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INTEGRER L'AGRICULTURE DANS LES POLITIQUES D'ATTENUATION CHINOISES

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INTEGRATING AGRICULTURE INTO THE CHINESE MITIGATION POLICIES

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Abstract in English

This thesis provides an integrated evaluation of the overall technical and economic mitigation potential in Chinese agriculture and the conditions of putting a carbon price in this sector. The research scope is cropland emissions and particularly those related to synthetic nitrogen fertilizer use. The thesis is articulated around the construction of a bottom-up marginal abatement cost curve (MACC) which offers a rational framework for combining biophysical and economic data to reflect mitigation costs and potentials. This tool allows the aggregation of mitigation potentials arising from the application of a subset of cost-effective measures above a notional baseline level. An analysis of Chinese climate policies reveals that agriculture is nearly absent in the current national mitigation strategy. We therefore intend to assess the technical, economic and political feasibility of integrating agriculture into domestic mitigation policies. In the first place, the emissions trends and calculation methods are assessed to determine a rigorous approach to build baseline scenarios from projected business-as-usual activities to 2020. Secondly, we identify nine cropland mitigation measures and evaluate their abatement rates and future applicability beyond the baseline scenario to conclude a total feasible technical mitigation potential. The economic potential of each scenario is then compared by using estimated implementation costs of different mitigation measures relative to conventional farming practices. The MACC results show that agriculture provides significant mitigation potentials and is able to offset about one-third of the baseline emissions. In addition, realisation of one-third of this mitigation potential is cost-negative for farmers. We finally examine the conditions of using economic instruments to reduce emissions at the lowest cost for the agricultural sector. Given the institutional, behavioural and social obstacles, we strongly suggest restructuring the current fertilizer subsidy regime to send a clear political signal from central planning. Scaling-up offset projects using carbon intensity as the standardized baseline is recommended and could pave the way for an experimental emission trading scheme in agriculture. In light of China’s strong concern on safeguarding its food security, case studies on regional cereal production are carried out in all steps, including the analysis of provincial greenhouse gas intensity of production, regional abatement potential related to synthetic nitrogen fertilizer use as well as disparities of implementation cost among certain regions.

Key words: Chinese agriculture, marginal abatement cost curve, cost effectiveness, mitigation potential, pricing carbon, Chinese climate policy

Résumé en français

Cette thèse offre une évaluation du potentiel d'atténuation technique et économique global dans l'agriculture Chinoise et des conditions nécessaires à la formation d'un prix du carbone dans ce secteur. Le champ de recherche inclut les émissions venant de terres cultivées et en particulier celles liées à l'utilisation d'engrais azotés synthétiques. Il s'articule autour de la construction d'une courbe de coûts marginaux de réduction (MACC), qui offre un cadre rationnel pour combiner les données biophysiques et économiques afin de refléter les coûts et les potentiels d'atténuation. Cet outil permet d'agréger le potentiel d'atténuation découlant de l'application d'un sous-ensemble de mesures coût-efficacité en dessus d'un niveau de référence désignée. Une analyse des politiques climatiques chinoises révèle que l'agriculture est presque absente de la stratégie nationale d'atténuation. Nous avons donc l'intention d'examiner la faisabilité du point de vue technique, économique et politique, d'intégrer l'agriculture dans les politiques domestiques d'atténuation. En premier lieu, la tendance et les méthodes de calcul des émissions sont évaluées afin de déterminer une approche rigoureuse permettant de construire des scénarios de référence à partir de prévisions des activités 'business-as-usual' pour 2020. Deuxièmement, nous identifions neuf mesures d'atténuation des sols cultivés, nous évaluons leur taux d'abattement et leur applicabilité future au-delà du scénario de base pour obtenir un potentiel total d'atténuation techniquement faisable. Leur traduction en potentiel économique est alors faite en comparant les coûts de mise en œuvre des différentes options d'atténuation relatives aux pratiques agricoles conventionnelles. Les résultats des MACC montrent que l'agriculture offre un potentiel d'atténuation important, qui pourrait compenser environ un tiers des émissions de référence et dont un tiers pourrait être réalisé au coût négatif pour les agriculteurs. Nous examinons enfin l'utilisation des instruments économiques pour réduire les émissions au moindre coût dans le secteur agricole. Compte tenu des obstacles institutionnels, comportementaux et sociaux, nous suggérons fortement d'engager une réforme dans le système des subventions d'engrais afin d'envoyer un signal politique clair aux agriculteurs. L'utilisation de l'intensité du carbone comme référence normalisée est recommandé pour améliorer et élargir l'accès aux projets de compensation, et peut aussi préparer le terrain pour un possible programme expérimental d'échange de quotas d'émissions dans l'agriculture. En cohérence avec la priorité de protéger la sécurité alimentaire en Chine, des études de cas sur la production régionale de céréales sont introduites dans toutes ces étapes, y compris l'analyse de l'intensité des gaz à effet de serre de la production dans chaque province, le potentiel régional de réduction des émissions liées à l'utilisation d'engrais azotés ainsi que la disparité de coûts de mise en œuvre dans certaines régions.

Mots clés: agriculture chinoise, courbe de coûts marginaux de réduction, coût-efficacité, potentiel d'atténuation, tarification du carbone, politique climatique chinoise

Table of Contents

Abstract in English **5**

Résumé en français **6**

Introduction **13**

Chapter 1 - General overview of climate policies in China **25**

1. Chinese GHG emissions and drivers of CO_2 emissions growth **26**

1.1. GHG emissions according to national inventories 26

1.2. Historic trends of CO_2 emissions from fossil fuel combustion and cement production 27

1.3. Drivers of CO_2 emissions growth in China 29

2. From international engagement to national climate policies **31**

2.1. Framework of climate policy marking in China 31

2.2. China's engagement in international climate negotiations 31

2.3. Inclusion of climate targets into national development plans 33

3. Policies and actions to meet the mitigation targets **34**

3.1. Climate and energy related policies throughout 2011-2015 34

3.2. Energy intensity evolution in China 36

3.3. Scenarios of future CO_2 emissions in China 38

3.4. China on track to meet the 2015 and 2020 climate targets? 39

4. Conclusion **40**

Chapter 2 - Agriculture emissions: past trends and construction of future baseline scenario **43**

1. Chinese agriculture development and major policies **44**

1.1. Rapid agriculture development and food security priority 44

1.2. Current and near-future national agriculture policies 46

2. Accounting agricultural emissions and past trends **47**

2.1. Primary source of CH_4 and N_2O emissions 47

2.2. Origin of important CO_2 emissions 49

3. Study on GHG emissions from cropland **50**

3.1. General methodologies of accounting N_2O emissions from cropland 50

3.2. A case study of GHG intensity of cereal production 52

4. Building a baseline for future GHG emissions from Chinese agriculture 55

4.1. Projection of future agriculture activities 55

4.2. Projection of N consumption by crop production.. 56

4.3. Results of business as usual scenario emissions from agriculture 58

5. Conclusion 59

Chapter 3 - Technical mitigation potential in China's croplands.................... 61

1. Mitigation measures and abatement rates 62

1.1. Selection of mitigation measures.. 62

1.2. Estimation of abatement rates of mitigation measures 66

1.3. Treatment of measure interactions... 70

2. Which technical potential can be realized from Chinese croplands?....................... 72

2.1. Measure adoption additional to the baseline scenario 72

2.2. Maximum feasible mitigation potential from croplands.. 74

3. Case study: mitigation potential from cereal production in China 74

3.1. Current and historical GHGI of cereal production at the provincial level 75

3.2. Abatement rate of N fertilizer management at provincial level 80

3.3. Mitigation potential from best N management practices 81

4. Conclusion 84

Chapter 4 - Economic mitigation potential in Chinese croplands 87

1. MACC tool introduction and approach selection 88

1.1. Literature review of existing agriculture MACCs 88

1.2. Choice of methodological approach for MACC construction 90

2. Estimation of measure implementation cost 91

2.1. Evaluating benefits/costs implications of adopting mitigation measures 92

2.2. Forecast of future production input and input price... 94

2.3. Illustration of cost calculation of the measure organic addition to rice paddies 95

3. MACC results and abatement scenarios 96

3.1. MACC construction and results... 96

3.2. Abatement scenarios of emissions from China's agriculture... 98

3.3. Regional disparities in MACC results 100

4. Discussions and sensibility analysis 102

4.1. Comparisons with other MACC studies 102

4.2. Uncertainties and sensitivity analysis 105

4.3. Significances and limits of MACC studies 106

5. Conclusion 109

Chapter 5 - How can economic incentives trigger abatement in Chinese agriculture? 111

1. Pricing carbon in agriculture from theory to practice 112

1.1. Theoretical background on integrating environmental issues into the economy 112

1.2. States of carbon pricing around the world 113

1.3. States of carbon pricing in agriculture 114

2. States of carbon pricing in China 116

2.1. Development of carbon emission trading schemes in China 116

2.2. Regulatory conditions of domestic offset markets 121

2.3. Carbon tax discussions in China 122

2.4. Subsidies for synthetic fertilizers is a negative carbon tax 123

3. Difficulties and specificities of pricing carbon in China's agriculture 124

3.1. Diffuse nature of agriculture emissions and mitigation actions: the need for aggregator 125

3.2. Difficulties in accurate measurement and verification of emission and abatement 126

3.3. The specialty of soil carbon storage 128

4. Suggestions on pricing carbon in China's agriculture 129

4.1. Restructure fertilizer subsidies 129

4.2. Up-scaling carbon crediting schemes 130

4.3. ETS pilot covering the agricultural sector 132

5. Conclusion 134

General Conclusion 137

References 141

Annex 1: Selected values for estimating N_2O emissions from croplands 151

Annex 2: Detailed quantification methods of GHGI of cereal crops 153

Annex 3: Past and predicted future agriculture activities 154

Annex 4: Mitigation potential from the livestock sector 156

Annex 5: GHGI at regional level in 2010 and implications for mitigation strategies 163

Annex 6: Estimates of N_2O abatement rates of mitigation measures C1-C4 165

Annex 7: Considerations for measure implementation cost estimation 168

Annex 8: Comparative table of market design among pilots 169

Annex 9: VER rules and development of CCER projects.............................. 174

Annex 10: Discussions on levying carbon tax in China 177

List of Figures

Figure 0-1 Development process of a bottom-up MACC for Chinese agriculture (cropland emissions) 21

Figure 1-1 Composition of China's GHG emissions by gas in 2005(a) and 1994(b) 27

Figure 1-2 Composition of China's GHG emissions by sector in 2005(a) and 1994(b) 27

Figure 1-3 Evolution of China's CO_2 emissions and global share from 1980 to 2010 28

Figure 1-4 Growth trends of CO_2 emissions and energy use in China from 1990 to 2008 29

Figure 1-5 Drivers of CO_2 emissions growth in China from 1981 to 2010 30

Figure 1-6 World cumulative CO_2 emissions 1850 – 2005: total and per capita 32

Figure 1-7 Energy intensity evolution during the FYP periods from 1980 to 2010 37

Figure 1-8 China's position towards target reaching (2011-2020) 39

Figure 2-1Trend of crop production and N fertilizer inputs in China from 1960 to 2010 44

Figure 2-2 Trend of animal products in China from 1980 to 2009 45

Figure 2-3 Historical changes of national average GHGI of rice, wheat and maize production ... 54

Figure 2-4 Baseline GHG emissions from Chinese agriculture to 2020 59

Figure 3-1 GHGI of rice (a), wheat (b) and maize (c) production in different provinces in 2010. 75

Figure 3-2 Geographic pattern of rice (a), wheat (b) and maize (c) GHGI in 2010........ 77

Figure 3-3 Historical changes of provincial GHGI of rice (a), wheat (b) and maize (c) production 78

Figure 3-4 Relationship between reduction percentages of N fertilizers and N_2O emissions 81

Figure 4-1 Illustration of a 'histogram' MACC 89

Figure 4-2 Illustration of a 'curve' MACC 89

Figure 4-3 MACC for China agricultural sector (a) and cropland (b): maximum feasible abatement potential in 2020 (discount rate = 4.5%) 97

Figure 4-4 Projected BAU and abatement emissions scenarios for the whole agricultural sector (a) and croplands (b) 99

Figure 4-5 Illustration of MACC for wheat and maize related measure C1, C2 and C6 in three provinces 101

Figure 4-6 Relationship between technical, economic and market GHG mitigation potential.... 109

List of Tables

Table 1-1 Key energy and climate indicators in the China's Five-Year Plans 34

Table 1-2 Major climate and energy related policies throughout 2005-2015 in China 35

Table 1-3 Comparisons of levels of CO_2 emission drivers in China and other countries (2010) ..38

Table 2-1 GHG emission sources and contribution in Chinese agriculture 48

Table 2-2 GHG emission factors for N inputs to China's croplands 51

Table 2-3 Total N fertilizer use in agriculture and national average application rate 57

Table 3-1 Initial list of crops/soils measures and reasons for inclusion/exclusion 62

Table 3-2 Description of selected crops/soils mitigation measures and target crops 65

Table 3-3 Mitigative effects and stand-alone abatement rates of cropland mitigation measures ..69

Table 3-4 Measure adoption under baseline and abatement scenarios 73

Table 3-5 Average abatement rate, additional application area and mitigation potential of cropland measures 74

Table 3-6 Application area and overall N_2O mitigation potential from measures C1-C4 83

Table 4-1 Benefits/costs implications of cropland measure implementation................................93

Table 4-2 National average inputs (price) of rice production in 2010 and 2020 94

Table 4-3 Illustration of calculating cost of measure organic addition to rice paddies..................95

Table 4-4 Cost and cost effectiveness of cropland mitigation measures 96

Table 4-5 Example of cost effectiveness of some measures at the regional level 101

Table 4-6 Comparison of agriculture MACCs results..103

Table 4-7 Comparison of MACCs results across sectors...104

Table 5-1 Average annual credits issued ($ktCO_2e$) and number of agricultural projects registered under each standard (as of May 2014) 116

Table 5-2 Explanation, unit and values of variables for estimating the negative carbon tax rate 124

Acronyms

ACR	American Carbon Registry
BAU	Business as usual
BP	British Petroleum
CAAS	Chinese Academy of Agricultural Sciences
CAIT	Climate Analysis Indicators Tool
CAR	Climate Action Reserve
CAS	Chinese Academy of Sciences
CCB	Climate, Community and Biodiversity
CCER	Chinese Certified Emissions Reduction
CCS	Carbon Capture and Storage
CCX	Chicago Climate Exchange
CDIAC	Carbon Dioxide Information and Analysis Center
CDM	Clean Development Mechanism
CER	Certified Emission Reduction
CFI	Carbon Farming Initiative
CH_4	Methane
CMA	Chinese Meteorological Administration
CO_2	Carbon dioxide
CO_2e	Carbon dioxide equivalent
COP	Conference of the Parties
CPAs	Component project activities
DNA	Designated National Authority
DOE	Designated Operational Entity
DRC	Development and Reform Commission
EB	Executive Board
EDGAR	Emission Database for Global Atmospheric Research
EF	Emission Factor
FAO	Food and Agriculture Organization of the United Nations
GS	Gold Standard

ERU	Emission Reduction Unit
ETS	Emissions Trading Scheme
EU	European Union
EU ETS	European Union Emissions Trading Scheme
FAPRI	Food and Agricultural Policy Research Institute
FYP	China's Five-Year Plan for social and economic development
GDP	Gross Domestic Product
GHG	Greenhouse gas
GHGI	Greenhouse gas intensity
$GtCO_2e$	Gigaton of carbon dioxide equivalent
GWP	Global warming potential
HFC	Hydrofluorocarbon
HFC-23	Trifluoromethane hydrofluorocarbon 23
IEA	International Energy Agency
IPCC	Intergovernmental Panel on Climate Change
JI	Joint Implementation
JRC	EU's Joint Research Centre
$ktCO_2e$	Kiloton of carbon dioxide equivalent
LULUCF	Land Use, Land-Use Change and Forestry
MACC	Marginal abatement cost curve
MEP	Ministry of Environmental Protection
MFA	Ministry of Foreign Affairs
MOA	Ministry of Agriculture
MOST	Ministry of Science and Technology
MRV	Monitoring, Reporting and Verification
$MtCO_2e$	Megaton of carbon dioxide equivalent
Mt	Megaton
MW	Megawatt
N_2O	Nitrous oxide
NAMA	Nationally Appropriate Mitigation Action
NBS	National Bureau of Statistics

NCCC	National Committee on Climate Change
NDRC	China's National Development and Reform Commission
NEA	National Energy Administration
NPV	Net Present Value
NUE	Nutrient Use Efficiency
NZ ETS	New Zealand Emissions Trading Scheme
OECD	Organization of Economic Cooperation and Development
PFP_N	Partial Factor Productivity of N fertilizer
PoA	Program of Activities
REDD	Reducing Emissions from Deforestation and Forest Degradation
RGGI	Regional Greenhouse Gas Initiative
SAIN	China-UK Sustainable Agriculture Innovation Network
sCER	Secondary Certified Emission Reduction
SF_6	Sulfur hexafluoride
SOC	Soil Organic Carbon
tce	Tons of coal equivalent
tCO_2e	Ton of carbon dioxide equivalent
UK	United Kingdom
UN	United Nations
UNFCCC	United Nations Framework Convention on Climate Change
US	United States
USDA	US Department of Agriculture
US EPA	US Environmental Protection Agency
US IEA	US Energy Information Administration
VAT	Value Added Tax
VCS	Verified Carbon Standard
WB	World Bank
WRI	World Resources Institute
Yr	Year

Introduction

China has become the world's largest energy consumer and emitter of greenhouse gas (GHG), primarily attributed to accelerated economic development and industrial growth since the economic takeoff in 1980s and the country's heavy reliance on coal. China's per capita carbon emissions were only one-third of the US levels in 2010 but were expected to surpass the EU soon. China also consumes a lot more energy than western countries to produce one unit of GDP, giving rise to serious environmental problems that appeared to be hurdles to sustainable development. The Chinese government thus devoted to significantly improve the efficiency of domestic energy use. Consequently, China in the nation's Five-Year Plans for social and economic development started to include a reduction target for energy intensity since 1980. The target was renewed in the 12th Five-Year Plan to cut the energy use per unit of GDP by 16% between 2011 and 2015.

Additionally, facing the negative impacts of global climate change as a result of excessive anthropogenic GHG emissions, the Chinese government for the first time in history set a goal in the recent national Five-Year Plan to improve the carbon intensity of the economy by 17% by 2015 from 2010 levels. This target is an interim translation of China's voluntary commitments submitted to the Copenhagen Accord pledging to reduce its carbon emissions per unit of GDP by 40-45% by 2020 compared to the 2005 levels. While addressing climate change issues, the central government uses it as a vehicle to drive the comprehensive transition to a low-carbon economy. China has implemented a range of policies to improve energy efficiency and increase use of non-fossil energy and forest coverage. However, given the amplitude of global climate change challenge, targeting only the energy and forestry sectors without engaging other economical sectors, the current national mitigation strategy may not be sufficient to achieve the ambitious goal to decouple GHG emissions from economic growth.

