UK (UK government, 2020) and the United States (US government, 2019) are also planning to vigorously develop nuclear power. Ensuring the safety of nuclear power is the basis for nuclear development, and it is necessary to continuously improve the safety of nuclear power facilities that are in operation and under construction. The accidents in Chernobyl (Beaugelin-Seiller et al., 2020; WHO, 2011) and at the Fukushima Daiichi (Povinec et al., 2013a; Povinec et al., 2013b) nuclear power plant caused serious environmental pollution and public alarm (Huang et al., 2013; Zhu et al., 2020). Since then, the policies for using nuclear energy to generate electricity have been more cautious than those for other forms of renewable energy that are more publicly acceptable (Sovacool et al., 2020). These obstacles to the further development of nuclear power in China will require effective communication strategies (Huang et al., 2013; Zhu et al., 2020) and cooperation with other countries (Cao et al., 2016) to overcome.

3 Increasing carbon uptake

3.1 Increasing the capacity of terrestrial carbon sinks

3.1.1 Forests, shrublands and grasslands

Carbon neutrality is determined by both emissions and uptake, and terrestrial carbon sinks playing a significant role in carbon uptake (Fang et al., 2018; Friedlingstein et
China’s terrestrial carbon sinks are estimated at 0.2-0.3 Pg C yr\(^{-1}\), offsetting ~10% of its fossil fuel emissions (Fig. 4a), and forest, shrubland, and cropland soils contribute the majority of the sink capacity (Fang et al., 2018; Huang and Sun, 2006; Liu, S. et al., 2018; Lu et al., 2018; Piao et al., 2009; Shi and Han, 2014; Yu et al., 2012). The total C sink capacity in national ecological restoration project regions was 132 Tg C yr\(^{-1}\), and 56% is attributed to the implementation of projects (Lu et al., 2018). However, a recent study using the atmospheric inversion method showed that previous estimates have been greatly underestimated and that the actual sink capacity could be 1.1 Pg C yr\(^{-1}\) (Fig. 4a), which would offset 45% of fossil fuel emissions (Wang, Jing et al., 2020). These substantial uncertainties will require further studies using multiple updated data sets (space-air-ground-based measurements and inventories) and ensembles of inverse models, such as the GCP (Friedlingstein et al., 2020) and TRANSCOM (Peylin et al., 2013) models and the OCO-2 MIP (Crowell et al., 2019). Moreover, as ecosystems (mainly forests) mature, their carbon sink capacity decreases (Fig. 4b). Xu et al. (2010) showed that the current forest carbon sequestration rate will decrease from 152.3 Tg C yr\(^{-1}\) in the 2000s to 39.5 Tg C yr\(^{-1}\) in the 2050s, which is only 25% of that in the 2000s; the sequestration rate of new forests is projected to increase from 30.4 to 65.9 Tg C yr\(^{-1}\) in the corresponding period, resulting in a decline in the total forest carbon sink capacity to half of that in the 2000s (Fig. 4b). Similarly, using stage-classified matrix models, Hu et al. (2015) projected a decrease in the carbon sink capacity of China’s forests from 92.0 to 75.5 Tg C yr\(^{-1}\) between 2020 and 2050. However, these projections do not include the recent “Large-scale Land Greening Operations” policies (NGC and NFGA, 2018) or President Xi’s commitments and measures at the 75th session of the United Nations General Assembly (Xi, 2020b) and the Climate Ambition Summit (Xi, 2020a), which will certainly greatly increase the terrestrial carbon sink capacity in China, and need to be accurately assessed in future studies.
Figure 4. Historical terrestrial carbon sink capacity from 1980-2018 (a) and projected sink capacity to 2050 (b). Data were derived from Fang et al., (2018) (Fang et al., 2018), Piao et al., (2009) (Piao et al., 2009), Pan et al., (2011) (Pan et al., 2011), Hu et al., (2015) (Hu et al., 2015); Xu et al., (2010) (Xu et al., 2010); Yu et al., (2013) (Yu et al., 2013); Wang et al., (2020) (Wang, Jing et al., 2020). Blue line and shaded areas indicate TRENDY ensemble mean and ±1 standard deviation, while grey lines indicate individual model results (Friedlingstein et al., 2020).

Anthropogenic management plays an important role in increasing carbon sink capacity. There is a growing recognition that wood harvesting and utilization are important carbon sinks (Ellison et al., 2011; Lippke et al., 2011; Ruddell et al., 2007; Scholz and Hasse, 2008; Zeng, 2008). Zeng (2008) (Zeng, 2008) proposed a carbon sequestration strategy in which certain trees are harvested and buried in the soil or stored in aboveground shelters. The anaerobic conditions under the thick soil layer
would prevent the buried wood from decomposing. Because a large flux of CO$_2$ is constantly being assimilated into China's forests through photosynthesis, wood harvest and burial/storage represent an inexpensive yet effective carbon sink. The surface carbonization of such wood can provide a long-term carbon sink. Buried wood can also prevent the release of massive amounts of carbon due to wildfires that occur during El Niño periods, such as the wildfires in Australia (Wang, J. et al., 2020), the USA, and tropical forests (Liu, J.J. et al., 2017).

Unlike forests, grasslands were mainly carbon neutral from the 1980s to the 2000s (Fang et al., 2018). Grasslands in China cover an area of 331 Mha, and their total carbon storage is ~25.4-29.1 Pg C, 94.6%-96.6% of which is stored in SOC (Fang et al., 2010; Fang et al., 2018; Tang et al., 2018). Specifically, the aboveground biomass of grasslands increased at a low rate of 1.0 Tg C yr$^{-1}$ from 1982-1999 (Piao et al., 2007). Grassland SOC has not been shown to exhibit significant changes based on either statistical models (Liu, S.S. et al., 2018; Piao et al., 2009) or in situ measurements (Yang et al., 2010); however, these conclusions are subject to notable uncertainty due to methodological differences and a lack of extensive long-term observations (Fang et al., 2010; Huang et al., 2010). The Tibetan Plateau grassland represents a carbon sink of 10-30 Tg C yr$^{-1}$ (Lin et al., 2017; Piao et al., 2012; Zhuang et al., 2010), which is projected to decrease from 2010 to the 2060s (Han et al., 2019).

3.1.2 Cropland soils

Increasing cropland soil organic carbon (SOC) by sequestering carbon in soils will play an important role in both maintaining food security and mitigating climate change; moreover, it is an inexpensive and safe method for long-term carbon sequestration (Han et al., 2016; Lal, 2004; Pan et al., 2009; Yu et al., 2012). Covering an area of 130 Mha, China’s cropland soils are mainly carbon sinks due to good management practices such as increasing carbon inputs through straw return (Han et al., 2016; Wang et al., 2016) The cropland carbon sequestration rate increased from 10 to 30 Tg C yr$^{-1}$ from the 1980s to the 2010s (Huang and Sun, 2006; Huang et al., 2010;
Yu et al., 2012). In addition, increasing C inputs from 2.2 to 4.0 Mg C ha$^{-1}$ yr$^{-1}$ from the 2010s to 2040s by enhancing straw return and organic matter incorporation (especially manure application) will increase the sequestration rate for cropland soils from 30 Tg C yr$^{-1}$ in the 2010s to 90 Tg C yr$^{-1}$ in the 2040s (Fig. 4b) (Yu et al., 2013). However, the overall balance between SOC increases and CH$_4$ and N$_2$O emissions needs to be carefully handled (Zhang et al., 2014). In addition, the SOC losses in Northeast China need to be reversed by increasing carbon inputs (Han et al., 2016; Huang and Sun, 2006; Yu et al., 2012). Promoting organic farming and no-tillage practices can also increase SOC and reduce CH$_4$ emissions (Foley et al., 2011; Gattinger et al., 2012; Powlson et al., 2014).

3.2 Oceans

The ocean carbon sink is also called the “blue carbon sink”, in contrast to the terrestrial “green carbon sink” (Nellemann et al., 2009). The known marine carbon sink mechanisms are the "biological pump" (BP) (Longhurst, 1991; Raven and Falkowski, 1999) and the "solubility pump" (Siegenthaler and Sarmiento, 1993). Plankton produce large amounts of organic matter that sustains the food chain. The extra carbon accumulates on the sea bed at geological timescales; this process constitutes the BP (Raven and Falkowski, 1999; Siegenthaler and Sarmiento, 1993). The marine microbial carbon pump (MCP) represents a new concept for increasing marine carbon sinks (Jiao et al., 2010; Jiao et al., 2020; Jiao et al., 2018). The MCP theory notes that marine microbial ecological processes convert active dissolved organic carbon into inert dissolved organic carbon that can be stored in the ocean for up to 5000 years and does not lead to ocean acidification.