So far, the agricultural sector has been only insufficiently incorporated in national mitigation policies. This absence is partially due to significant fragmentation in the agricultural systems and emissions sources, particularly in rural China where small hold farms is in predominance each only possessing in average less than one hectare of cropland. In addition, land-based emissions resulting from biological processes are spatially distributed depending on local bio-physical and climate conditions and farming practices. Owing to this high regional heterogeneity and diversity, it is difficult to accurately measure and verify GHG emissions and abatement potentials from agriculture. More importantly, the agricultural sector has historically played an essential role in China's economy as it accounts for about 10% of the national GDP, providing food and primary material for other sectors, and in support of the livelihood of over 600 million farmers. In fear of imposing negative impacts on the agricultural sector and consequently national food security, which has consistently been a top priority in China, the authorities have little motivation to regulate agriculture emissions.

However, exclusion of agriculture from national mitigation strategy presents a missed opportunity since the sector is a major contributor to national GHG emissions only second to the energy sector. It accounted for 11% of China's GHG emissions in 2005 and was responsible for over 70% of national N_2O emissions and approximately 50% of CH_4 emissions. Agriculture emissions mainly arise from livestock enteric fermentation, croplands, rice cultivation and livestock manure management. However, little progress has been made in crop and animal productivity in the past decade despite of the continuously increasing production inputs, particularly synthetic N fertilizers. This has led to major environmental issues, such as water contamination, soil degradation and erosion and GHG emissions. Therefore, Chinese agriculture stands now at the pivotal crossroads to shift to a more sustainable production mode by updating its production systems and improving farming practices.

In this regard, inclusion of climate mitigation efforts into agricultural policies could enhance the efficiency of agricultural production and accelerate low-carbon transition in this sector if agricultural management techniques could be upgraded to reduce GHG emissions without negative impacts on food security. In addition, if there exist mitigation options in this sector that are more cost-effective in reducing GHG emissions compared to those in other sectors of the economy and if they could deliver significant potentials, integrating agriculture into the national mitigation strategy would lower the overall social cost of addressing climate change. Indeed, existing global reviews (e.g. Oenema et al., 2001; IPCC, 2007; Smith et al., 2008, 2013) suggest that agriculture offers significant technical potential to mitigate climate change through both emissions reductions and carbon sequestration in terrestrial ecosystems. In addition, some reviews (e.g. Wreford et al., 2010) indicate that many abatement measures in agriculture afford win-wins solutions, i.e. simultaneously mitigating GHG emissions and improving productivity.

There have been many efforts to identify technically feasible mitigation measures that are applicable in both arable and livestock systems and to examine their technical abatement potentials in China (Lin et al., 2005; Lu et al., 2009; Huang and Tang, 2010; Nayak et al., 2014). These research results provide insights on how mitigation potentials can be applied across the range of biophysical conditions that characterize Chinese farming systems. However, there are few synthesis estimates of the overall technically feasible mitigation potential offered by agriculture, nor any estimation of the cost-effectiveness of abatement measures in this sector. Such kind of information is crucial in assisting policy-makers to decide the extent to which agriculture could be incorporated into national mitigation policies and which abatement options should be prioritized. This is the core question that this thesis tries to explore, giving a special focus to cropland emissions from fertilization.

The objective of this thesis is therefore fourfold, articulating around the evaluation of the cost-effectiveness and the potential of agricultural mitigation.

- It aims to firstly review the emission status and emission sources of Chinese agriculture and assessing its role in the national mitigation strategy. This will allow us to see if agricultural emissions can be estimated sufficiently rigorous to construct baseline scenarios to project near-term GHG emissions.
- The second objective is to identify possible options to change farming practices that could be compatible to the objectives of safeguarding national food security, and to evaluate the magnitude of abatement potentials offered by the mitigation measures against the baseline.
- In the third place, we will investigate the economic implications for farmer when adopting mitigation measures compared to the baseline farming practices, based on which we will deliver the cost-effectiveness analysis of identified abatement options and suggestion on priority of mitigation actions for purpose of policy-making.
- The final objective is to examine the development of carbon pricing mechanisms (tradable permits and taxes) in China and elaborate how the economic instruments could be used to reduce agricultural emissions most cost-effectively.

Responding to these questions requires a research tool that is able to incorporate the assessments on technical mitigation potentials with that of the economic analysis from the agricultural sector: the Marginal Abatement Cost Curve (MACC). MACCs show the cost-effectiveness of different mitigation options as well as the overall abatement potential that can be offered. They also enable comparison of the cost-effectiveness of abatement options within a special sector and/or between different sectors of the society. This made MACCs an important tool to assist policy makers in prioritizing mitigation options during the last decades with a plentiful of literature dedicated to MACC construction and analysis.

There are two families of approaches that have been used to construct a MACC: top-down and bottom-up. A top-down analysis allocates an exogenously determined emission reduction requirement downward through modeling assumptions to conclude an overall abatement cost to the economy. This approach employs either microeconomic supply-side models (De Cara et al., 2005; Hediger, 2006; USEPA, 2006; De Cara & Jayet, 2011) or macroeconomic partial or general equilibrium models (IPCC, 2007; Schneider et al., 2007; Smith et al., 2008; Domínguez et al., 2009). Engineering-oriented bottom-up approaches of MACC studies (Beach et al., 2008; Moran et al., 2011; Schulte et al., 2012; Pellerin et al., 2013) analyze the abatement cost and mitigation potential of individual measures by taking into account heterogeneities in terms of abatement potential, applicability and implementation costs of mitigation options.

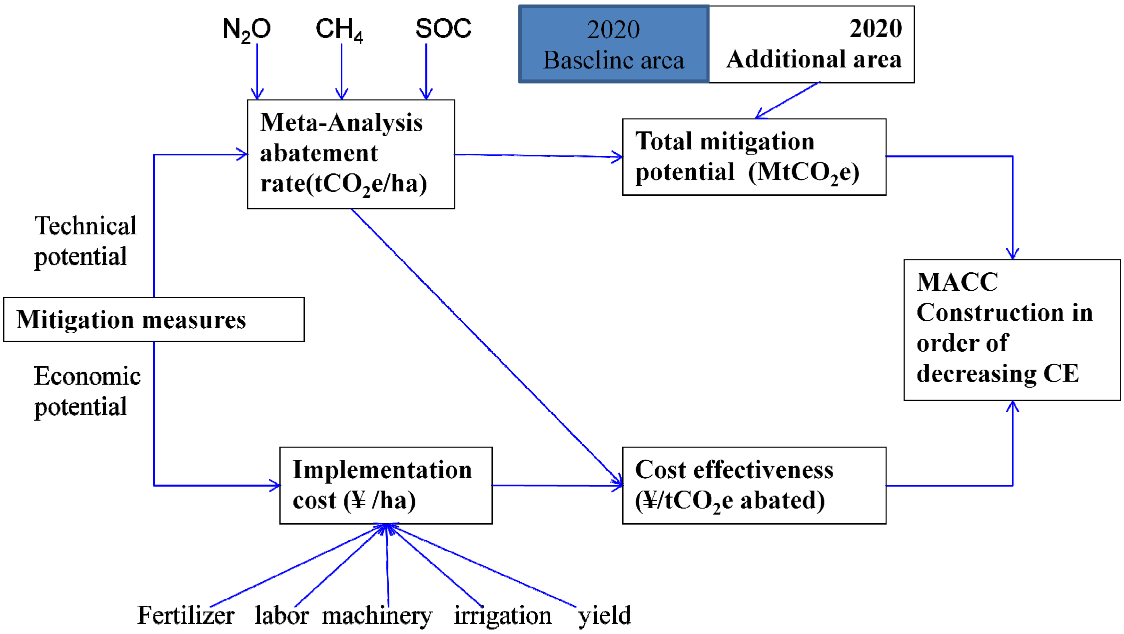
Owing to the fragmented nature of Chinese agriculture emissions and the attempt to making maximum use of the field experiment data collected, this thesis will apply the bottom-up approach to construct a MACC for the Chinese agricultural sector. This MACC excise will consider the extent of biophysical data on agricultural mitigation measures and thereby allowing for the aggregation of the mitigation potential arising from the application of a subset of cost-effective measures above a notional baseline level of activity that we denote as the business as usual (BAU) scenario.

This thesis is structured around the derivation and utilization of a MACC for Chinese agriculture i.e. i) identification of BAU GHG emissions; ii) evaluation of the technical and economic potentials; iii) assessment of possible economic tools to realize the identified mitigation potentials. The basic steps for constructing a bottom-up agriculture MACC is illustrated by Figure 0-1, following the general methodology (e.g. as described in Moran et al. (2011)) but with adjustments to better accommodate national context and reflect country specificities. Another point to bear in mind is that this study will only address GHG emission within the farm gate as opposed to conducting life-cycle analysis. This thesis is therefore structured in five chapters, which articulate and extend the contents of the three major academic papers (Wang et al., 2014, 2015; Quemin & Wang, 2014) that the candidature published during the PHD study.

The first chapter is a general overview of Chinese climate policies to assists us in understanding the current status and challenges of integrating agriculture into the national mitigation strategy. The reader will first be informed of the historical and current GHG emissions in China and the contributors to the increasing carbon emissions. This will help understand Chinese government's positions in international climate negotiations as well as its considerations of incorporating climate change objectives into national social and economic development plans. After, we will discuss major energy and climate policies adopted in China, examining their efficiency in reducing energy intensity in the past and the sufficiency to achieve future climate targets. This background overview will show that despite agriculture being a major source of increasing GHG emissions in China, this sector is currently hardly integrated in the national mitigation strategy.

Chapter 2 focuses on the construction of a most plausible baseline for near-term GHG emissions from predicted BAU activities of the Chinese agricultural sector until 2020. We will first present agricultural development in China and its implications on GHG emissions. It will show high level of emphasis from the authorities' concerning food security that can be seen in all agriculture policies. This core principle will be incorporated into the construction of baselines used to assess mitigation potentials in the following chapters. For instance, increase of agricultural production is well underlined in projecting future agricultural activities that will align mitigation objectives with the food security priority. An analysis of agriculture GHG emissions will be carried out to identify the most appropriate methodologies for calculating emissions in this sector. It will show that agriculture GHG emissions will continue to increase in the timeframe to 2020 under the BAU scenario with livestock emissions growing faster than those from croplands. In addition, the studies will specially focus on the GHG intensity of the three major cereal crops on a provincial and historical scale to reveal the geographic variations and historical curve of the carbon footprint of crop production.

Figure 0-1 Development process of a bottom-up MACC for Chinese agriculture (cropland emissions)



Source: Author

Chapter 3 assesses the technical mitigation potentials that are feasible in Chinese agriculture, primarily those from cropland emissions, following the steps presented in the upper part of Figure 0-1. The bottom-up MACC exercise starts with identifying agriculture mitigation measures that have wide application opportunities in China and offer significant abatement potentials. By comparing the selected mitigation measures with the common practices under the defined baselines, we will be able to estimate the per area abatement rates, i.e. tCO_2e abated per hectare. Quantification of the abatement rates will take into account effects of measure interactions and rely on meta-analysis that collected data from hundreds of experiments carried out throughout China. After, the additional areas available for measure application will be examined to allow for estimating the overall feasible mitigation potentials from indentified measures. This potential represents approximately one-third of the baseline agriculture emissions in 2020. Since synthetic and organic fertilizer management practices are expected to provide the largest mitigation potentials from cropland emissions, we will perform an in-depth analysis of mitigation measures related to N-use in cereal production at the regional level to calculate an aggregated national figure. We will see that abatement rates and mitigation potentials vary significantly among provinces corresponding to the high differences in regional GHG intensity of cereal production.

In Chapter 4, a literature review of existing agriculture MACCs will first be carried out to explain the choice of applying the bottom-up approach to construct the MACC for Chinese agriculture. After, we will outline the stages in moving from a technical potential to an estimate of feasible economic mitigation potential, as illustrated by the lower half of Figure 0-1. First, we will estimate the implementation cost of each measure by comparing changes in the economic performance of applying mitigation options by farmers with those under the baseline. The costs/benefits considered in our approach will only include those occurring directly to farmers, i.e. without extending beyond the farm gate. We also account for anticipated future price rise for various agricultural inputs/outputs. Per area abatement cost is then combined with the per area abatement rate to estimate cost effectiveness of mitigating one tonne GHG (expressed as ¥/tCO_2e) of each measure. We will then construct the MACC and the abatement scenarios for the agricultural sector to 2020 and conclude that over one-third of the total mitigation potential could be realized at negative costs while another one-third from low-cost abatement measures. We will also discuss regional MACCs since a national MACC is not able to reflect the heterogeneity in cost-effectiveness and mitigation potentials at the regional level. This information would be more valuable to assist regional policy-makers. A comparison of the Chinese agriculture MACC with similar studies in the other countries will allow us to see the pertinence of the thesis outcome. Additionally, comparing our results with MACCs from other economic sectors will show the importance of integrating agriculture into national mitigation strategy. A sensitivity analysis will be performed to test the robustness of the various assumptions underpinning MACC construction. Finally significances and limits of the MACC study will be discussed.

Chapter 5 explores how the research outcomes on MACC analysis could support policy making on combating climate change in agriculture. The authorities generally adopt “command and control” approaches by setting up standards and rules to address environmental issues. But the economists argue that economic tools are more cost effective in alternating the behavior of economic agents through the introduction of a price on carbon, which reflects the cost of excessive emissions in a context where traditional markets fail to account for environmental externalities. In China, the development of seven carbon emission trading pilots and the government’s intention to initiate a national scheme has attracted much attention and thereby an extensive analysis will be carried out. Another economic tool - carbon tax- is also under discussion in China, but we will put more emphasis on the adverse impacts of N fertilizer subsidies, which could be regarded as a negative carbon tax. We will see that agriculture is marginalized in the current carbon pricing schemes because of high transaction costs arising from the diffuse nature of emission sources and abatement actions, difficulties of accurate emission measurement and verification, and specificities of carbon storage in soils. Taking these barriers into account, we will propose some suggestions on an effective use of economic instruments in regulating agricultural emissions. It is urgently needed to restructure the current subsidy systems to create a better economic signal on fertilizer use. Finally, we shall recommend the designing elements to scale-up crediting schemes, especially via the use of regional GHG intensity results as baselines. Advancement in the national offset markets shall aliment the preparation of a pilot ETS covering agriculture with innovative features.

Chapter 1 - General overview of climate policies in China

China has made remarkable progress in terms of economic development since the Reform and Opening-up in the early 1980s. With GDP growth rates averaging 9-10% over the past 30 years, China has now become the second largest economy in the world (World Bank, 2013). However this strong economic performance has been achieved at a high cost of environment with air, water and soil pollution spreading to large areas of the nation (e.g. MEP et al., 2014). Regarding climate change challenges, China is now the world's largest annual emitter of GHG emissions and the top energy consumer. During the 2000s, environmental and climatic issues have gradually stepped in the society as major concerns and they are progressively prioritized in the National Plans for Economic and Social Development.

Collective actions from all nations are needed to effectively tackle the vast threats of global warming and climate change. It is hard to achieve the ambitious climate goals successfully without considerable efforts from China, which is determined by China's engagement in international climate negotiations and its policies and actions domestically to address climate change. Full comprehension of China's climate strategies needs to be built on a better understanding of the nation's development challenges and the causes to its GHG emissions. It is also essential to analyze how mitigation strategies and actions will shape the future emissions curves and whether they could guarantee the fulfillment of climate targets. For the purpose of this thesis research, it is central to consider the agriculture contribution to the national GHG emissions and how this sector is taken into account in the national climate strategies.

In this chapter, we are first going to present China's GHG emissions profile and the growth drivers of CO_2 emissions (section 1), followed by an overview of China's participation in international climate negotiations and how climate dimension is gradually integrated into national development designs (section 2). Section 3 gives an in-depth analysis of the climate policies and actions and how they framed the energy intensity evolution in the past and will affect future energy and climate performance in China. This background overview allows us to identify the challenges of incorporating agriculture into national climate strategies (section 4).

1. Chinese GHG emissions and drivers of CO_2 emissions growth

This section first presents the magnitude of GHG emissions in China and the major contributing sectors (section 1.1). Investigation on the historic evolution of CO_2 emissions is performed to understand the emission trends (section 1.2) and the factors driving CO_2 emissions growth in China (section 1.3).

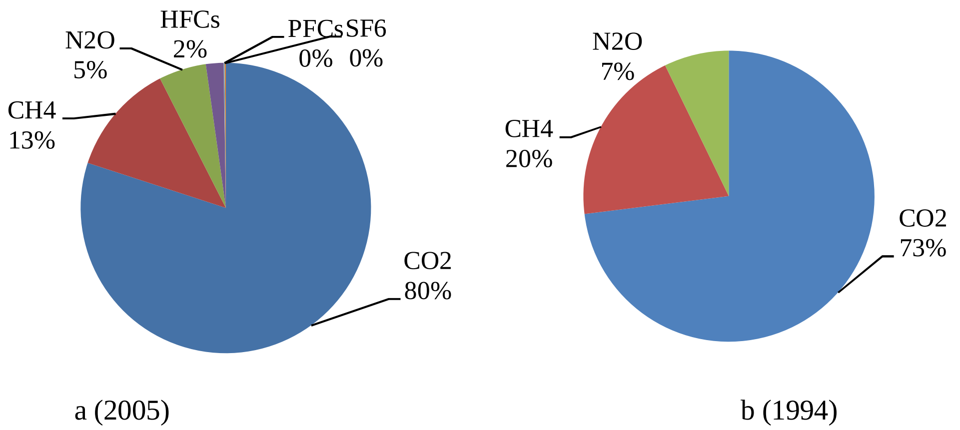
1.1. GHG emissions according to national inventories

China, as a non-Annex I Party to the United Nations Framework Convention on Climate Change (UNFCCC), is not obligated to report its GHG emissions on an annual basis. As of 2014, the Chinese Government has published two national inventories of GHG emissions - those of 1994 and 2005, incorporated into the Initial and Second National Communication on Climate Change submitted to the UNFCCC in 2004 and 2012, respectively (NCCC, 2004, 2012). In compiling the inventories, China followed the IPCC Guidelines for National Greenhouse Gas Inventories (revised 1996, Good Practice and 2006 versions) and selected emission sources, activity data and emission factors according to national circumstances. China chose to estimate only carbon dioxide (CO_2), methane (CH_4), nitrous oxide (N_2O) emissions when conducting the first inventory, but reported on all the six GHGs for the second one. In the meantime, more emission sources under the five sectors, i.e. energy, industrial processes, agriculture, waste management and land use change and forestry, were accounted in the 2005 inventory.

According to the national inventories, in 2005 China's GHG emissions totaled 7,976 million tons (Mt) CO_2 equivalent (CO_2e) excluding Land Use, Land Use Change and Forestry (LULUCF), more than double the amount in 1994 (3,650 Mt CO_2e). The total net carbon removal through LULUCF was estimated to be about 421 and 407 Mt CO_2e in 2005 and 1994, respectively, attributable to the government's continuous efforts to raise the national forestry coverage (a net increase of about 4.1 Mha forestry each year during 2003-2008). Among the GHG emission sources, fossil fuel combustion was the largest contributor accounting for over 70% of global warming potential (GWP) weighted emissions (Figure 1-1 and Figure 1-2). Agricultural production came at the second place and was the primary source of CH_4 and N_2O emissions followed by industrial processes which were an importance source of CO_2 emissions. The two sectors together represented approximately 21% of national total emissions in 2005 while around 2% arose from waste management in the form of CH_4.