The total marine carbon sink capacity in China could reach 100 Tg C yr$^{-1}$, and this rate could be doubled by developing ocean negative carbon emission (ONCE) strategies (Jiao, 2021). ONCE strategies include: 1) Ecological nutrient regulation-based negative emission projects that link land and sea and maximize BP and MCP by reducing nitrogen and phosphorus inputs from the land (Jiao et al., 2014);
2) implementing negative emission technology (NET) in anaerobic/acidified sea areas by applying minerals and increasing the sea pH (Jiao et al., 2020); and 3) comprehensive negative emissions projects in marine aquaculture areas, e.g., artificial upflows driven by clean energy sources that bring nutrient-rich water from the bottom of the cultured sea area to the upper water layer (Jiao, 2021). The carbon sink capacity along coastlines is relatively small, but it is much larger in the open seas. The sedimentary organic carbon sink capacity of the sea along the coast of China is 20 Tg C yr\(^{-1}\), and the East and South China Seas deliver organic carbon to the adjacent oceans at rates of 15-37 and 43 Tg C yr\(^{-1}\), respectively (Jiao, 2021; Jiao et al., 2018). The sink capacity along the coast, which includes mangroves, seagrass beds, salt marshes and other marine ecosystems, is \(\sim 1\) Tg C yr\(^{-1}\). Measurements from the edges of mangroves along the coast of the Leizhou Peninsula and at Jiulongjiang Estuary showed that the mangrove sediment C sequestration rate is 1.1-4.1 t C ha\(^{-1}\) yr\(^{-1}\) at 40-100-year time scales (Alongi et al., 2005; Yang et al., 2014). The amount of carbon sequestered by cultured shellfish and algae in China is \(\sim 3\) Mt C yr\(^{-1}\), and harvest removes at least 1.2 Mt yr\(^{-1}\). Among them, carbon removal by shell is approximately 0.7 Mt and represents a relatively long-term carbon sink (Yue and Wang, 2012; Zhang et al., 2005).

### 3.3 Carbon capture, utilization and storage (CCS or CCUS) and negative emissions technology (NET)

Carbon capture, utilization and storage (CCS/CCUS) is a vital and potentially effective technology to substantially offset CO\(_2\) emissions from the use of fossil fuels (Cai et al., 2020; Jiang and Ashworth, 2020; Jiang et al., 2020). There is an urgent need for research and development to deliver cost-effective CCUS technologies for the capture, conversion, utilization, transportation, and storage of CO\(_2\) (Jiang and Ashworth, 2020; Yu, S. et al., 2019).

Various CCUS technologies have been implemented in China, such as CO\(_2\) storage in deep saline aquifers, CO\(_2\)-enhanced oil recovery (CO\(_2\)-EOR), and coalbed methane
displacement by CO$_2$ (Cai et al., 2020; Liu, H.J. et al., 2017). By 2019, nine capture demonstration projects and twelve geological utilization and storage projects had been implemented. The total amount of geologically stored CO$_2$ in all CCUS projects in China is ~1 Mt, which is much smaller than the 21 Mt yr$^{-1}$ stored in the USA. Compared to the total CO$_2$ emissions (10 Pg), the current captured amount is almost negligible, though it is optimistically projected to reach 880 Mt in 2050 in the 1.5°C scenario (He et al., 2020).

The greatest obstacle for the use of CCUS is its high economic cost, and it is also the most expensive method of all NETs (Liu, H.J. et al., 2017; Palmer, 2019). Capture alone is the most energy-consuming and expensive part of the entire CCUS process, which includes capture, transportation, utilization, and storage. The current capture cost for low-concentration CO$_2$ is 300-900 yuan RMB t$^{-1}$, and the transportation cost is 0.9-1.4 yuan RMB t$^{-1}$ km$^{-1}$ by tanker (Cai et al., 2020) or 80-90 yuan RMB t$^{-1}$ km$^{-1}$ by pipeline (Jiang et al., 2020). The cost of CO$_2$-EOR varies widely depending on the technical details (Cai et al., 2020; Liu, H.J. et al., 2017). The crude oil price, at 70 USD per barrel, can balance the extra cost of CCUS. In several scenario studies, CCUS has been projected to absorb 0.24 Pg C in 2050 (He et al., 2020; Yu, S. et al., 2019); this is the same size as the forest and cropland soil sinks and represents a major NET for achieving C neutrality by 2060 (He et al., 2020; Piao et al., 2009).

There is a large potential market for CCUS in China. For example, EOR could be applied to 13 billion tons of crude oil, which could sequester 4.7-5.5 billion tons of CO$_2$. Currently, EOR projects are in progress at the Jilin, Daqing, and Zhongyuan oilfields and have a capacity of 0.20-0.35 Mt yr$^{-1}$ (Cai et al., 2020). However, CCUS is in its initial stages of development and faces challenges due to its high costs and technological, environmental and policy problems (Jiang et al., 2020; Liu, H.J. et al., 2017). The captured CO$_2$ is easily utilized and then rereleased into the atmosphere through physical, chemical and biological uses, e.g., beer, other carbonated beverages, food, and chemical compounds. Moreover, geological storage risks the leakage of the stored CO$_2$ and requires monitoring throughout the whole process (Jiang and Ashworth, 2020; Lui and Leamon, 2014).
4 Reductions in non-CO₂ GHG (CH₄, N₂O and F-gases) emissions

Non-CO₂ GHGs (CH₄, N₂O and F-gases) (non-CO₂) have global warming potentials (GWPs) that range from ten to twenty thousand times that of CO₂ (IPCC AR5, 2013; MEE, 2018). Non-CO₂ accounted for 27.5% of the global total CO₂ eq. emissions in 2018 (Olivier and Peters, 2020). The potential for non-CO₂ emission reduction is significant and will reach 27% of the total non-CO₂ by 2030.

At present, China's non-CO₂ emissions account for 18.4% (2 Pg CO₂ eq.) of its total GHG emissions, and the increasing trend in non-CO₂ emissions is obvious (Fig. 5a) (MEE, 2018). Non-CO₂ emissions come mainly from the energy, agriculture and industrial sectors (Fig. 5b). Non-CO₂ emissions cannot be ignored in the process of achieving peak carbon emissions before 2030 and carbon neutrality before 2060. The implementation of cost-effective emission reduction measures and economically viable technologies could bring about an early peak and reduce the peak magnitude by 27.6% using a 100-year GWP. By 2050, China has the potential to achieve a 47% reduction in non-CO₂ emissions compared to the reference scenario (Lin et al., 2019), and sectoral reduction measures should be proposed based on their respective emissions characteristics.
4.1 CH$_4$

CH$_4$ is a potent GHG with a warming potential that is 21-fold higher than that of CO$_2$ considering a 100-year GWP (IPCC AR5, 2013). Global methane emissions are ~570 Mt per year, and anthropogenic emissions contribute 60% of the total (Saunois et al., 2020). CH$_4$ makes up the largest share (67.9%) of global non-CO$_2$ emissions and comes mainly from the production and transmission of coal, oil and natural gas in the energy industry; rice cultivation and livestock enteric fermentation and manure management in agriculture; and landfills and sewage treatment in waste management (Fig. 5b) (Olivier and Peters, 2020; Saunois et al., 2020). The potential to reduce CH$_4$ emissions is very large, accounting for 76% of the overall non-CO$_2$ reduction potential (EPA, 2019).

China's CH$_4$ emissions account for 10.4% of its total GHG emissions and 55.5% of its non-CO$_2$ emissions (Fig. 5b), and anthropogenic CH$_4$ emissions have increased rapidly in recent years (Lin et al., 2020; MEE, 2018). Reducing CH$_4$ emissions is a cost-effective method of progressing toward carbon neutrality. By 2050, CH$_4$
emissions are projected to increase by 46% in the business-as-usual scenario but decrease by 41% in the 1.5°C scenario (Fig. 5c) (He et al., 2020).