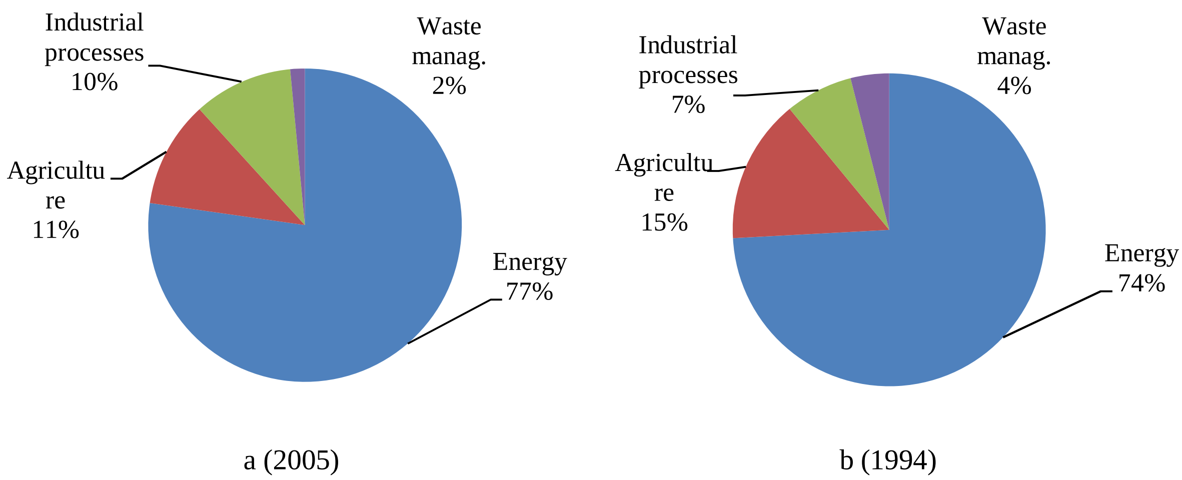
Prior to the release of the second national inventory, the International Energy Agency (IEA, 2011) and the Climate Analysis Indicators Tool (CAIT, 2013) of the World Resources Institute (WRI) also estimated China's GHG emissions for 2005. Their results of 7,527 and 7,059 Mt CO_2e are lower than the national statistic (7,976). The discrepancies can be partially attributed to the inconsistencies between the energy consumption data from the national statistical system and that of the IEA (Zhu, 2013). In the future, the Chinese government is committed to start reporting its emissions more frequently, even on a biennial basis with the assistance from developed countries, as outlined in the Copenhagen Accord.

Figure 1-1 Composition of China's GHG emissions by gas in 2005(a) and 1994(b)



Source: China's Second and Initial National Communication on Climate Change (NCCC, 2012, 2004)

Figure 1-2 Composition of China's GHG emissions by sector in 2005(a) and 1994(b)

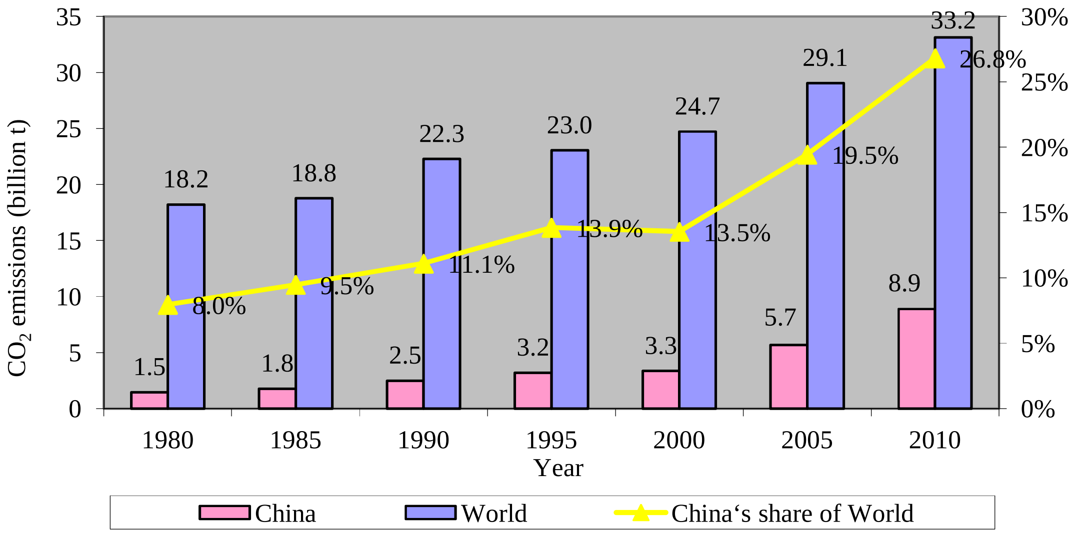


Source: China's Second and Initial National Communication on Climate Change (NCCC, 2012, 2004)

1.2. Historic trends of CO_2 emissions from fossil fuel combustion and cement production

For the purpose of illustrating historic trends, data on GHG and CO_2 emissions need to be updated at regular time intervals or preferably on an annual basis. However, such information is absent in official statistics and it is thus necessary to direct to other sources that release CO_2 emissions data at constant intervals, such as those by the IEA, the British Petroleum (BP), the Emission Database for Global Atmospheric Research (EDGAR) of the EU's Joint Research Centre (JRC)/PBL Netherlands Environmental Assessment Agency, the Carbon Dioxide Information and Analysis Center (CDIAC), the US Energy Information Administration (EIA) and the WRI's CAIT. National experts (e.g. Zhu, 2013) indicated the compatibility of IEA estimates with national inventories of CO_2 emissions; we thereby use the CAIT database to illustrate CO_2 emissions evolution in China (CAIT, 2013). The CAIT database include CO_2 emissions from fossil fuel combustion, drawing from estimates by the IEA, and those from cement production, drawing from estimates by the CDIAC. Generally, CO_2 emissions from cement production account for over 70% of CO_2 emissions from the whole industrial processes in China. Figure 1-3 presents the evolution of national CO_2 emissions and China's share of the world's total from 1980 to 2010. Figure 1-4 highlights the growth rates of CO_2 emissions in relation to the growth rates of energy use in China from 1990 to 2008 (CAIT, 2013).

Figure 1-3 Evolution of China's CO_2 emissions and global share from 1980 to 2010

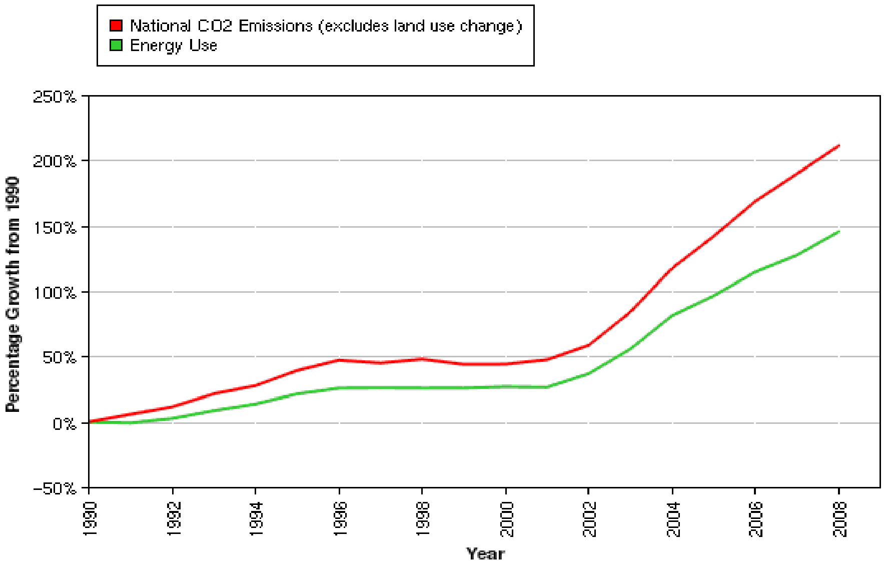


Source: WRI's CAIT (2013)

China's CO_2 emissions from fossil fuel combustion and cement production, excluding LULUCF, amounted to 8,896 Mt in 2010 and accounted for about 26.8% of the world emissions. As Figure 1-3 illustrates, while the national CO_2 emissions in 2010 were more than twice their 1990 levels, this growth occurred principally between 2002 and 2010 after a slight decrease during 1996-2000. Since 1979, following on China's Economic Reform and Opening-up, annual carbon emissions increased nearly 6-folds. But it should be mentioned that about 33% of the emissions were produced making goods for export in 2006 compared to only 12% in 1987(Wang & Watson, 2008). National CO_2 emissions have been evolving at slightly higher pace than the total primary energy consumption - the essential driver of CO_2 emissions.

The per capita CO_2 emissions in China were about 6.65 tons in 2010, which were about one-third those of the US (18.33 tons) and approached the EU-27 levels (8.08 tons). The BP report (2014) predicts that per capita carbon emissions in China will surpass the EU in 2017 and the OECD average in 2033, but remain below the US level in 2035.

Figure 1-4 Growth trends of CO_2 emissions and energy use in China from 1990 to 2008



Source: WRI's CAIT (2013)

1.3. Drivers of CO_2 emissions growth in China

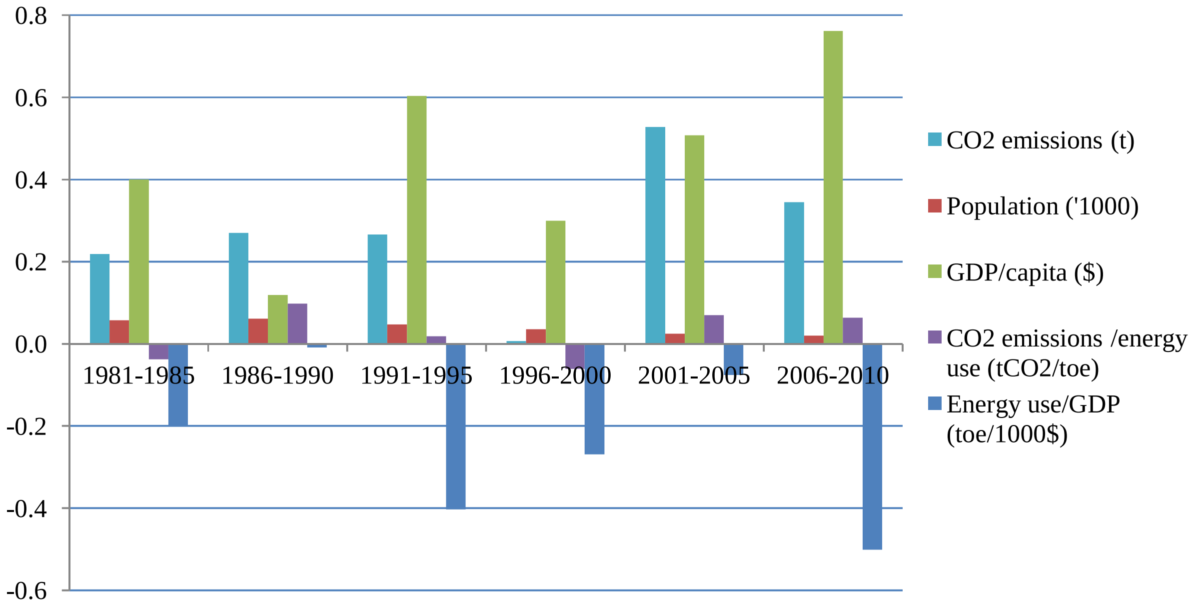
To help explain the causes of CO_2 emissions growth, carbon emissions can be divided into four drivers: population, per capita GDP, energy intensity of the economy, and the CO_2 content of the energy use (Kaya, 1990). The relationship can be expressed as:

$$CO_2\ emissions = population \times \frac{GDP}{population} \times \frac{CO_2\ emissions}{energy\ use} \times \frac{energy\ use}{GDP}$$

$$\underbrace{\frac{GDP}{population}}_{\text{Per capita GDP}} \quad \underbrace{\underbrace{\frac{CO_2\ emissions}{energy\ use}}_{CO_2\text{ intensity}} \quad \underbrace{\frac{energy\ use}{GDP}}_{\text{Energy intensity}}}_{CO_2\text{ intensity per unit of GDP}}$$

The contribution of the four drivers to overall CO_2 emissions at 5-year intervals during 1981-2010 is calculated by the author using the method developed by Kaya. Results are plotted in Figure 1-5. CO_2 emissions data are from the CAIT database, GDP and population information is extracted from the World Bank database (2013), and the energy use is collected from the BP statistical review (2013). The results illustrate that among the four drivers, only the energy intensity of GDP has been in a steady decrease since 1980 accelerated by policy interventions and concerted efforts, especially after 2005 (see section 3.2 for more details). Figure 1-5 also indicates that the growth in total CO_2 emissions from 1981 to 2010 was largely triggered by the increase in per capita GDP (responsible for over 70% growth), but was nearly inelastic to population growth. The impact of China's carbon intensity of energy consumption on CO_2 emissions was not consistent over time. It declined from 1996 to 2000 as the share of coal in the primary energy mix reduced from 73.5% in 1996 to 68% in 2002 while the share of oil increased from 18.7% to 22.3 % (NBS, 2011). The trend reversed when oil prices started to rise quickly in 2002, stimulating the consumption of coal. As a result, the share of coal reached a periodic peak of 70.4% in 2009. Not only was more coal consumed in this period, but lower-grade coal with higher carbon content was used to meet the unexpected energy demand brought on by rapid economic growth (Levine & Aden, 2008). Consequently, carbon intensity of energy use grew by 15% from 2001 to 2010, contributing 8.8% to the increase in CO_2 emissions since 2001.

Figure 1-5 Drivers of CO_2 emissions growth in China from 1981 to 2010



Note: For each period, the evolution of CO_2 emissions (expressed as a difference of natural logarithms) is presented by the first bar on the left, which is the net sum of the impact of the evolution of its drivers on the same period shown on the right.

Source: Author's calculation based on data from CAIT, WB and BP (2013)

The decomposition analysis of CO_2 emissions drivers indicates the possible channels that would be preferred by the Chinese government to control its emissions growth in the future while sustaining required economic development. Efforts are therefore needed to reduce the energy use per GDP by upgrading industrial performance and conserving energy, and to move towards a cleaner energy mix by alleviating reliance on coal and accelerating the development of non-fossil fuels. This corresponds to the core pieces of energy- and climate-related policies that will be elaborated in section 3.

2. From international engagement to national climate policies

2.1. Framework of climate policy marking in China

Originally, the Chinese Meteorological Administration (CMA), along with the Chinese Academy of Sciences (CAS), the Ministry of Science and Technology (MOST) and the Ministry of Foreign Affairs (MFA), directed climate change coordination efforts to reflect China's initial perception of climate change as a scientific and international issue. As climate change evolved from a scientific topic into an issue also involving economic development and political negotiations, the State Planning Commission (SPC) became the hub for climate change policies in China in 1998. The SPC was renamed as the National Development and Reform Commission (NDRC) in 2003 and holds the broad administrative and planning control over China's social and economic development.

NDRC's climate responsibilities are entrusted to the Department of Climate Change, including formulating key strategies and policies dealing with climate change, representing China in international climate negotiations, and coordinating the work of conducting national GHG inventories. Other ministries and government agencies participate in climate-related policy-making by providing their corresponding expertise. For example, the Ministry of Environmental Protection (MEP) formulates concrete regulations and standards, the MFA assists the NDRC in international climate negotiations, the MOST provides technical advice, and the CMA participates in the work of IPCC.

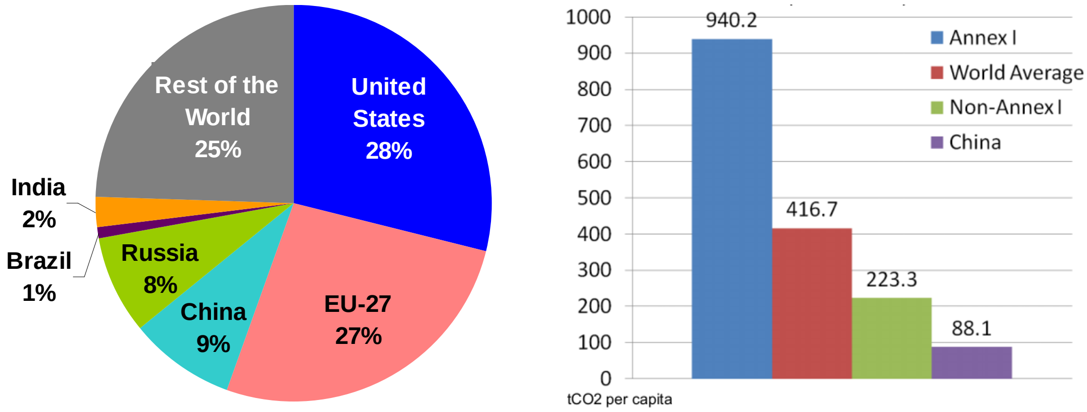
2.2. China's engagement in international climate negotiations

Grounds underpinning China's positions in climate negotiations

In order to better understand China's standpoints in climate negotiations and consequent actions, it is worth taking a look at the basis underpinning the government's positions, which are the nation's cumulative CO_2 emissions. Although China now contributes over 25% of the world's annual CO_2 emissions as the top emitter, it is responsible for merely 9% of cumulative emissions between 1850 and 2006 (Figure 1-6 (a)). In terms of cumulative CO_2 emission per capita (Figure 1-6 (b)), it is far less than the levels in developed countries and is about 1/5 of the world average.

China therefore advocates taking into account these elements in proposing its international mitigation obligations and considering equity in sharing atmospheric resources. Throughout various climate talks and negotiations, China has reiterated the principle of 'Common but Differentiated Responsibilities' that urges developed countries to bear primary responsibility for the historical concentration of GHGs in the atmosphere and thus to take the lead in combating climate change. China also states that developed countries should provide financial resources and technical assistance for developing countries to adopt appropriate measures to mitigate and adapt to climate change.

Figure 1-6 World cumulative CO_2 emissions 1850 – 2005: total and per capita



(a) (b)

Source: CAIT of WRI and Tsinghua University

Evolution of China's engagement in climate negotiations

China has been engaged in international climate discussions since the early 1990s. It formally ratified the UNFCCC in 1992 as a non-Annex I country and the Kyoto Protocol on August 30, 2002, as a non-Annex B country. China was not bound by any emission reduction obligations under the Kyoto Protocol, nor was it willing to take commitments that might constrain its economic growth. In the following years, although global warming gradually received more attention in China as well as internationally, no explicit climate goal was defined nationally. Still, the First National Assessment Report on Climate Change (NARCC) was released in 2006, which assessed the impact of climate change on the main range of economic sectors and put forward both mitigation and adaptation policies and measures. The Second NARCC was published in November 2011.