4.1.1 Fossil fuel emission reductions

Energy activities contributed 45% of the total anthropogenic methane emissions, including those from coal, oil and gas production and transportation (MEE, 2018). Coal is the primary energy source in China, making up 57% of the total energy production (NBS, 2020). The national average utilization rate for coal mine methane is 38%, with higher rates (60-70%) in Chongqing and Shanxi provinces (Sheng et al., 2019). The potential for methane emission reduction in the coal mining sector is high, mainly due to the expected decline in coal demand and the maturity of emission reduction technologies. Increasing the utilization rate of coal mine methane costs nothing and can therefore be converted into income; thus, this strategy should be given higher priority. CO₂ coalbed methane recovery is in the pilot stage and is being demonstrated by China United Coalbed Methane in Qinshui-Linfen Basin (Cai et al., 2020).

Moreover, due to the high uncertainty in methane emission factors, it is very important to strengthen methane emission monitoring and accounting. Global anthropogenic fossil fuel methane emissions were estimated to be 25-40% higher than those determined in previous studies using $^{14}$CH$_4$ (Dyonisius et al., 2020; Hmiel et al., 2020). After 5 years of field monitoring, oil CH$_4$ emissions were found to be 60% higher than US Environmental Protection Agency (EPA) estimates (Alvarez et al., 2018). Coal-related methane emissions have been underestimated by 25% and 21% when abandoned mines and the exploitation depth are not taken into account (Kholod et al., 2020). Therefore, it is necessary to improve disclosures of coal mining depth as well as data on abandoned coal mines to more accurately compile CH$_4$ emissions inventories and predict future CH$_4$ emissions (Lin et al., 2020).
4.1.2 Agriculture emission reductions

Agricultural CH₄ emissions come mainly from rice cultivation and livestock enteric fermentation and manure management. Emissions from rice cultivation were estimated to be 8.9 Tg yr⁻¹ in 2014 (MEE, 2018), with hot spots in southern China due to the double-rice cropping system. Under warming conditions, the CH₄MOD model projected that the CH₄ flux would increase by 14% by approximately 2050 (Zhang et al., 2011). Modern rice breeding to reduce CH₄ emissions from rice cultivation could reduce annual emissions by 7.1% (Jiang et al., 2017). The adoption and promotion of midseason drainage and intermittent irrigation rather than continuous flooding can decrease emissions, especially in Southwest China; moreover, the use of composted straw and film mulching in cultivation also substantially decreases emissions and promotes production (Yan et al., 2009; Yan and Xia, 2015). With regard to livestock emissions, enteric fermentation is responsible for 90% of emissions, while manure management contributes the remaining 10% (Lin et al., 2020; Lin et al., 2011). Emissions can be reduced by 20-40% with improved feeding techniques such as increasing the fraction of concentrates in roughage and adding inhibitors to livestock diets (Duthie et al., 2017; Patra, 2013). Emissions from manure management can be reduced by 17-26% through aerobic composting and biochar addition (Chen et al., 2017; Hou et al., 2017; Ma et al., 2018).

4.1.3 Waste emission reductions

Landfills and sewage treatment plants are important sources of waste methane emissions, each contributing 50% of the total waste emissions (Lin et al., 2020; MEE, 2018; Peng et al., 2016). The low-carbon emissions scenarios in China's landfills would reduce CH₄ emissions by 54% and protect nearly 10 million people from landfill odors (VOCs, etc.), providing a win-win situation for GHG emission reduction and environmental improvement (Cai et al., 2018). Experts have proposed nine key emission reduction technologies for landfills (Cai et al., 2018; Cai et al., 2015). The solid waste and wastewater sectors have higher emission reduction costs
than other sectors, but lower-cost emission reduction measures could reduce emissions by 20% (Du et al., 2018; Zhao, X. et al., 2019). National-scale initiatives to reduce food waste would also reduce CH₄ emissions by reducing emissions from landfills.