Since 2007, climate change has quickly become a much-talked-about topic in both the political and scientific spheres, and has rapidly emerged as one of the priorities on the Chinese government's agenda. The cornerstones of climate-related policies in China were the National Climate Change Program (NCCP), released in June 2007, and the China's Policies and Actions for Addressing Climate Change (CPAACC) in October 2008. The latter was updated by the NDRC on an annual basis thereafter. In addition, the National Climate Change Adaptation Strategy was published in November 2013. These national communications outlined China's efforts both to mitigate and adapt to the impacts of climate change, its long-standing positions in climate negotiations, its consideration of integrating climate change into national economic and social development strategies and the significance of setting up a national leading committee on addressing climate change.

With the issue of climate change continuing to heat up on the international agenda and as the top annual CO_2 emitter, China has been under increasing pressure from industrialized countries to take on more mitigation responsibilities. More importantly, the adverse impact on living conditions, agriculture production and health caused by rapid growth in carbon emissions triggered wider discussion at the national scale on the need for China to switch to a more environment-friendly development pattern. In this context, the Chinese government has begun to consider the possibility of making firm commitments on climate change as an important vehicle to redirect economic development pathway. A notable milestone is China’s climate mitigation actions submitted under the Copenhagen Accord, the core elements of which were the voluntary pledges to reduce its CO_2 emissions by 40-45% per unit of GDP by 2020 compared to 2005 levels, and increase the share of non-fossil fuels in primary energy consumption to around 15%, forest coverage by 40 million hectares, and forest stock volume by 1.3 billion cubic meters by 2020 from 2005 levels. These engagements reflect both China’s long-held position to conserve its development rights by putting an intensity constraint on carbon emissions, and its willingness in making greater strides to minimize its carbon footprint.

China stepped up its efforts at the Conference of Parties (COP) 17 in Durban, voicing its intention to be engaged in a post-2020 legally binding framework for emission reduction under certain conditions. This was the first time that China mentioned a timeline for taking on future legally binding obligations to control its emissions growth, although contingent upon progress of international climate talks and China’s domestic development by 2020. Such proposals may be presented at the end of 2014 or early 2015.

2.3. Inclusion of climate targets into national development plans

Early 2011 witnessed the translation of these voluntary international commitments into domestic policies as they were integrated into the national economic and social development plan, i.e. the 12th Five Year Plan (FYP), as the vehicle for the transition towards a low-carbon economy. The Outline of the 12th FYP, released in 2011 to cover the period of 2011-2015, established the policy orientation of promoting green and low-carbon development, and explicitly set out mandatory targets on both energy intensity and carbon intensity among a range of sustainable development goals. In the meantime, compulsory goals were set for the share of non-fossil fuel in China’s energy mix and the increase of forest coverage and forest stock volume. While objectives for carbon intensity and forest stock volume were the first-ever to be introduced in a FYP, the other goals followed up on and expanded the ambitions of the 11th FYP (2006-2010). Table 1-1 illustrates the progression of energy and climate related targets in the 11th FYP, the 12th FYP and those for 2020.

Table 1-1 Key energy and climate indicators in the China's Five-Year Plans

Indicators	11th FYP (2006-2010) Target	11th FYP (2006-2010) Result	12th FYP (2011-2015) Target	13th FYP (2016-2020) Anticipated target	Nature of target
Energy Intensity (% reduction in 5 years)	20%	19.1%	16%	Not yet available	mandatory
Carbon Intensity (% reduction in 5 years)	NA	20.5%(according to ERI research)	17%	40-45% vs 2005	mandatory
Non-fossil fuels in primary energy consumption	9%[2]	9.6%[2]	11.4%	15%	mandatory
Forest coverage	Up to 20% from 18.2%	20.4%	21.7%	23%	mandatory
Forest stock volume	NA	13.7 from 12.5billion m^3	14.3 billionm^3	15 billion m^3	mandatory
Annual GDP growth rate	7.5%	11.2%	7%	Not yet available	expected
National energy consumption[3]	NA	3.25 from 2.36 billion tce[1] (+6.6% annually)	4.0 billion tce (+4.2% annually)	Not yet available	expected
National electricity consumption[3]	NA	4192 from 2494 billion Kwh (+10.9% annually)	6150 billion Kwh(+8.0% annually)	Not yet available	expected

Note: 1. *tce* stands for metric tons of coal equivalent, unit used by China for energy statistics. 1 tce equals 29.31 GJ or 7 million kcal at low heat value.

2. The target was set for renewable energy instead of non-fossil fuel and was outlined in the '11th FYP for Energy Development -released in 2007.

3. Cap on total energy consumption and electricity consumption were identified in the "12th FYP for Energy Development", released in January 2014.

Source: Data compiled from FYPs and government reports

3. Policies and actions to meet the mitigation targets

3.1. Climate and energy related policies throughout 2011-2015

To address both the economic development and climate change challenges, climate policies in China shall accommodate the GDP growth priority but reorient it to be less carbon intensive. In line with this core principle, policies in four fields were prioritized to control emissions: accelerate the adjustment of the industrial structure, promote energy conservation, develop low-carbon energy sources, and increase forest carbon sinks (Table 1-2). Most of the policies and measures for 2011-2015 under these four pillars showed a continuation and expansion of those defined for the 11th FYP period.

Table 1-2 Major climate and energy related policies throughout 2005-2015 in China

Policy pillars	Goals	Major policies and programs
Sectoral structure adjustment	Raise the GDP contribution of the service industry and new strategic industries	- GDP contribution of the service industry rises to about 47% in 2015 up from 43% in 2010. - 7 new strategic industries specified in the 12th FYP: advanced materials, information technology, innovative equipment manufacturing, biotechnology, etc.
	Inhibit excessive growth of energy-intensive and emission-intensive industries	- Reinforce the entrance standards for energy-intensive industries by imposing taxes and raising safety, energy and environmental standards - Restrict the export of energy-intensive products - Prevent shift of polluting and backward production facility to central and west China - Introduce punitive electricity tariffs for energy-intensive industries
	Phase-out obsolete production facilities	The Ministry of Industry and Information Technology (MIIT) allocated concrete tasks of eliminating outdated and polluting capacities to provinces and individual enterprises and released the list of enterprises subject to public supervision in 19 sectors.
Energy conservation and energy efficiency improvement	Save the equivalent of some 300 million tce during 2011-2015(Ten Key Energy Conservation Programs)	The billion dollar effort to provide financial incentives to promote a wide range of energy saving projects (coal industrial boilers or kilns, waste heat recovery/waste power recovery, petrochemical conservation or substitution, electrical machinery energy saving system and energy system optimization).
	Top-1000 Energy Consuming Enterprises Program in 11th FYP, extended to 10,000 Enterprises Program in the 12th FYP	This program involved initially the top 998 most energy-intensive enterprises in 9 industrial sectors, which accounted for 43% of the nation's total CO_2 emissions in 2006. The extended 10,000 enterprises program in fact covers more than 17,000 top emitters representing 2/3 of China's total energy consumption. They are required to achieve an absolute energy-saving target of 250 Mtce by 2015.
	Promote energy efficiency improvements in other sectors	Standards and detailed actions will be set for the building, transportation and rural energy use sectors.
	Foster market-based mechanisms	Promote energy service companies (ESCOs), energy cap and trade trading, trading of energy conservation certificates.
Energy mix	- Develop hydropower taking into account environmental protection - Develop safe nuclear power. - Promote wind, solar, biomass and geothermal energy adapted to local conditions. - Increase share of natural gas and clean coal.	The "renewable energy law", enacted in 2006 and amended in 2009, introduced a series of incentivizing polices: a provision for renewable portfolio standards (also called 'mandated market share'), feed-in tariffs for biomass, 'government-guided' prices for wind power, an obligation for utilities to purchase all renewable power generated, new financing mechanisms and guarantees (e.g. exempts renewable energy projects from local income taxation), and other market-enhancing provisions.
Forest coverage and sequestration	Afforestation programs , forestry conservation programs and restoration of desertification land	A range of indicators were set for 2015 in terms of land acreage dedicated to forestry and increase of restored desertification, wetland, and natural forestry conservation area.

Source: Compiled by the author from government laws, guidelines and plans

While the two compulsory targets of 17% cut in carbon intensity and 16% cut in energy intensity are both intensity-based, the "12th FYP for Energy Development" puts caps on total energy and electricity consumption to accelerate the country's switch from energy-intensive growth patterns and to limit exposure to energy dependence risks. The government intends to restrict the national 2015 energy consumption to below 4.0 billion tce and total electricity use to 6,150 billion Kwh. The suggested target of 4 billion tce implies that China will have to rein in growth of energy use at 23%, or 4.2% annually during 2010-2015: this corresponds to a massive effort considering domestic energy consumption increased by 37.7% over 2005-2010, or 6.6% annually. In 2013, energy consumption in China reached 3.76 billion tce or a 3.9% increase from 2012. In 2014, controlling excessive growth of total energy consumption topped the list of priority tasks put forward by the National Energy Administration (NEA), indicating China's urgent challenges in meeting its energy need. The energy consumption cap for 2014 is reported to be set at 3.89 billion tce.

In terms of adjusting the sectoral structure, a more ambitious goal was set for the growth of the service industry: 4% growth in GDP share from 2011 to 2015 (while only 2.8% increase was reached during 2005-2010), and the number of sectors forced to eliminate backward production technology and facilities has been extended from 12 to 19. With regards to energy savings, the number of firms brought under a national absolute energy-saving program was increased tenfold to hold more enterprises accountable for energy efficiency and conservation targets.

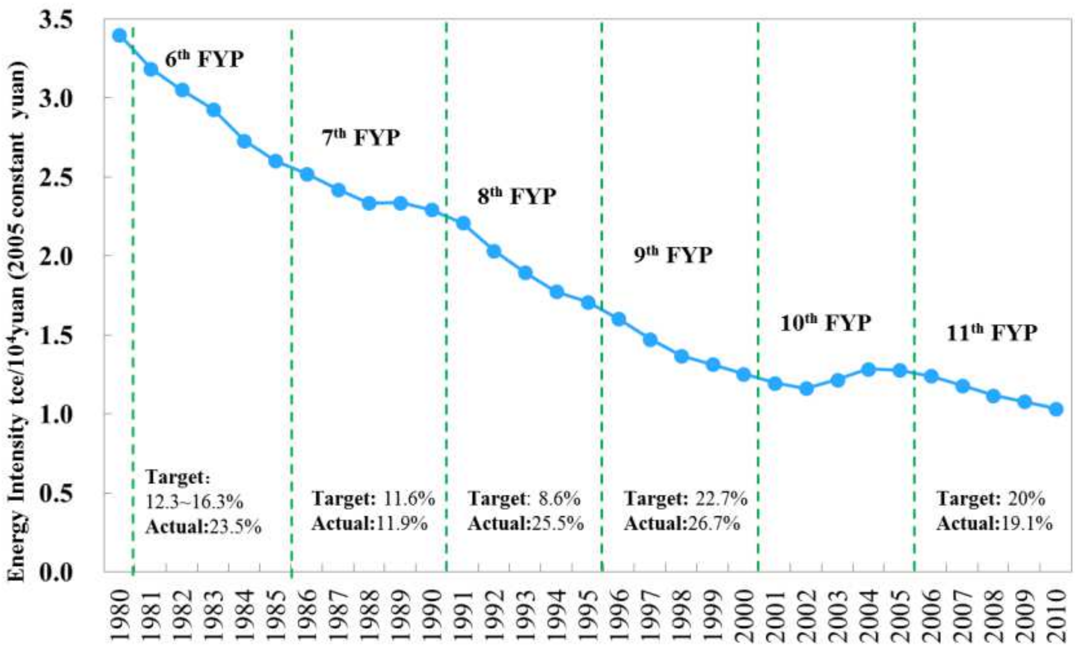
Renewable energy development is a key element of energy policies in China. The "12th FYP for Renewable Energy Development" (NEA, 2012) calls for a total of 4,780 million tce capacity from renewable energy to be built by 2015, accounting for at least 9.5% in the energy mix. One hundred and sixty million kilowatts (or GW) of new installed power generation capacity from renewable energy, including 61 GW hydropower, 70 GW wind power, 20 GW solar power and 7.5 GW biomass, are planned to make renewable energy provide over 20% of electricity generation in 2015. Efforts should be strengthened to integrate wind energy into the electricity grid, raise requirements for wind energy technology and quality, improve the subsidy system for solar energy, promote the application of distributed solar electricity generation, and reinforce renewable energy use in rural areas.

3.2. Energy intensity evolution in China

Since energy intensity and carbon intensity of the economy are internally correlated, the achievements of carbon intensity goal shall to a large extent be determined by the evolution of energy intensity. Figure 1-7 illustrates China's continuous decline in energy intensity despite an important increase in energy demand (Figure 1-4) from the onset of the economic reform in 1978 up to 2000. As a result, in 2000 the energy use per GDP was nearly two-thirds lower than it was in 1980. Ma et al. (2008) conducted a study to examine the drivers of this decline and concluded that technological change was the dominant factor in bringing down energy intensity while structural change at the industry and sector (sub-industry) level actually increased energy intensity over the 1980–2000 period. In the light of overachievements of energy intensity targets subscribed to previous FYPs, the government removed the energy intensity target from the 10th FYP.

The improvement on energy intensity begun to level off as the Chinese economy moved into a phase of intensified industrialization and urbanization from 2001 onwards. The energy intensity rose in 2002 and 2003, resulting in a 1.8% increase during 2001-2005. In response, the government re-introduced an energy efficiency target into the 11th FYP and plotted out a diverse range of policies to promote energy conservation, energy efficiency improvement and renewable energy during 2006-2010. These actions reversed the upward trend in energy intensity that had been experienced in the 5 years prior to 2006. At the end of 2010, China achieved a 19.1% reduction in energy intensity against the 2005 level, barely missing the 20% target defined in its 11th FYP. According to the NDRC, this energy intensity improvement represents energy savings of 630 Mtce against a business-as-usual baseline, and a CO_2 emissions reduction of 1460 Mt during 2006-2010. Climate Policy Initiative of Tsinghua University (2011) concluded that energy efficiency improvement was the main driver of this CO_2 emissions abatement (about 87% contribution) while a cleaner energy mix was a less significant contributor.

Figure 1-7 Energy intensity evolution during the FYP periods from 1980 to 2010



Note: data were collected from FYPs; energy intensity (tce/10,000RMB) was calculated based on 2005 constant prices.

Source: Climate Policy Initiative at Tsinghua University (2011)

Although China almost achieved its energy intensity target under the 11th FYP, it did not come without difficulty, especially in 2010 when several provinces were still far short of their goals. Missing the assigned targets would negatively affect local officials' job evaluation and career promotion opportunities under the target accountability system. Under this pressure, some local governments adopted political intervention measures such as rationing power to industries, residential buildings, public lighting, and even hospitals in some cases. Such irrational blackouts and enforced power cuts disrupted industry production and people's lives and was quickly halted by the Central Government. This experience suggests that traditional administrative measures have been used to the full. To go further, stronger commitments on emissions and energy consumption shall require longer-term and sustainable mechanisms and additional policy measures, such as emission trading schemes that will be elaborated in chapter 5.

3.3. Scenarios of future CO_2 emissions in China

It can be predicted that, in the near future, China's CO_2 emissions will continue to rise rapidly along with its fast economic growth and social development. The key determinants that affect the trajectory of future emissions in China shall be the economic growth rate and the level of national efforts in lowering its energy intensity (Figure 1-5). Starting from a very low point of economic development, in 2010 China's per capita GDP was still less than half of the world average and far more behind the levels of the United States and the EU 27 (Table 1-3). The nation's development is also characterized by low energy efficiency or elevated energy intensity- 3 times higher than developed countries. In addition, due to high reliance on coal, energy use in China is more carbon intensive than in the US and the EU since the combustion of coal emits almost 30% more CO_2 than oil and over 50% than natural gas (BP, 2013).

Table 1-3 Comparisons of levels of CO_2 emission drivers in China and other countries (2010)

Factors Unit	CO_2 emissions /capita tCO_2	Total CO_2 emissions Mt	Population Million	GDP/capita in current US$	Energy use /GDP toe/$	CO_2 emissions /energy use tCO2/toe
World average	4.8	32900	6826	9307	231	2.8
US	18.3	5670	309	48358	153	2.5
EU 27	8.1	4057	502	**32074**	121	2.3
China	6.7	8896	1338	4433	394	3.8
China to reach EU27 living standards	48.1	64357	1338	**32074**	394	3.8
China to reach EU 27 levels	9.0	12045	1338	**32074**	**121**	**2.3**

Source: Author's calculation based on data from CAIT (2013) and WB (2013)

Assuming that China reaches the EU living standard but holding levels of energy intensity and carbon intensity of energy use fixed at 2010 levels, its CO_2 emissions would amount to 60,145 Mt, about 8 times the current emissions. However, if China manages to achieve the same level of energy efficiency and technological improvements as experienced in the EU, CO_2

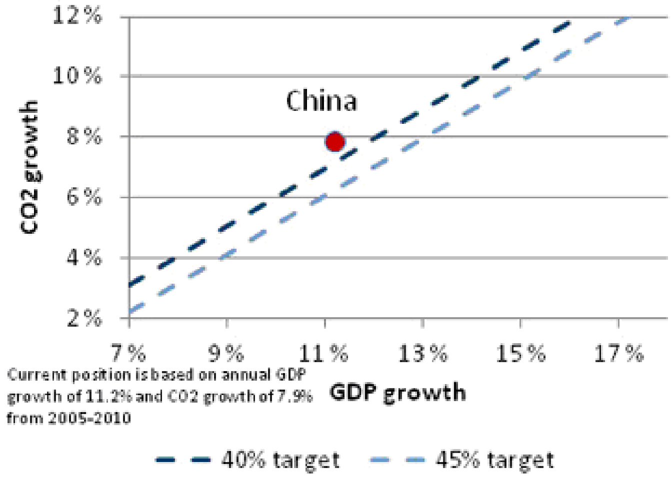
emissions would be 9,567 Mt, assuming a stable population and social development comparable to the EU. Even in this case, the emissions would be 35% higher than the current level. As coal is historically predominant in the energy supply in China, reducing the energy intensity will require substantial technological improvements and financial investment to develop clean coal and non-fossil fuels (in particular renewable energies).

Given the necessity for further economic and social development, China is reluctant to put an absolute cap on its CO_2 emissions in the near or middle term. Nevertheless, addressing the carbon intensity of the economy constitutes key elements of national mitigation strategies since such actions are more relevant to China's current development stage. According to the 'China Energy and CO_2 Emissions Report for 2050' (ERI, 2009), led by the government think-tank Energy Research Institute (ERI) of the NDRC, the growth of national GHG emissions will gradually slow down towards 2020 and is likely to peak around 2030 if appropriate policies are put in place. Such actions are needed since China has to cope with a limited resource constraint and its consequent environmental and health issues, now leading to growing public pressure.

3.4. China on track to meet the 2015 and 2020 climate targets?

Carbon intensity reductions are expected to bend China's emission curve in the next decade, although the rate at which total carbon emissions will continue to rise is largely dependent on the rate of GDP growth. Chai et al. (2011) plot China's emissions trajectory (Figure 1-8) and show that if China strictly follows the expected 7% annual GDP growth rate defined in the 12th FYP, CO_2 emissions growth should be limited to 3.1% on a yearly basis and as low as 2.3% respectively for the 40% and 45% reduction scenario. The red point in Figure 1-8 indicates that the current development path, with GDP growth at 11.2% and CO_2 emissions at 7.9% during 11th FYP will not allow China to meet the upper range of its climate ambitions.

Figure 1-8 China's position towards target reaching (2011-2020)



Source: Chai et al. , Point Carbon analysis (2011)

Chai et al. (2011) also examine the possibility of China reaching the target relying solely on cleaner energy sources, concluding that a 15% share of non-fossil fuels in the overall energy mix in 2020, combined with average GDP growth rate achieved in the 11th FYP period, will make the 40% target achievable, while meeting the 45% target will require substantial additional efforts.

Reports from the Climate Policy Initiative (2011) point out that it will be challenging for China to meet its climate and energy related targets. The 4 trillion RMB economic stimulus packages that the government initiated in 2008 in response to the global financial crisis maintained high economic growth rates and provided strong support to energy intensive industries. At the same time, many of the “low-hanging fruits” in energy efficiency savings have already been picked. For example, the replacement of small plants has nearly reached saturation and will offer very limited room for improvement in the next ten years. The marginal costs of energy conservation and emission reductions will continue to rise, making the targets under the 12th FYP more difficult to achieve. This shows the difficulty of an effective transition to a low carbon economy with mitigation strategies targeting solely energy-related emissions.

4. Conclusion

China’s current reluctance to make compulsory mitigation commitments in fear of constraining its economic growth builds on the grounds that cumulative CO_2 emissions in both the total and per capita terms are both low in China. Still, the rapid but carbon-intensive economic development put it as the world’s top annual GHG emitter and raised serious environmental issues. Tackling these challenges to shift it to a more sustainable and low-carbon development pathway requires the nation to decouple the GHG emissions from future GDP growth, the latter of which remains the national priority. China started to take bold actions against climate change, which centered on improving energy efficiency, developing renewable energy and increasing forest carbon stocks since 2010.

Analysis of Chinese climate policies reveals that reduction activities are mainly on CO_2 emissions related to energy use and industrial processes, which have been the centerpiece of its international commitments and specified targets in the 12th Five-Year Plan, and it is the catalyst to prompt the national’s experimentation with introducing seven pilot carbon emission trading systems. Meanwhile, intensive programs on forestation and improving forestry management were launched to increase the storage of atmospheric CO_2. However, as a major contributor to national GHG inventories that is only second to the energy sector and the primary source of CH_4 and N_2O emissions, agriculture has so far been merely mobilized under mitigation policies. This absence is owing to the difficulty in accurate measurement of agriculture emissions and the strategic priority given to food security. Integrating agriculture into the national mitigation strategies, however, could enhance the effectiveness of agricultural production if with possible solutions improve agricultural techniques to reduce emissions without harms to food security. This is the question tries to explore in the following chapters, focusing on crop production and examining the potential and condition of mitigating GHG emissions by using economic tools.

Chapter 2 - Agriculture emissions: past trends and construction of future baseline scenario

The previous chapter tells us that agriculture is merely considered in the current climate mitigation policies in China but this absence presents a missed opportunity given the magnitude of GHG emissions from this sector. Maximizing agriculture's contribution to climate mitigation efforts is essential to China's overall transition to a low-carbon economy from both the technical and cost-effective perspective. The first step towards putting in place a mitigation strategy for agriculture is to examine the characteristics of its GHG emissions, including the emission sources, quantification methods, past trajectory and driving forces. When projecting the future agriculture activities, an essential requirement that should be kept in mind is China's growing population for increasingly resource-intensive diets. All these elements are fed into the construction of a future baseline scenario, describing how the emissions shall evolve without additional abatement actions. The examination of BAU situation lays the foundation for further evaluation on technical and economic potential in order to provide reference for policy makers.

In this sense, this chapter will first brief China's agriculture development situations and the priorities attached to food security as well as major policies related to agriculture production (section 1). Section 2 will investigate the evolution and features of GHG emissions in Chinese agriculture. The priority on safeguarding national food security requires a closer analysis of GHG emissions from cropland and its relations with land productivity, i.e. the GHG intensity of crop production (section 3). Based on projected agriculture activities, section 4 will set up an emission baseline for Chinese agriculture to 2020, reflecting the climate impacts of a most likely BAU scenario (section 4). Section 5 concludes.

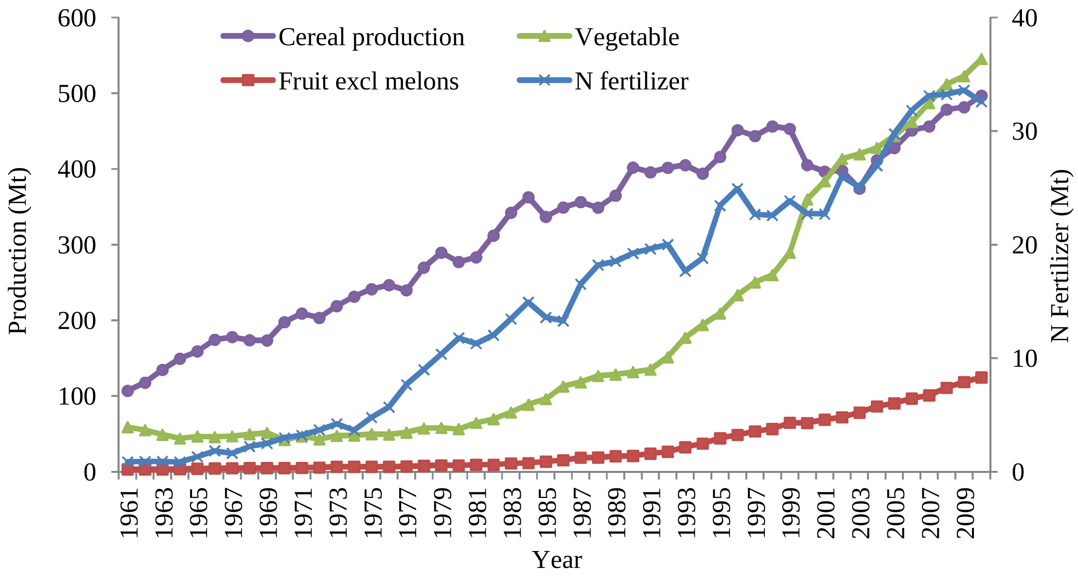
1. Chinese agriculture development and major policies

1.1. Rapid agriculture development and food security priority

Along the history, Chinese governments have been consistently attaching great importance to agriculture due to its fundamental role in nourishing the population, providing essential primary materials to other sectors of the economy, supporting rural employment, and raising farmer's income. The historic economic and social transformation of China in the past three decades as well as population explosion also resulted in robust growth in domestic demand for agricultural products. Accordingly, enhancing overall grain production capacity, sustaining food sufficiency and advancing rural development have been and will continue to be state policy priorities (NDRC, 2009) despite the fact that the proportion of GDP produced by agriculture declined from over 30% in the early 1960s to about 10% in 2010 (NBS, 2011).

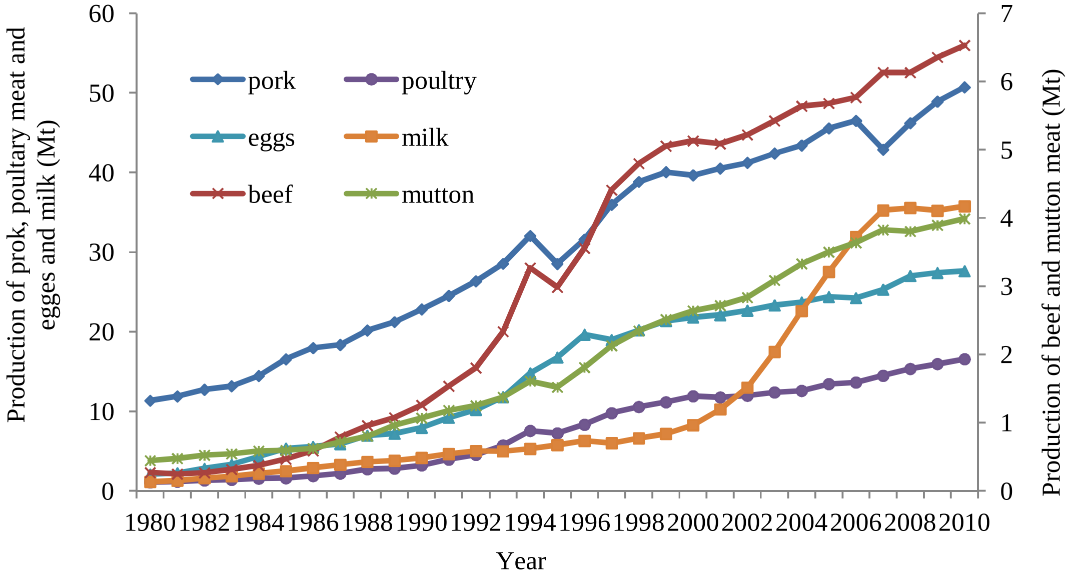
China has made substantial efforts to enhance national crop production to feed about 20% of the global population with only 8% of the world's arable land (World Bank, 2013). Figure 2-1 illustrates that, from 1961 to 2010, total cereal production has increased 4.6-fold from 107 to 497 Mt and crop yields have improved at almost the same pace (FAO, 2013). Over this period, vegetable and fruits (excluding melons) outputs have risen more rapidly than cereal production, by a factor of 9 and 38, respectively. However, the improvement in productivity was outpaced by the growth in N fertilizer inputs, which is responsible for over 70% of N_2O emissions from Chinese croplands (Gao et al., 2011). Cereal production (rice, wheat and maize) accounted for about 47% of chemical N fertilizer consumption in agriculture in the late 2000s (Heffer, 2009).

Figure 2-1Trend of crop production and N fertilizer inputs in China from 1960 to 2010



Source: FAO (2013) and N fertilizer from IFA (2013)

Figure 2-2 Trend of animal products in China from 1980 to 2009



Source: NBS (2011)

Growth in both crop production and animal products (Figure 2-1 and Figure 2-2) has particularly accelerated after the economic and rural reforms in the late 1970s. Measured by FAO's net agricultural production index, per capita agricultural output increased at a modest rate of 1.1% annually from 1961 to 1978 but grew at 3.8% per year from 1978-2011 (OECD-FAO, 2013). The annual growth of per capita livestock products (5.6%) was stronger than that of the crop production (2.9%) from 1980 onwards. After 2000, China experienced major changes in the consumption pattern with per capita direct consumption of grains declining while the demand for higher value food continually increasing, especially livestock products (Zhou et al., 2012). As a result of this soaring demand for foods of animal origin, livestock numbers have shown dramatic increase, originating large amount of CH_4 and N_2O emissions from livestock feeding and manure management. Meantime, this demand also accelerated cereal imports by China, especially soybean as animal feed. China has now become a net importer of rice, wheat, maize as well as soybean relying on imports for nearly 80% of its domestic soybean consumption relative to about 45% in 2000 (FAO, 2013).

This thesis chose not to pay much attention to the effects of land use change on emission evolution for two reasons. First, the focus of this study is to investigate the impacts of agricultural management activities rather than those of land use change. Secondly, land change information in Annex 3 indicates that there will be no significant change in crop land use.

It should be highlighted that from 1998 to 2003 the overall grain production declined by as much as 18% attributable to shrinking cropping area (a decrease of 12% in the 5 years and 71% contribution to production decrease), slightly declining yields, lack of rural labors due to more attractive jobs in urban areas, diminishing cultivation motivation of farmers because of sharp decline in grain market prices, backward agriculture infrastructure, and severe natural disasters (mainly drought) (Huang, 2004). The Chinese government soon reacted with a range of policies to benefit farmers: a reversal of its centuries-old practice of taxing agriculture to subsidy farmers. Starting in 2004, it eliminated production taxes on farmers, introduced the nationwide direct subsidies for farmers, and set minimum grain purchasing prices. Since then, agriculture has consistently been the subject of “No. 1 Document” issued each year by the Central Government that gives top priority to the goal of safeguarding food security, raising farmers’ income and accelerating rural development.

Although the proportion of rural population in China’s total population has decreased from 81% in 1980 to 50% in 2010 with an absolute decline of 125 million (NBS, 2011), the agriculture system is still dominated by small-scale farms that are responsible for the majority of national crop production. Large-scale state-hold farms occupied only 4.5% of Chinese croplands and provided 5.4% of national total grain products in 2010 (MOA, 1986-2013). Over 90% of the 200 million households in China have less than 1 hectare of cropland in size which itself is fragmented into 3 or 4 plots (Huang et al., 2012). Albeit the gradual increase of large-scale livestock production, small farms still play a major role in swine and dairy production. Another key challenge facing Chinese labor-intensive agriculture production system is the lack of labors in rural areas since young people are floating to cities for more economically attracting jobs.

Deep understanding of China’s food security challenge is crucial since it will condition the following research in the way that construction of baseline and mitigation scenarios as well as choosing mitigation measures should premise on no negative impacts on productivity. And extremely segmented farming system shall determine to a large extent the economic viability and implementation challenges of mitigation measures.

1.2. Current and near-future national agriculture policies

Since mitigation potential will be assessed against a projected BAU scenario, it is fundamental to determine an accurate baseline that reflects changing production environment and accounts for on-going structural change across agriculture. China’s agriculture now stands at a cross road and its performance over the next decade will be shaped by both the broad macroeconomic and demographic factors, but also impacted by the emerging challenges and relevant policy response.

China’s continuous GDP growth (though gradually slowing down) and the rapid increase in urban population despite of small growth in total population over the next ten years will sustain China’s continual demand for food, especially for feed grain and protein meal (OECD-FAO, 2013). Accordingly, although further reductions in per capita direct consumption of grains could be expected over time, increased indirect demand of grains used to produce animal feed will balance the overall grain consumption. But China is facing multiple and complicated challenges in sustaining national food supply given concerns over shrinking arable land, declining water and resource availability, increasing opportunity cost of rural labors and others. On top of these limiting factors, climate change has emerged as a significant threat to agriculture systems. In view of these increasing production constraints, growth in agriculture production is expected to slow down in 2011-2020 than in the first ten years of the 21st century. OECD-FAO (2013) predicts a growth rate of 1.7% per year during 2013-2022 against the 3.2% annual increase in the previous decade. Looking towards 2030, it is suggested that annual crop production should be increased to around 580 Mt assuming Chinese population stabilizes at around 1.6 billion and the dietary changes to high proportion of animal protein (Fan et al., 2012). The government set a goal of at least 545 Mt for national grain (rice, wheat, maize and soybean) production capacity in 2020, maintaining the domestic food self-sufficiency rate at 95% (NDRC, 2009). To meet this increasing demand on the limited arable land, grain yield in China must grow by at least 0.9% annually during 2011-2020.

In terms of climate related policies in the agricultural sector, the 12th FYP called for controlling agriculture GHG emissions. In response, the Ministry of Agriculture (MOA) has initiated programs to improve fertilizer use efficiency by 3% and enhance irrigation water use efficiency by 6% in 2015 from 2010. In addition, the government planned to bring an additional 11.3 Mha of croplands under conservation tillage systems during 2009-2015 in north China. These policies are not directly designed for the purpose of regulating GHG emissions in agriculture, but could have side effects on mitigation in agriculture.

2. Accounting agricultural emissions and past trends

2.1. Primary source of CH_4 and N_2O emissions

Globally, agriculture accounted for about 10-12% of the world's total GHG emissions excluding LULUCF in 2005 (IPCC, 2007). In China, agriculture emitted 820 $MtCO_2e$ in 2005, representing 11% of the national GHG emissions (Table 2-1). Agricultural emissions increased by 36% from the levels in 1994, when agriculture contributed 15% of the national total. Agriculture was responsible for over 70% of national N_2O emissions and approximately 50% of CH_4 emissions, arising from livestock enteric fermentation, croplands, rice cultivation and livestock manure management (Table 2-1). In both the 1994 and 2005 inventories, 310 and 21 were applied as the direct GWP of N_2O and CH_4, respectively, at the 100yr horizon.

From 1994 to 2005, livestock enteric fermentation has consistently been the largest source of agricultural GHG emissions. CH_4 is produced as a byproduct in the normal digestive processes of animals, in particular ruminant animals. Dairy cows, cattle, buffalo, sheep, goats and swine were identified as key emission sources. As the second contributor, cropland was responsible for over

70% of agricultural N_2O emissions which is produced naturally in soils through the microbial processes of nitrification (the aerobic microbial oxidation of ammonium to nitrate) and denitrification (the anaerobic microbial reduction of nitrate to nitrogen gas). Human-induced net N additions to soils (e.g., synthetic or organic fertilizers, crop residues) will trigger both direct and indirect N_2O emissions. The latter occur via either N deposition (associated with ammonia volatilization) or nitrate leaching and runoff. CH_4 produced by the anaerobic decomposition of organic matter in flooded rice fields accounted for around 20% of agricultural emissions. Livestock manure management generates both CH_4 – produced during the anaerobic decomposition of manure, and N_2O emissions – produced by the nitrification and denitrification of the organic N content in livestock manure and urine. The amount of emissions depends on the types of manure treatment or storage, the composition of the manure, climate conditions and other factors.

Table 2-1 GHG emission sources and contribution in Chinese agriculture

Sources	2005			1994			Increase
	N_2O Mt CO_2e	CH_4 Mt CO_2e	% of agri total GHG	N_2O Mt CO_2e	CH_4 Mt CO_2e	% of agri total GHG	% 2005 from 1994
Enteric fermentation		302.0	36.8%		213.8	35.3%	41%
Croplands N_2O	207.7		25.3%	194.7		32.2%	7%
Rice cultivation		166.5	20.3%		129.1	21.3%	29%
Livestock manure management	83.7	60.1	17.5%	13.6	18.2	5.3%	351%
Others (grazing, residue burning...)				35.7		5.9%	
Agriculture total	291.4	528.6		244.0	361.1		36%

Source: First and Second National GHG Emissions Inventories

The quantification of GHG emissions from agriculture was primarily based on the IPCC Guidelines involving the identification of key emission sources to be accounted for and the choice of estimation methods. Methodological choice defines the degree of precision of emission estimations in a sense that Tier 1 are simple methods with IPCC default values for emission factors and Tier 2 use country specific emission factors and detailed activity data if available while Tier 3 are more complex applying modelling or measurement approaches. IPCC Tier 1 and Tier 2 methods were generally employed in compiling national inventories for agriculture emissions. It is worth noting that the 2005 inventory is not completely comparable with the 1994 inventory for two main reasons. Firstly, accounting perimeters were different as emissions from grazing, residue burning as well as manure burning were no longer capped in the 2005 inventory. Secondly, emission factors have been gradually updated by the IPCC along with research advancements. For example, the default emission factor for N_2O emissions from N inputs to flooded rice and upland are distinct at 0.03% and 0.1%, respectively, in the 2006 Guidelines which is set at 1% both in the 1996 Guidelines.

As emissions are positively correlated with agriculture activity levels, the increase in agriculture GHG emissions was driven by the rapid increase in N inputs and animal products production (Figure 2-1and Figure 2-2). Changing diets to more animal products since the 1990s also triggered higher growth rate in meat, egg and milk production than grain, justifying the more evident increase in livestock related emissions (both fermentation and manure management) increases at a higher rate than cropland emissions. For example, CH_4 emissions from enteric fermentation raised by 41% from 1994 to 2005 compared to only a 7% increase in N_2O emissions from croplands.