4.2 N₂O

N₂O emissions account for 30% of China's non-CO₂ emissions. N₂O comes mainly from agriculture, energy, industrial processes and waste (Fig. 5b) (MEE, 2018). Agriculture contributes 60% of anthropogenic N₂O emissions, and nitrogen (N) fertilizer use is the main emission source (Huang and Tang, 2010; Sun and Huang, 2012; Tian et al., 2020). An increase in demand for food and animal feed will further increase N₂O emissions. Relevant emission reduction measures include reducing the amount of N fertilizer used; increasing crop N use efficiency (NUE); performing precision fertilization; applying controlled fertilizers, inhibitors, and biochar; and implementing organic agriculture (Gu et al., 2015; Sun and Huang, 2012; Tian et al., 2020; Xia et al., 2017; Yan et al., 2013).

High priority should be given to reducing the use of N fertilizer and improving NUE. China's annual agricultural N fertilizer use reached 1.9×10⁷ tons in 2019 and has decreased at a rate of -4% over the last five years (NBS, 2021b). The NUE of China's cropland is 30%-40% (Gu et al., 2015; Huang and Tang, 2010), which is much lower than that in European countries and the United States (50-60%). When the N input exceeds the amount of N needed by crops, N₂O emissions increase exponentially (Hoben et al., 2011; Li and Ju, 2020; McSwiney and Robertson, 2005). Therefore, it is crucial to match the timing of N fertilizer application with critical periods of crop demand (e.g., the jointing stage). Moreover, implementing deep fertilizer application and increasing the number of fertilizer applications can reduce N₂O emissions by 14.6% and 5.4%, respectively (Xia et al., 2017). Various breeding improvements have also contributed greatly to increasing NUE (Liu et al., 2021). The use of suitable N fertilizer forms, e.g., nitrate fertilizer in drylands and urea in rice paddies, can greatly
reduce emissions (Li and Ju, 2020). A decrease of 30% in the amount of N fertilizer applied generally does not affect crop yields (Li et al., 2017). The promotion of soil testing and fertilizer recommendations (STFRs) and precision agriculture can effectively reduce N$_2$O emissions in agricultural land (Rees et al., 2020; Sun and Huang, 2012). Since the late 1990s, China has implemented an STFR program to reduce the overuse of synthetic N fertilizer on cereal crops, and N$_2$O emissions were reduced by 379 Gg N$_2$O in 2001-2008; if STFR practices were implemented across the country, the total emission reduction could reach 26%-41% (Sun and Huang, 2012).

The use of controlled N-release fertilizers, the addition of N inhibitors (urease inhibitors or nitrite inhibitors, e.g., dicyanide (DCD) and 3,4-dmethyl phosphate (DMPP)), the application of biochar, and the use of lime, dolomite powder and other amendments on acidic soils can reduce N$_2$O emissions by 28%-48% (Li et al., 2017; Xia et al., 2017; Yao et al., 2018). The use of biochar can reduce emissions by 10%-90%, and pH-improvement substances can reduce emissions by 40% (Cao et al., 2019).

4.3 F-gases

Although the proportion of F-gases (mainly HFCs, PFCs, SF$_6$ and NF$_3$) is smallest among all greenhouse gases (Fig. 5a and 5b), the GWP (Global Warming Potential) of F-gas reach could be higher than 20,000 (e.g. SF$_6$). As for the emissions sources, HFC-23 mainly comes from the fluorine industry, HFC-32 and HFC-125 from air conditioning, HFC-134a from automotive air conditioning, SF$_6$ from electric power industry, and PFCs from primary aluminum production or electronic industry (China City Greenhouse Gas Working Group, 2019; MEE, 2018), of which a proportion comes from refrigerants. F-gases emissions are growing rapidly in recent decades, and with great mitigation potential and significant benefits for climate change mitigation (Xu and Ramanathan, 2017). For example, China's existing HFC-23 disposal and destruction policy can significantly reduce fluorine emissions (NDRC, 2015). Strict
implementations of the Ozone Depleted Substances Regulations and the Montreal Protocol, emissions reduction targets can be further determined on the basis of the road map for emission reductions established by the Kigali Amendment. Moreover, the Green High Efficiency Refrigeration Action Plan, released in 2019, proposed "the development and use of air conditioning lines with low GWP refrigerants and limit the use of HFCs" (NDRC, 2019). Therefore, it is highly necessary to strictly control the leakage and discharge of refrigerants in the production process, strengthen the recovery and disposal of refrigerants, and develop refrigerants with low GWPs.