2.2. Origin of important CO_2 emissions

In national inventories following IPCC classification, emissions attributed to the category 'Agriculture' contains only CH_4 and N_2O emitted within the perimeter of farm gates. It does not include emissions related to the use of fossil fuel in agricultural production, which is accounted in the category 'Energy'. Nor does it accounted for upstream emissions, such as the manufacture and transport of agriculture inputs and goods, and downstream emissions, such as the transport of food and feed products. In addition, carbon sequestration in croplands and above-ground biomass, which has not yet been reported in the Chinese inventories, is generally not reported in the 'Agriculture' category but classified under the 'LULUCF',

Some sources (SAIN, 2011) estimate that China's agriculture and agro-chemical industries together accounted for about 20% of China's total GHG emissions. It is reported that energy used for agriculture, forestry, animal husbandry and fisheries amounted to 60.8 Mt tce in 2007 (Lin et al., 2011), accounting for about 2.17% of China's total fossil energy consumption. This amount is equivalent to approximately 125 $MtCO_2e$ emissions in 2005 by referring to total emissions in the 'Energy' category (5,770 Mt CO_2e). In terms of emissions related to agro-chemicals manufacture and transportation, it is difficult to draw an exact figure constrained by data availability. Nevertheless, Zhang et al. (2013) reported emission factors for NH_3 synthesis, N fertilizer manufacture, N fertilizer transportation and distribution at 5.1, 0.9 and 0.1 tCO_2e/tN, respectively, implying that about 171 Mt CO_2e were emitted before N fertilizer being applied to lands in 2005. In the same vein, studies on European agriculture also indicates that agriculture-related energy use, upstream and downstream emissions represent nearly same amount of emissions as those reported in the 'Agriculture' category (Dequiet, 2012).

Another particular feature of agriculture lies in its ability to sequester atmospheric carbon into soils and aboveground biomass. But accounting for carbon sequestration raises several issues primarily because of the short-run and non-permanent characteristics. For example, carbon stored in agricultural soils could be released back into the atmosphere in case of switching from no tillage to conventional tillage. Albeit the exclusion from national inventories, soil organic carbon (SOC) content in the surface layer of Chinese croplands are found to steadily grow over the past

30 years in most regions triggered by the continuous increase in crop yields, conservation tillage area and return of crop residues (Pan et al., 2010; Yan et al., 2011; Yu et al., 2012). The increase rate was estimated at 13-28 TgC/yr or 28-44 $MtCO_2$/yr (Cai & Yan, 2011), implying that carbon accumulation in croplands was able to offset 3.4-5.4% GHG emissions from China's agriculture in 2005.

To keep consistent with the national inventory and highlight farmers' direct role in altering climate impacts of agriculture production, our research is centered on the CH_4 and N_2O emissions within the farm gate without targeting the broader lifecycle carbon footprint. In the meantime, improvement in SOC is also taken into account in evaluating the mitigation potential of certain measures owing to its importance in enhancing land fertility, raising crop yields and offering mitigation opportunity through carbon sequestration. However, more-broad or life-cycle analysis is worthwhile in future research given the general positive effects of mitigation measures in reducing upstream or downstream emissions.

3. Study on GHG emissions from cropland

3.1. General methodologies of accounting N_2O emissions from cropland

This PhD study chose to focus on the climate effects of cropland farming activities and management practices: N_2O emissions from cultivated croplands, CH_4 emissions from rice paddies and potential carbon sequestration by agricultural soils. N_2O emissions will be given particular attention. Such a choice was made based on available data and time constraints. Analysis of emissions and mitigation potential related to livestock production, manure management and grassland was carried out by another PhD candidature (Frank Koslowski) under the same research framework, which were combined with those under this PhD research to deliver a full picture for Chinese agriculture.

The first step to the construction of baseline scenarios for future emissions as well as the quantification of abatement potential of mitigation measures is to determine a robust emission estimation method. Ideally, such a method should be consistent with that used in the national inventories. This is the case for CH_4 emissions from rice paddies since historical and predicted emission data are available from peer-reviewed papers ((Zhang et al., 2011) using the CH4MOD model which was applied for compiling the national inventories. Regarding N_2O emissions from croplands, the IAP-N (Improving Anthropogenic Practices of managing reactive Nitrogen) model (Zheng et al., 2004) was employed to quantify direct N_2O emissions in compiling national inventories while IPCC default emission factors used for indirect N_2O emissions. However, it is unable to get access to this model to reproduce similar emission estimations. This raises the need to choose another method that is widely recognized but easy to operate.

Many scholars have quantified N_2O emissions from Chinese croplands applying two broad categories of methods: empirical or measurement formulae with representative emission factors (Yan et al., 2003; Zou et al., 2010; Gao et al., 2011) and complicated biogeochemical models (Li et al., 2003; Zheng et al., 2004;). It is worth noting that even those studies following IPCC Guidelines, results may vary depending on the selection of reported N sources, activity database and emission factors. A comparison of these methods leads to the choice of following the IPCC 2006 Guidelines combined with Chinese specific emission factors (Gao et al., 2011; Zhang et al., 2013; Table 2-2). This research takes into account both direct and indirect N_2O emissions from the three major N input sources-synthetic fertilizers, organic manure and crop residues, which are consistent with the sources accounted in the national GHG inventories. N_2O emissions can be released directly when anthropogenic N is added the soils or indirectly resulting from volatilisation and subsequent redeposition of NH_3 and NO_x and their products (NH_4^+ and NO_3^-) as well as leaching and runoff of N as NO_3^- from soils. General calculation is conducted following Eqn (2-1).

$$
\begin{aligned}
&Emissions_{N_2O} = N_2O - N_{input} \bullet EF = N_2O - N_{input} \bullet (EF_{direct} + EF_{indirect}) \\
&N_2O - N_{input} = F_{SN} + F_{AW} + F_{CR} \\
&EF_{direct} = EF_1 \bullet 44/28 \bullet GWP_{N_2O} \\
&EF_{indirect} = (Frac_{GAS} \bullet EF_4 + Frac_{LEACH} \bullet EF_5) \bullet 44/28 \bullet GWP_{N_2O}
\end{aligned}
\qquad (2\text{-}1)
$$

$Emissions_{N2O}$ is the N_2O emissions from croplands (tCO_2e). N_2O-N_{input} represents total N input (tN). F_{SN}, F_{AW}, F_{CR} represent N inputs from synthetic fertilizers, animal manure and crop residues, respectively (tN). EF_1, EF_4, EF_5 are the emission factors for N_2O emissions from N inputs, N volatilization, and N leaching and runoff, respectively (kg N_2O–N/kg N input). GWP_{N2O} is the direct GWP of N_2O at the 100yr horizon, 298. $Frac_{GAS}$ and $Frac_{LEACH}$ are fractions of N that are lost through atmospheric deposition of N volatilised and leaching or runoff. 44/28 is to convert the emissions from kg N_2O-N to kg N_2O gas. Refer to Table 2-2 for selection of EF_1, $Frac_{GAS}$, EF_4, $Frac_{LEACH}$ and EF_5 and subsequent results of EF_{direct} and $EF_{indirect}$.

Table 2-2 GHG emission factors for N inputs to China's croplands

Data sources	Crop systems	Direct N_2O*		Indirect N_2O†					Total EF
		EF_1 (%)	EF(tCO_2e tN^{-1})	$Frac_{GAS}$ (%)	EF_4 (%)	$Frac_{LEACH}$ (%)	EF_5	EF(tCO_2e tN^{-1})	(tCO_2e tN^{-1})
China specific	Rice paddy	0.41	1.92	17.9	0.01	1.4	0.0075	0.89	2.81
	Upland field	1.05	4.92	12.9	0.01	9.8	0.0075	0.95	5.87
IPCC default	Rice paddy	0.30	1.40	10.0	0.01	30.0	0.0075	1.52	2.93
	Upland field	1.00	4.68	10.0	0.01	30.0	0.0075	1.52	6.20

* Direct N_2O emission factors are from a study by Gao et al. (2012) based on 456 N_2O emission measurements in China.

† Indirect N_2O emission factors are obtained from Zhang et al. (2013) based on 397 N_2O emission measurements in China.

F_{AW} is estimated following Eqn (2-2).

$$F_{AW} = \sum_{T} N_T \bullet (1 - Frac_{Grazing(T)})\ Nex_T \bullet (1 - Frac_{Loss(T)})$$
$$Nex_T = N_{rate(T)} \bullet \frac{TAM_T}{1000} \bullet 365 \qquad (2\text{-}2)$$
$$N_T = Days\_alive_T \bullet \frac{N_{S(T)}}{365} \quad if \quad Days\_alive_T < 365$$

N_T is annual average population of livestock T. T denotes livestock category. $Frac_{Grazing(T)}$ is the fraction of grazing population of livestock T (%). Nex_T represents the annual N excretion for livestock category T (kgN/animal/yr). $Frac_{Loss(T)}$ represents the amount of managed manure nitrogen for livestock category T that is lost in the manure management system (%). $N_{rate(T)}$ denotes the default N excretion rate (kgN/(1000 kg animal mass/day)). TAM_T is the typical animal mass for livestock category T (kg/animal). $Days\_alive_T$ is the average breeding days before slaughter. $N_{S(T)}$ is the annual slaughtered number of livestock T in average (or use stock number if average breeding days exceed a complete year). Selected default values for parameters in Eqn (2-2) are summarized in Table A in Annex 1.

F_{CR} is estimated following Eqn (2-3).

$$\begin{aligned} F_{CR} &= \sum_{i} F_{CR\text{-}AG(i)} + F_{CR\text{-}BG(i)} \\ &= \sum_{i} Pdt_i \bullet R_{ST\text{-}GR(i)} \bullet N_i \bullet (R_{SR(i)} + R_{BG\text{-}AG(i)}) \end{aligned} \qquad (2\text{-}3)$$

$F_{CR\text{-}AG(i)}$ and $F_{CR\text{-}BG(i)}$ represent N input from aboveground and belowground crop residues (tN). i denotes the crop type. Pdt is the annual crop production (t). $R_{ST\text{-}GR}$ is the ratio of straw to grain in terms of dray matter. N is residue N content (g/kg). R_{SR} is the proportion of above-ground straw returned to land (%). $R_{BG\text{-}AG}$ is the ratio of below-ground residue weight to above-ground plant weight. Values of parameters in Eqn (1-3) are mainly obtained from Gao et al. (2011), summarized in Table B, Table C and Table D in Annex 1.

3.2. A case study of GHG intensity of cereal production

Justification of quantification

Firstly, cereal production accounted for about 47% of chemical N fertilizer consumption in agriculture (Heffer, 2009) in the 2000s and was thus the source of nearly half of cropland N_2O emissions. Including also rice CH_4 emissions, cereal production is responsible for about 2/3 GHG emissions from cropland. Secondly, as discussed in section 1.2, Chinese national policy aspirations for agricultural development have traditionally concentrated on food security objectives, with any convergence of production and climate objectives focusing mainly on increasing productivity. In the future, low carbon agriculture, which is characterized by high productivity, more efficient use of resources and low GHG emissions intensity, should be considered as a major component of sustainable development (Norse, 2012). The concept of GHG intensity (GHGI), expressed as the overall GHG emissions per unit of product or yield-scaled GHG emissions, is suggested as a useful metric to evaluate nitrogen use efficiency (NUE) and to help identify mitigation strategies (Chen et al., 2011; Venterea et al., 2011; Tubiello et al., 2012). Applying such an indicator can encourage better management practices resulting in higher crop production per area and reduced N losses and GHG emissions (van Groenigen et al., 2010).

In this context, the Global Research Alliance on Agricultural Greenhouse Gases was launched in December 2009 to help reduce the GHGI of agricultural production.

FAO (Tubiello et al. 2014) reported that over the period 1961-2010 the world average GHGI of rice decreased by 49% while that of wheat and maize increased by 45%, suggesting that effective mitigation strategies are needed to achieve sustainable intensification; i.e. ensuring that efficiency improvements can lead to reduced absolute emissions. Bonesmo et al. (2012) investigated the GHGI of 95 arable farms in Norway, showing that increased gross margins in grain and oilseed production could be achieved with decreasing GHGI. The GHGIs of cereal production on experimental sites were also quantified in China indicating that economic and climate benefits can be simultaneously achieved by some improved management practices (Shang et al. 2011; Huang et al. 2013; Ma et al. 2013).

But to date there is no synthetic estimate of current and historical GHGI of cereal production on a national, regional or provincial level in China. Such investigation will allow the abatement estimation of certain mitigation measures to be performed at the provincial level with better accuracy than a national-scale treatment and is thus crucial for indentifying efficient regional mitigation strategies and actions tailored to local agricultural production systems and management practices. These estimates can also form benchmark values or baseline emissions levels as a premise for integrating agriculture into any market-based approaches. The regional results will be presented in the case study section of chapter 3.

Methods and data sources

GHGI refers to the climatic impacts of agriculture practices in terms of per unit of production and is calculated by dividing total GWP-weighted emissions from cereal production by crop yields. Using agro-statistics data, here we provide estimates of GHGI for rice, wheat and maize production on a national scale from 1985 to 2010 at 5-year intervals. N_2O emissions are accounted for quantifying GHGI of wheat and maize production while both CH_4 and N_2O are considered for rice paddies following Eqn (2-4). Although indirect N_2O emissions via N deposition and nitrate leaching and runoff could be significant depending on the local conditions (e.g. Venterea et al. 2011; Maharjan et al. 2014), especially in cases where there is a high rate of

N application, they were not taken into account into this study due to high uncertainty. Estimation of N_2O emissions follow the general methodology described in section 3.1 but rest on a per hectare basis. Method adjustments are described in Annex 2.

$$GHGI = \frac{Flux_{N_2O(Direct)} + Flux_{CH_4(FR)}}{Yield}$$
$$Flux_{N_2O(Direct)} = \frac{Emissions_{N_2O(Direct)}}{CA} \qquad (2\text{-}4)$$

GHGI is the greenhouse gas intensity of crop production ($kgCO_2e/t$). $Flux_{N2O(Direct)}$ and $Flux_{CH4(FR)}$ represents the N_2O flux (from both upland and rice paddies) and CH_4 flux from rice paddies, respectively ($kgCO_2e/ha$). $Emissions_{N2O(Direct)}$ is the direct N_2O emissions from rice, wheat or maize fields ($kgCO_2e$). CA denotes relevant cropping area (kha). Refer to Annex 2 for detailed treatment of $Flux_{N2O(Direct)}$.

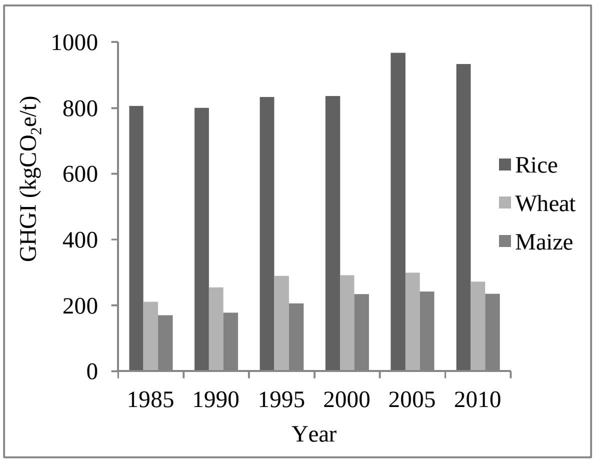
Agriculture activity data (cropping area, production, yield, and livestock number) were extracted from the China Rural Statistical Yearbooks (MOA, 1986-2013) and the China Livestock Yearbooks (MOA, 2001-2011). Per hectare N application rates for individual crops were collected from the China Agricultural Products Cost-Benefit Yearbooks (NDRC 2001-2012), which are the sum of N fertilizer (pure nutrient) and 30% N fraction in compound and mixed fertilizers(Sun & Huang, 2012). CH_4 fluxes of rice paddies were direct CH4MOD modeled results from studies by Zhang et al. (2011). The fraction of grazing cattle or sheep was the ratio of total grazing animals (the sum of livestock numbers in grazing areas and half-grazing areas) to the total stock number (MOA 2001-2011). The proportion of above-ground straw residues returned to land in 2006 was derived from results report by Gao et al. (2009). The nationwide ratio of straw retuned to land was reported at 15.2% in 1999 (Han et al. 2002) and rose to 24.3% in 2006 (Gao et al., 2009), implying an annual rate of increase of 6.93%. This rate was employed to estimate the percentage of straw recycled to farmland in target years (Table C and Table D in Annex 1).

Historical trends of national GHGI of cereal production

The national average GHGI of rice production in 2010 amounted to 933 $kgCO_2e/t$. In general, CH_4 made up about 90% of total GHG emissions and was therefore the dominant gas in determining the carbon footprint of rice cultivation. The national average GHGI of wheat and maize production in 2010 was 271 $kgCO_2e/t$ and 234 $kgCO_2e/t$, respectively. In general, synthetic N fertilizer made up at least 70% of total emissions and was therefore the primary emission contributor.

Figure 2-3 shows that the national GHGI of rice production evolved at a different way to those of wheat and maize production, and the latter has always been the least carbon intensive of the three crops. Rice GHGI saw little variation beween 1985 and 2000, which can be explained by nearly the same rate of growth in the CH_4 flux, yield as well as the N application rate over this period. However, when rice yield reached a periodic peak in 1998 the CH_4 flux continued to climb, resulting in a sharp rise in GHGI in the first decade of the 21^{st} century. Wheat and maize GHGIs had been steadily increasing from 1985 to 2000 since the growth rate of N application exceeded the rate of yield improvement. The GHGI began to stablize or even decrease after 2000 as the combined effects of increasing yields, abeit at a lower rate, and a stabilized synthetic N rate promoted by the national "Soil testing and fertilizer recommendation program" (MOA, 2005) initiated in 2005.

Figure 2-3 Historical changes of national average GHGI of rice, wheat and maize production



Source: Results calculated by the author and incorporated in Wang et al. (2015)

4. Building a baseline for future GHG emissions from Chinese agriculture

This section will present the forecast of future agriculture activities (section 4.1) including particularly the use of synthetic N fertilizers in croplands (section 4.2), based on which the baseline emissions are projected for the agricultural sector in 2020 (section 4.3).

4.1. Projection of future agriculture activities

There have been several attempts to project China's near-term food production, including USDA's annual Agricultural Projections, OECD -FAO agricultural outlook (2013), and FAPRI's US and world agricultural outlook, and results of the China's Agricultural Policy Simulation Model (CAPSiM). The results of the CAPSiM model ware chosen to build the baseline emissions scenario in this research since this model analyzes the impacts of policy changes and other external factors on China's agricultural production, consumption, prices and trade (IAASTD, 2009) and therefore provides the most comprehensive and robust predictions of future agricultural activities in China. Model output was provided by the Center for Chinese Agricultural Policy of the Chinese Academy of Sciences.

Table E in Annex 3 presents the past and predicted cropping area, production and yield as well as prices of major crops in 2020 based on CAPSiM model forecast. The projected total cereal production of 540 Mt is in line with the 545 Mt national objective and the average yield growth rate of 0.9% per year is consistent with the expected productivity enhancement. The OECD-FAO, FAPRI and USDA projections have also concluded similar trends: rice production is going to decline and wheat production will stay relatively stable while maize production and cropping area will see tangible expansion due to the strong demand for maize as animal feed.

Predicted livestock numbers from 2011 onwards (Table F) are calculated using relevant product (meet, milk, eggs) growth rates revealed by the CAPSiM model assuming per head production remain constant to 2020 as in 2010 which were also the case in the past decade. A comparative growth rate in animal products is predicted for 2010-2020 as during the previous decade. In general, the CAPSiM output projects higher growth rates in animal products than estimates by OECD-FAO, FAPRI and USDA.

4.2. Projection of N consumption by crop production

Other key factors in setting up the BAU emissions scenario include the forecast of total synthetic N fertilizer use in agriculture and per hectare N rate of various crops. Projecting the overall agriculture N fertilizer consumption is challenging because of the significant differences among various databases as well as difficulties in determining a reasonable growth rate for the target period. For example, China Nitrogen Fertilizer Industry Association reported a total of 28.1 Mt N used in agriculture in 2010 (Zhang et al., 2013), but National Agricultural Yearbook pointed to 28.9 Mt assuming N fraction of 30% in the compound and mixed fertilizers while FAO reported 35.1 Mt and IFADATA 32.6 Mt in the same year. In this study, the IFA data was employed since summing up the N use of each crop (per hectare N rates multiplied by cropping area) is closest to 32.6 Mt. In the future, the demand for agricultural N fertilizer in China will continue to grow but will slow down at an annual rate of 1-2% (FDCNCIC, 2011; Good & Beatty, 2011; Zhang et al., 2013) compared to over a 2% increase each year from 2000 to 2010. At a conservative 1% annual increment over 2010–2020 China’s demand for N fertilizer in agriculture would reach 37 MtN by 2020 (Table 2-3).

Per hectare N application rates of various crops were projected based on linear extrapolation of historical N rates collected from the China Agricultural Products Cost-Benefit Yearbooks (NDRC, 1998-2013) (Table 2-3), but growth rates for rice, wheat, maize, greenhouse vegetable, openfield vegetable and fruit from 2010 to 2020 were assumed less than half of those during 2005-2010. These results are reasonable since above analyis on historical evolution of GHGI of cereal production indicate the stablizing or decreasing trend of GHGI - yield improvement should outpace the additional N inputs into croplands. In addition, the BAU estimates accommodates the national target of improving fertilizer use efficiency by 3% since crop-wide PFP_N rises to 45.7 in 2015 and 47 in 2020 from 40.9 in 2010, and are therefore sound from the political perspective.

Table 2-3 Total N fertilizer use in agriculture and national average application rate

	2005	2010	2015	2020				
National total N fertilizer use (kt)	29,761	32,599	35,172	36,967				
	N fertilizer rate(kg/ha)				% of total N consumption			
	2005*	2010*	2015†	2020†	2005*	2010*	2015†	2020†
Rice	190	187	182	177	18.4%	17.2%	13.8%	12.3%
Wheat	189	209	219	238	14.5%	15.6%	14.2%	14.2%
Maize	186	208	211	221	16.5%	20.7%	20.4%	21.2%
Soybean	49	54	53	53	1.6%	1.4%	1.3%	1.2%
Cotton	235	246	237	237	4.0%	3.7%	3.4%	3.3%
Oils	116	125	123	123	5.6%	5.3%	5.0%	4.9%
Sugar	256	347	322	322	1.3%	2.0%	1.6%	1.6%
Total vegetable	298	368	335	336	17.7%	21.5%	18.2%	17.3%
Greenhouse vegetable‡	581	719	655	656				
Openfield vegetable‡	232	288	262	262				
Fruit§	357	492	507	565	11.4%	16.5%	15.9%	16.8%

* N fertilizer application rates of different crops were collected from the China Agricultural Products Cost-Benefit Yearbooks (NDRC, 1998-2013), and we adopted N fraction of 30% in the reported compound and mixed fertilizers (Sun and Huang, 2012).

† Extrapolation of future N fertilizer rates were based on 2005-2011 data for rice, wheat and maize，1998-2011 data for fruits and vegetables, and average of 2006-2011 data for other crops.

‡ According to survey results (Chadwick et al., 2013; Zhang et al., 2013), N application rate for greenhouse vegetables is generally about 2-3 times as that for openfield vegetables (here we assume 2.5 times).

§ Due to lack of data for other fruits, we used average fertilizer rate of apple, mandarin and orange to represent general fruits.

Source: Author's calculation

Pertinent projection of per hectare synthetic N application is vital since it not only determines the trajectory of baseline emissions from cropland activities, but also constitutes the starting point for quantifying the abatement potential of cropland mitigation measures. Out results (Table 2-3) show that N rates for rice and vegetable production will decrease from 2010 onwards, but those for wheat, maize and fruit production will continue to grow, albeit at a less significant rate. For comparison, several field surveys and the IFA publication (Heffer, 2009) reported similar results on the N application rates for different crops. For example, a large-scale survey (Zhang et al., 2013) conducted in 2009 reported N application rates at 209, 197, 231, 383 and 550 for rice, wheat, maize, vegetable and fruits, respectively, which slightly exceed out results. In short, our baseline as well as mitigation estimations could satisfy the requirement of being conservative to avoid overestimation.

4.3. Results of business as usual scenario emissions from agriculture

Forecasting N_2O emissions from Chinese cropland followed the general methodology described in section 3.1combining the emission factors in Table 2-2 and N use predicts in section 4.2. CH_4 emissions from rice paddies were directly cited from prediction results using CH4MOD model (Zhang et al., 2011) adjusted for rice cropping area in 2020. For reference, the approach for estimating emissions from the livestock sector is described in Annex 4.

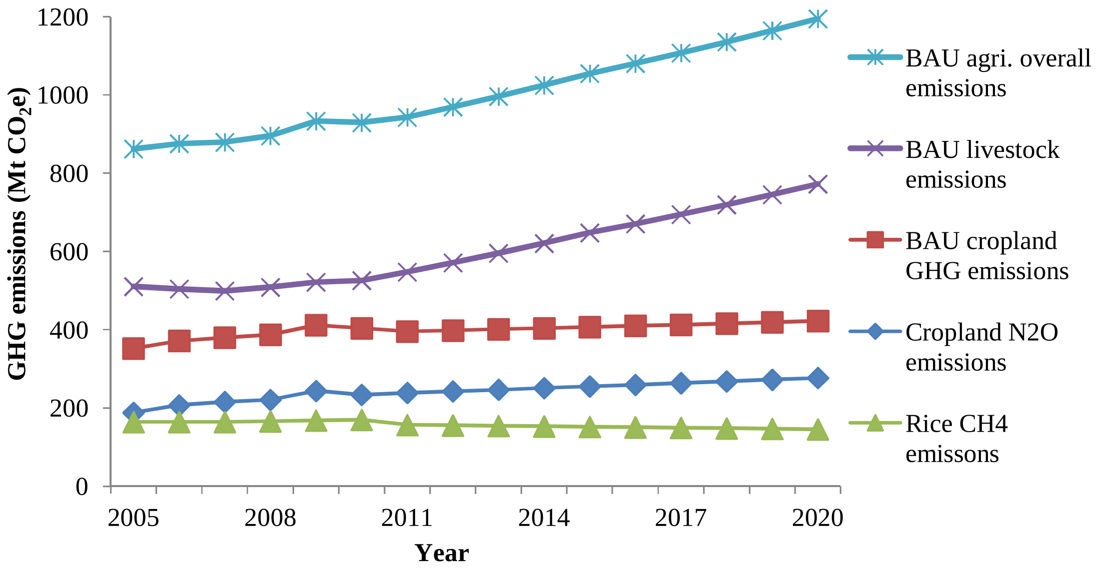
The integrated results of both cropland and livestock sectors are illustrated in Figure 2-4. GHG emissions from Chinese agriculture under the baseline will continue to increase from both crop fields and livestock in the near decade to reach 1195 $MtCO_2e$ in 2020, a 29% growth from 2010 levels. This increase is mainly driven by growth in livestock-related emissions (47% increase from 2010 to 2020). Cropland GHG emissions are predicted to be 422 Mt CO_2e in 2020, which is 4.7% higher than the 2010 levels. Such trends correspond to the ongoing change in peoples' diets for more animal and dairy products.

Within the cropland sector, N_2O emissions see a significant growth by 18.5% between 2010 and 2020 resulted from increasing synthetic N fertilizer application while a declining trend is observed for CH_4 emissions from rice paddies due to improved water regimes.

The Second National GHG Inventory reported 208 $MtCO_2e$ emissions in 2005 from cropland (N_2O) and 143 $MtCO_2e$ from rice paddies (CH_4), excluding CH_4 emissions from winter-flooded paddy fields, using 310 and 21 as the GWP of N_2O and CH_4 (NCCC, 2012). Our estimates of 188 $MtCO_2e$ N_2O emissions and 164 $MtCO_2e$ CH_4 emissions are therefore comparable to these figure and the differences are due to different GWPs used.

Defining a robust BAU scenario is crucial since it is the basis for evaluating the overall technical and economic mitigation potentials that could be mobilized from the Chinese agriculture. This BAU scenario provides a broad framework under which a specific baseline will also be defined for each mitigation measure against which the abatement potential and subsequent implementation cost is quantified. This study adopted a dynamic baseline anticipating future changes in agriculture production while a stationary baseline could also be used. The French MACC study adopted a static baseline referring to emissions of the 2010 although the mitigation potential is evaluated to 2030 since no projection is available encompassing the elements required and relying on existing data has the advantage of minimizing research uncertainty (Pellerin et al., 2013). In other studies, baseline scenarios are mainly constructed on projected level of agriculture production, e.g. the UK MACC exercise did not only integrate general agricultural policy commitments already put in place, but also accommodated assumptions about some policy reforms under discussion (Moran et al., 2011).

Figure 2-4 Baseline GHG emissions from Chinese agriculture to 2020



Note: BAU agriculture overall emissions = BAU livestock emissions+ BAU cropland GHG emissions; BAU cropland GHG emissions = Cropland N_2O emissions + Rice CH_4 emissions

Source: Author's calculation

5. Conclusion

In general, the derivation of an accurate baseline is important to the whole exercise of defining an efficient mitigation target. And in the case of this study, the mitigation potential and implementation cost of each measure needs to be quantified relative to a projected level of agricultural production activities and practices, the choice of baseline is therefore crucial to the robustness of the PhD research outcome. Considering the changeable agriculture production environment in China, a dynamic baseline is adopted in this study, but may lead to overestimation or underestimation of the actual abatement potential as well as the baseline emissions.

This chapter was therefore structured around the key objective of building a robust baseline for future GHG emissions from the Chinese agriculture under the BAU situation. As such, the core tasks are to identify an emission calculation convention and to forecast of future agriculture activities that reflect current agriculture development trends and incorporate government's core policies, among which national food security is undoubtedly the top priority. This supply prioritized policy leads to the continuous increase in agriculture GHG emissions in China by 29%

in 2020 relative to the 2010 levels, primarily more from the livestock sector (emission growth by 47%) than from crop production (emission growth by 4.7%). The baseline emissions are quantified based on an investigation of emission sources and estimation approach adopted in the national inventories, particularly focusing on those for cropland N_2O emissions and rice paddy CH_4 emissions in this PhD research. A special attention is given to the GHG intensity of cereal production justified by the double challenges to minimize agriculture's climate impacts and maintain land productivities in China. GHGI evolution showed stabilization or even decreasing trends at the national scale.

Having a baseline emission scenario built for agriculture that would have prevailed without mitigation incentives or additional abatement measures, we would now like to question if there are opportunities to slow down the emissions growth or even bend the emissions curve downward without negative effects on productivity. And, which are the possible measures that could contribute to the mitigation efforts and their relevant abatement effectiveness? Responding to these questions leads us to the investigation of the technical mitigation potential from Chinese agriculture in chapter 3.

Chapter 3 - Technical mitigation potential in China's croplands

Chapter 1 and chapter 2 illustrate the importance of agriculture GHG emissions in China and the significance of integrating agriculture into the national mitigation strategies. In fact, agriculture itself is part of the solution to tackle global warming since it offers substantial technical potential to mitigate climate change through both emissions reduction and carbon sequestration in terrestrial ecosystems (e.g. Oenema et al., 2001; IPCC, 2007; Smith et al., 2008, 2013). Technically feasible GHG mitigation measures applicable in both arable and livestock systems can be broadly grouped into five categories: increased NUE in croplands to avoid excessive N_2O emissions, better management to limit CH_4 emissions from livestock rumen and rice paddy, sequestering C into cultivated and grassland soils, and energy efficiency to reduce CO_2 emissions.

Using a bottom-up approach to estimate the mitigation potential in agriculture, this chapter will first screen the mitigation measures applicable to the Chinese agriculture conditions and examine their effectiveness in reducing GHG emissions and/or enhancing carbon sequestration, i.e. the per area potential of each measure to mitigation GHG emissions beyond the baseline activities identified in chapter 2 (section 1). The results on per hectare abatement rates will be combined with the additional area available for measure implementation to conclude the mitigation potential of each measure, which will be aggregated to generate the value for the whole sector (section 2). Akin to the analysis of GHGI of cereal production in chapter 2, a special focus is given to N-use related measures in cereal production and their regional variations are analyzed (section 3). Section 4 draws a conclusion and fosters next-step research.

1. Mitigation measures and abatement rates

1.1. Selection of mitigation measures

A review of the literature (Oenema et al., 2001; King et al., 2004; IPCC, 2007; Beach et al., 2008; Smith et al., 2008; Moran et al., 2011; US EPA, 2013) reveals that there is an extensive list of technically possible measures for mitigating GHG emissions from agriculture production activities. Based on these international as well as other national studies, an initial list (Table 3-1) of 16 cropland abatement measures was drawn up that appear to be applicable to China's agricultural and land use conditions. It should be noted that Table 3-1 does not provide an exhaustive list of abatement measures and there exist other options that may have potential to mitigate GHG emissions but were not taken into account to give focus on major mitigation opportunities arising from croplands.

Table 3-1 Initial list of crops/soils measures and reasons for inclusion/exclusion

Measure	Brief description	Include?	Exclusion reason
Avoid excessive use of synthetic N fertilizer	Reduce gross N rates since over-application of N fertilizers above agronomically sound and environmentally sensible recommendations is common in China.	Yes	
Improved timing of N application	Adjust N fertilizer application timing – less at sowing and planting stage, more during growth season to achieve a better match in nutrient demand and supply.	Yes	
Subsurface placement of N fertilizers	Use appropriate machinery to deep place fertilizers instead of surface application to decrease ammonia loss and increase NUE.	Yes	
Replace part of ammonium-based fertilizers with nitrate-based fertilizers when appropriate	In places where denitrification dominants N_2O generation can help minimize N_2O emissions and ammonia loss.	Yes	
Further reduce N fertilizers	Reduce N rate below recommendation level to trigger more emission reductions.	No	Negatively affect food production targets.
Improved irrigation systems in uplands	Promote fertiligation (e.g. drip irrigation, spark irrigation together with soluble fertilizers) to save cost and avoid emissions.	Yes	
More efficient recycling of organic manure	Increase animal manure amendment to soils to replace part of synthetic N fertilizers.	Yes	
Conservation tillage	Reduce tillage and soil disturbance to a minimum extent.	Yes	But not applicable to rice paddies.
Straw residues retention in lands	Returning crop residues back to croplands instead of in-situ burning or moving-out.	Yes	But not applicable to rice paddies.
Biochar addition	Biochar application decreases N_2O emissions and improves soil quality.	Yes	
Addition of nitrification or urease inhibitors in N fertilizers	Inhibits the process of nitrification to reduce nitrate leaching and N_2O emissions,	Yes	

Table 3-1 Initial list of crops/soils measures and reasons for inclusion/exclusion (continued)

Measure	Brief description	Include?	Exclusion reason
Controlled- and slow-release fertilizers	Fertilizers physically altered (e.g. coated urea) or chemically amended with additives that reduce the transformation rate of fertilizer compounds, resulting in an extended time of nutrient availability in the soil and therefore better match with crops' demand for nutrients.	Yes	
Improved water regime in rice paddies	Practice intermittent irrigation which will cause anaerobic and aerobic cycling to significantly reduce CH_4 emissions during rice growing season.	Yes	
Integrated rice-duck/fish farming system	Kind of organic farming methods that use ducks' movement to control the plant disease, pests and weeds and increase both the production and income. Studies also show that N_2O and CH_4 emissions can be largely decreased.	No	Limited application.
Increase cover crops (including leguminous plants)	Use cover crops to slow soil and water erosion, improve soil quality and enhance nutrient and moisture availability,	No	Limited additional application.
Improved crop varieties	Genetically enhancing the effectiveness of nutrient utilization by plants to enable the reduction of fertilizers.	No	Complex to analyze the effects.

Source: Author

Selecting appropriate abatement options is the first and essential step to investigate the overall mitigation potential from the crops and soils sub-sector. This exercise is challenging because of the large number of possible abatement measures and the fact that the effectiveness of some measures depend on interactions with others. To address these issues, the range of measures in Table 3-1 should be reduced to a manageable number for further analysis and sub-sector measures need to be consolidated to highlight major mitigation effects. Measure screening and consolidation was carried out based on the following criteria and principles.

(a) Acceptance. Measures likely to harm yields were excluded to be consistent with the national food security priority. For example, further reduction of N rates below the recommended level for optimum productivity will not be acceptable for its adverse effects on food production.

(b) Applicability. Measures with limited applicability at the national level due to technical, political or social barriers, were eliminated, e.g. rice-duck/fish integrated farming systems.

(c) Mitigation effects. Measures currently being practiced but increasing overall GHG emissions were removed, e.g. net GWP-weighted emissions from direct straw return to rice paddies tend to be positive since the increased CH_4 emissions triggered by additional crop residue exceed the carbon sequestration gains in paddy fields.

(d) Needs for consolidation. Some detailed sub-sector measures or technically similar measures were aggregated to account for measure interactions, e.g. water regimes should generally be coupled with fertilizer management practices in rice paddies. Another example is the consolidation of techniques improving N application time, method and products under the broad category of best N fertilizer management practices, but distinguished between cereal crops and cash crops.

(e) Current availability. Measures that are still in the early research state and technically complex to distinct the absolute effects on emission avoidance: e.g. improving crop varieties with higher NUE.

Following the eligibility criteria and consolidation principles, we identified 9 mitigation measures in the arable land sector for in-depth investigation. Measure descriptions and target crops are presented in Table 3-2. In parallel, our research team also carried out similar work for the livestock and grassland sector; selected measures are described in Table G in Annex 4. It is worth noting that Table 3-2 should not be considered as operational guidelines for measure implementation due to the biological complexity and regional variations of agricultural systems; rather we intend to set clear objectives to be achieved and point out which actions could be potentially carried out. For example, we suggest two kinds of sub-actions to achieve an optimal NUE in wheat and maize production: applying a greater proportion of N fertilizers at later growth stages and fertilizer deep placement. In practice, the two kinds of sub-actions could be adopted independently or in combination and detailed implementation techniques should be fine-tuned to accommodate local circumstances.