5 Suggestions for policy makers

5.1 Top-level design and R&D of new technologies for reducing emissions and increasing sink capacity; establishing pilot regions

A good top-level design, such as enhancing basic research on carbon neutrality and R&D of key technologies for reducing emissions and increasing sink capacity, should be the first priority. The dynamic assessment and adjustment of policies based on comprehensive scientific data and new technological advances are also needed. In addition, more attention should be paid to reductions in non-CO₂ emissions. CH₄, N₂O and F-gases are GHGs with stronger GWPs than CO₂ and have more potential for reduction than CO₂ but have not yet been well explored. It is also necessary to strongly promote research and development for technologies including land carbon sinks, ocean carbon sinks, CCUS, etc., and to conduct comprehensive evaluations of the technological and economic feasibility of these technologies. In addition, it is important to establish carbon neutrality demonstration regions in provinces and cities with good economic, technological and resource conditions.

5.2 Construction of an integrated space-air-ground GHG monitoring and modeling system to support the dynamic assessment and adjustment of polices

Following the principle that data should be measurable, verifiable and reportable (MRV), integrated space-air-ground monitoring and modeling systems, such as those
piloted in the EU and North America in the last decade (e.g., in Paris and Washington (Baltimore)) and in recent years in Beijing-Tianjin-Hebei in China (Han et al., 2018), should be promoted. These systems include: 1) GHG satellites; continuous developments in space-based GHG monitoring equipment and inversion algorithms are required to take advantage of its large spatial coverage; 2) a ground-based monitoring network that includes both high-precision background stations and dense networks of low-cost, moderate-precision sensors in cities; 3) an air-based monitoring network that uses a variety of air-based tools including airplanes, tethered balloons, drones, and AirCores; and 4) a carbon emissions and assimilation simulation system that provides data on GHG sources and sinks at high spatial-temporal resolutions at various levels, from the national level to the provincial and city levels, and identifies illegal emissions for environmental law enforcement purposes.

The value of such a pilot system was demonstrated in a recent study showing that a 7-20% reduction in CO$_2$ emissions from the global scale to city scale caused by COVID-19 was detected in CO$_2$ concentrations (Zeng et al., 2020). The CO$_2$ emissions decrease rate for achieving carbon neutrality after 2030 is on the same scale, at 8-10%. These results indicate that an expanded, more developed system is needed to dynamically assess the process of achieving carbon neutrality.

5.3 Promotion of carbon exchange markets and green investments by governments and other entities

The national carbon trading market, for which pilot projects began in seven provinces and cities in China in 2011, covers eight sectors, including coal-fired electricity, iron and steel, and other heavy industries. In January 2021, the Ministry of Ecological Environment issued “Measures for the Management of Carbon Emissions Trading (Trial)” (MEE, 2021), which were implemented starting on 1 February 2021. The key GHG emissions enterprises were defined as those that 1) belong to an industry covered by the national carbon emissions trading market and 2) generate annual GHG emissions reaching 26,000 tons of CO$_2$ eq. These enterprises are required to prepare a
GHG emissions report for the previous year containing detailed emissions amounts and calculations and to provide the reports to the responsible authorities before 31 March of each year. Provincial ecological and environmental departments can then verify the reported emissions by employing the services of reputable technical companies and can penalize enterprises that disobey the regulations. These measures effectively regulate GHG emissions from key emission sources using economic means. Moreover, green investments financed by the government and by public companies are critical in terms of creating market signals towards clean energy and promoting the development of sink technologies, especially early-stage technologies that are high-risk but promising (e.g., CCUS and hydrogen) (Mazzucato and Semieniuk, 2018).

5.4 Involving the public

Popular science advocacy should be strengthened, and the public should become fully engaged in reducing emissions and increasing sink capacity by, e.g., reducing power usage in everyday life and participating in large-scale greening operations by planting trees. In recent years, members of the public have also actively participated in nationwide campaigns to reduce food waste. To reduce agricultural non-CO2 emissions, the government could provide subsidies for additives and inhibitors to promote yield production and reduce GHG emissions. Experts could be employed to provide emission reduction training through multiple means, e.g., on-site or online training, promotional films, leaflets, and other approaches. The public can play an important role in achieving carbon neutrality in China.

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