Table 3-2 Description of selected crops/soils mitigation measures and target crops

No.	Measure	Explanations	Target crops
C1	Fertilizer best management practices - Right rate	Reduce gross overuse of N fertilizers amount. We set regional optimal PFP_N* (Partial Factor Productivity of N fertilizer) derived from scientific fertilization recommendations (Zhang et al., 2009) as the indicator for fertilizer efficiency improvement objectives. This measure calls for a direct reduction in N fertilizer use for certain crops in targeted provinces to raise regional PFP_N to 70% of the optimal levels (Table 3-6).	Rice, wheat, maize, vegetable, fruit
C2	Fertilizer best management practices (Wheat &Maize) - Right time and right placement	This strategy suggests postponing N fertilizer to a later stage of wheat and maize growth with preferably two top-dressings compared to the current one top-dressing practice, and popularizing fertilizer deep placement by using appropriate machines for maize top-dressing, in a bid to reach optimal PFP_N (or optimum N management) by increasing yield and reducing N losses and further decreasing N rate (Table 3-6).	Wheat, maize
C3	Fertilizer and water best management in rice paddies	Split the total amount of N fertilizers into at least three applications for basal fertilization, early tillering, panicle initiation and heading stages; and shift from mid-season drainage (F-D-F) to intermittent irrigation (F-D-F-M) that accelerates anaerobic–aerobic cycling.	Rice
C4	Fertilizer best management practices (cash crops) - Right products, right time and right placement	Promote fertiligation (e.g. drip irrigation together with soluble fertilizers) for vegetables and cotton to save both fertilizer and irrigation inputs. As to fruits, controlling N rate and adjusting fertilization periods are essential to achieve sustainable fruit production. In addition, replacing part of ammonium-based fertilizers with nitrate-based products can also contribute to minimizing N_2O emissions and enhancing productivity.	Cotton, vegetable, fruit
C5	Enhanced-efficiency fertilizers	Use fertilizers added with nitrification inhibitors (NI) and/or urease inhibitors (UI) and slow- and controlled-fertilizers to reduce N_2O emissions.	All crops, vegetable, fruit
C6	More efficient recycling of organic manure	The general objective is to increase animal manure amendment to soils to supply 30% of crop N nutrients demand and 50% of vegetables and fruit. Efficient recycling of animal manure should be in form of composed manure or biodigester residues to replace part of synthetic N fertilizers.	All crops, open field vegetable, fruit
C7	Conservation tillage for upland crops	Conservation tillage (CT) is a series of agricultural practices aiming to reduce tillage and soil disturbance to a minimum extent with at least 30% of residues incorporated into soil to increase soil carbon content in upland cropping systems.	Wheat, maize
C8	Straw return in upland crops	Returning straw or residue back to field is considered a stand-alone farming practice in China which only involves changes in straw management compared with CT measure. This technique is an important way to improve soil fertility and soil physical properties if properly tailored to different cropping systems and local farming practices.	Wheat, maize
C9	Biochar addition	Application of biochar produced with crop straw pyrolysis can significantly decrease N_2O emissions and improve soil prosperities to enhance yields.	Rice, wheat, maize

* PFP_N -Partial Factor Productivity of N fertilizer is an indicator of NUE, measured by the grain yield per N input (kg/kgN)

1.2. Estimation of abatement rates of mitigation measures

Estimation methods using meta-analysis

A measure's abatement rate is defined as the per area emission savings and/or C sequestration amounts achieved by implementing the measure compared to a conventional practice under the baseline and is expressed as $tCO_2e/ha/yr$.

Existing research, both at the global and national scale, has examined and quantified technical abatement potentials for some agriculture mitigation measures (Lin et al., 2005; IPCC 2007b; Smith et al., 2008; Lu et al., 2009; Huang and Tang, 2010; Moran et al., 2011; US EPA, 2013). However, mitigation results concluded from these studies are not sufficient to construct a rigorous abatement scenario specifically dedicated to the Chinese agriculture for two reasons. Firstly, international research on the global scale could not fully reflect the specificities of Chinese agriculture systems and soil and management conditions. These specificities include the predominance of much smaller farms, higher reliance on human labors for crop production and in general lower soil fertility than in developed countries, where most of the international assessments are conducted.

Secondly, mitigation measures identified in our study are not completely identical to those included in these literatures. This raises the need for a comprehensive national-scale analysis which should be based on findings from a sufficient number of domestic studies to generate reliable weighted average value for each measure. This is possible since many experimental trails on mitigation effects have been carried out across China and therefore enabled a meta-analysis exercise to be performed. Meta-analysis is the use of statistical methods to combine results of independent studies and allows us to make the best use of all the information gathered.

Based on data from over 400 experimental studies in China, our research partner led a weighted meta-analysis exercise (Nayak et al., 2014) using MetaWin software (Rosenberg, 2000), under which mean effect size was calculated with 95% confidence interval. If crop-specific data was available, technical abatement potential was evaluated for each crop targeted by a mitigation measure, which were then weighted to derive the average abatement rate for the mitigation measure. Two assumptions and clarifications are needed to understand the meta-analysis results and subsequent analysis. Firstly, only emissions within the farm gate were captured in the whole study and wider life-cycle impacts of the measures were not within the scope of this exercise. For example, the abatement results do not include changes in CO_2 emissions generated from the manufacture and transport of fertilizers or on-farm energy use. Secondly, since mitigation measures adopted to reduce emissions of one GHG can sometimes result in corresponding changes in emissions of a non-target gas or SOC content, we evaluated the collective effects on SOC and N_2O and CH_4 emissions of introducing each abatement measure against the controlled treatments in experiment design. The abatement rate was expressed therefore as GWP-weighted tCO_2e/ha. We should also note that most of the sequestration measures are estimated to be effective for 15 to 30 years since the carbon sinks would saturate after a period of time (Sundermeier et al., 2011). This issue was not addressed given the short time span of 2010-2020 of this study, but should be taken into account for longer time horizon research.

Adjustment of meta-analysis results

Data used for meta-analysis were extracted from field experiment studies which are purposely designed to test the effects of mitigation action(s) against the controlled group. However, the controlled treatments don't necessarily represent common farming practices or baseline conditions identified in section 4 of chapter 2, adjustments of original meta-analysis results are therefore needed to better accommodate actual situations and partially internalize measure interactions (more detail in section 1.3).

Since agriculture systems and land use conditions are atomistic, heterogeneous and regionally diverse, the differences of measure abatement effects among different regions may introduce errors into the potential assessment. Consequently, in theory the smaller the geographic units the research is based on, the more accurate the mitigation results would be. However, data availability only enables the abatement rates of measures C1 and C2 for rice, wheat and maize to be quantified on the provincial level, while those of other measures were generally countrywide estimates.

For measures C1, C2, C3 and C4 which exclusively or partially target synthetic N fertilizer use, the N_2O abatement potential stemming from direct N cut was estimated based on the relationship between N fertilizer reduction and N_2O emissions reduction drawn from site experiments (Figure 3-4). Potential estimation of measure C1-C4 will be thoroughly elaborated in section 3. Drip-irrigation has been proven to be a prominent technology in improving cotton yields and reducing fertilizer and irrigation inputs, and was therefore considered the dominant mitigation technique in cotton production. Since both high-efficiency irrigation systems and replacement of ammonia-based fertilizers with nitrate-based fertilizers are able to lower N_2O emissions by at least 50% (SAIN, 2012) with same level of N rates, i.e. halving emission factors, this part of mitigation potential was also quantified in addition to emission reduction related to direct N rate decrease.

Measure C3 integrates fertilizer and water best management practices in rice paddies and therefore internal interaction between reducing N application rate and improving water regimes needs to be addressed. It is generally recognized that anaerobic–aerobic cycling or intermittent irrigation can stimulate N_2O emission from paddy field (Huang et al., 2007; Zou et al., 2007; Yang et al., 2012) while synthetic N rate is not considered a major factor affecting rice CH_4

emissions (IPCC, 2006). Indeed, meta-analysis results indicate that shifting from mid-season drainage (F-D-F) to intermittent irrigation (F-D-F-M) regime avoids CH_4 emissions by 1.781 CO_2e/ha but increases N_2O emissions by 0.525 CO_2e/ha, resulting in an overall potential of 1.256 CO_2e/ha. As the effects of water regime change on N_2O emissions were already taken into account in meta-analysis exercise, the abatement rate of measure C3 (1.337 CO_2e/ha) can be approximated as the sum of potential from individual actions – water regime improvement (1.256 CO_2e/ha) and reduced N fertilizer rate (0.081 CO_2e/ha).

Owing to limited dataset for a China specific meta-analysis, estimates of abatement rate from using enhanced efficiency fertilizers (measure C5) were based on the global meta-analysis results (Akiyama et al., 2010) suggesting addition of Nitrogen Inhibitors (NIs) can reduce N_2O by 34% in upland fields and 30% in rice paddies on average, compared with those of conventional fertilizers.

Meta-analysis results indicate that combined application of organic manure with chemical N fertilizer (measure C6) is able to sequester 1.435 tCO_2e per hectare per year and but increases N_2O emissions by 75% in uplands compared to application of chemical fertilizer alone. In case of rice paddies, CH_4 emissions also increase with the addition of organic manure and the overall abatement rate stands at 0.842 tCO_2e/ha/yr. These meta-analysis results were discounted because in practice organic manure has already been applied to croplands opposed to the zero organic manure arrangement under controlled experiments. According to Zhang et al. (2013) and Huang & Tang (2010), organic manure currently supplied about 9%-12% of total N input for grain crops. Chadwick et al. (2013) indicated that for greenhouse vegetables >50% of the N nutrients supply came from organic manures, and for open field vegetables and fruit manure supplied ca. 33% and 20% of the total N nutrients, respectively. Typical fertilization recommendations suggest organic manure providing 30% of N nutrients to crops and 50% to fruits and vegetables. The gap between optimal use and baseline use of organic manure offers the room for mitigation potential. The average abatement rate for wheat and maize were extended to other upland crops due to lack of data. Net emissions of adding manure to rice paddy were estimated under intermittent irrigation regime (F-D-F-M). Another point needs to be highlighted is that organic manure applied in combination with synthetic fertilizers should be previously composted/fermented or be biogas residues since incomposted manure could increase CH_4 emissions by over 100% in rice paddies while by only 30-40% when treated.

Direct meta-analysis outputs were used for conservation tillage (measure C7) and straw retention (measure C8) without additional treatment. Practice of conservation tillage in upland cropping systems increased SOC content significantly at a rate of 0.915 t CO_2e/ha/yr. However, N_2O emissions also increased by 46% compared to conventional tillage, offsetting part of C sequestration gains and leading to an overall technical mitigation rate of 0.611 tCO_2e ha/yr. In the same vein, the abatement rate of straw residues retention (0.263 tCO_2e/ha/yr) compared to farmlands with only chemical fertilizer application is also inferior to the rate of C sequestration at 0.294 tCO_2e/ ha/yr.

Application of biochar (measure C9) produced with crop straw pyrolysis could increase annual C sequestration by 17% (i.e.6.47 tC/ha) compared to controlled plots; however this value is based on very few short-term experiments and the maximum duration of experiment was 2 years. Because of lack of data on long term effects of biochar application on SOC, the current abatement rate estimation did not include the potential CO_2 removal benefits and only accounted the decrease in N_2O emissions in uplands by 40% and in rice paddies by 50% from global meta-analysis results (Pan, 2012).

Results of abatement rate of mitigation measures

The mitigative effects on each gas and stand-alone abatement rates of cropland mitigation measures are presented in Table 3-3. For reference, similar information for the livestock sector is illustrated in Table H in Annex 4. While C1, C2, C4 and C5 target exclusively N_2O emissions as synthetic N-use related measures, the main abatement contribution of measure C3 originates from avoidance of CH_4 emissions. Application of more organic manure (measure C6) and agricultural management practices (conservation till C7 and straw retention C8) all trigger increases in N_2O emissions, but such negative impacts could be neutralized by higher C sequestration potential. Consequently, these measures could be regarded as important mitigation options.

Table 3-3 Mitigative effects and stand-alone abatement rates of cropland mitigation measures

	Mitigative effects			Stand alone abatement rate (tCO_2e/ha)							
Measure No.	N_2O	CH_4	SOC	Rice	Wheat	Maize	Other upland crops	Greenhouse vegetable	Openfield vegetable	Fruit	Average
C1	+			0.075	0.351	0.406		1.225	0.505	1.266	0.412
C2	+				0.19	0.208					0.201
C3	-	+		1.337							1.337
C4	+						0.903 (cotton)	1.376	0.829	1.827	1.219
C5	+			0.127	0.273	0.256	0.274	0.667	0.369	0.616	0.271
C6	-	-*	+	0.460	0.689	0.574	0.631		0.227	0.462	0.596
C7	-		+		0.611	0.611					0.611
C8	-		+		0.263	0.263					0.263
C9	+		+	0.187	0.364	0.342					0.329

Notes: + denotes reduced emissions or enhanced removal (positive mitigative effect)
- denotes increased emissions or suppressed removal (negative mitigative effect)
* Here CH_4 emissions increase is only applied to rice paddies.

Source: Author's calculation incorporated in Wang et al. (2014)

For arable land, average abatement rates range from 0.201 tCO_2e/ha from better application time and methods to further reduce N rate in wheat and maize fields, to 1.337 tCO_2e/ha delivered by improved fertilization and irrigation regimes in rice paddies. Higher per hectare abatement benefits can generally be achieved from cash crops than cereals in implementing the same mitigation practice. This is because synthetic N overuse and misuse is more prevalent in fruits and vegetables than cereal crops (Zhang et al., 2012a) as well as the natural higher demand for N fertilizers of higher-value cash crops. However, this does not apply to measure C5 on organic manure since organic manure has already been widely used in fruit and vegetable fields but not popularized in cereal croplands. Among the three main crops, rice paddies offer lowest per area N_2O abatement potential since the overall emission factor for N_2O emissions from N inputs for paddy fields is less than half of that for uplands (Table 2-2).

1.3. Treatment of measure interactions

Effects of measure interactions

An important feature of mitigation in agriculture is that an abatement measure is usually not applied on its own, i.e. stand alone, but rather in combination with other measures. In other words, on the same piece of land more than one abatement measure are very likely to be adopted but the integrated mitigation potential is rarely the sum of the potential of individual measures implemented independently because agricultural activities involve complex biological processes. As a result of measure interactions, implementation of a mitigation action is likely to modify the efficacy or the baseline of another. For example, optimal addition of organic manure in uplands will largely reduce the room for further SOC increment from crop residues retention. In addition, the effect on the overall mitigation potential of taking into account measure interactions also depends on the order in which the measures are implemented. For instance, it is technically meaningless to adopt straw addition in uplands where conservation tillage has already been introduced since by definition the latter requires at least 30% of crop residues being incorporated to soils. It is therefore essential to account for these technical interactions to the maximum extent possible to avoid “double counting”. The French agriculture MACC (Pellerin et al., 2013) analysis shows that inclusion of measure interactions decreases the annual cumulative mitigation potential by 8-18% depending on calculation methods.

Methodologies on how to address measure interactions are explicitly stated in international literature on agriculture MACCs construction. In developing the MACC for UK agriculture, Moran et al. (2011) used a simple interaction factor (IF) to express the extent to which the efficacy of a measure is reduced (or in some cases, increased). Each time a measure is implemented, the abatement rates of all the remaining measures are recalculated by multiplying them by the appropriate IF. Regarding the French agriculture MACC (Pellerin et al., 2013) interactions between sub-actions within an action were firstly taken into account followed by considerations on interactions between actions, assuming that actions affecting crop rotation are implemented in priority. The baseline (e.g. available areas and changed N rates) for remaining actions to be introduced is also reevaluated each time after an action is adopted according to pre-defined criteria. Among sub-actions, the quantity of mineral fertilizers applied to each crop decreases successively after adjustment of productivity targets, inclusion of organic N inputs, removing or postponing the base fertilization, introduction of nitrification inhibitors, and finally a better localization of fertilizers. The Irish agriculture MACC also accounted for abatement measure interactions which were explicitly stated in the methodological description of each individual measure (Schulte et al., 2012). For example, the abatement potential for reducing N fertilizer rate has accounted for the reduction effects in N fertilizer use of improved manure management.

Treatment of measure interactions in this research

In this study, possible measure interactions were addressed in three steps. In the first place, aggregation of sub-actions into integrated mitigation measures has partially internalized some interactions. For example, practices such as adjusting fertilizer application time, integrated irrigation and fertilization system as well as switch of N products are likely to overlap in terms of application areas, but objective-oriented measure aggregation offers farmers the flexibility to choose the most appropriate (combination of) actions to achieve the highest NUE. Treatment of measures interaction in the livestock sector is described in Annex 4.

Secondly, the defined mitigation measures were then assigned implementation orders based on their relative importance. Considering the serious problem of N overuse in China, reduction of total fertilizer N amount (measure C1) is given top priority, followed by better application time, method and product to further address the low efficiency of N use in China (measure C2 and C4). Special attention is dedicated to water management regimes limiting emissions of CH_4 from rice paddies which is the dominant GHG (measure C3). After capitalizing the full potential of lowering chemical N rate, naturally the next-step is to deliberate on ways to enhance fertilizers efficiency (measure C5) and alternative types of N input that are more beneficial to soil quality and productivity (measure C6 on organic manure and C9 on biochar). In the meantime, better land management (measure C7) and crop residues management (measure C8) are encouraged to form integrated soil-crop management systems. In this regard, the N rate levels lowered by one measure serve as the basis for estimating abatement rates of subsequent measures. For example, if measure C1 and C2 allow N application rates to decrease from 300 kg/ha to 200 kg/ha, the mitigation effect of adding nitrification inhibitors (measure C5) will be evaluated based on the N rate of 200 kg/ha to avoid overestimation. The potential of adding organic manure to rice paddies (measure C6) was quantified under the intermittent water regime (F-D-F-M) realized through measure C3.

Finally, land and straw residues management practices generally don't interact with N fertilizer related options. However, adjustments do need to be made to accommodate potential overlapping application of measures with similar effects (e.g. organic manure and biochar) or subordinating relationships (e.g. conservation tillage and straw returning). Further, the efficacy of increasing organic manure to lands will be discounted when applied jointly with conservation tillage or straw returning, all of which achieve mitigation through carbon sequestration in soils. We therefore assigned an interaction factor (0.8) to the stand-alone abatement rates of the three measures (C6, C7 and C8) on wheat and maize areas. For example, abatement potential of adopting conservation tillage was discounted from 0.611 to 0.489 CO_2e/ha. We assumed that measure interactions shall not affect the implementation costs of measures.

2. Which technical potential can be realized from Chinese croplands?

2.1. Measure adoption additional to the baseline scenario

Apart from the abatement rate, information on the additional area (over and above the baseline area) that the measure could be applied to is also required to calculate the total mitigation potential for each measure in the given time horizon. It is important to emphasize the additionality of measure application compared to BAU or baseline activities.

In this study, we aim to identify the maximum mitigation potential as the upper limit that would result from the technically feasible level of measure implementation, despite that the actual mitigation extent depends on behavioral, political and market constraints measure adoption. Measure uptake under the BAU scenario (Table 3-4) was derived with reference to either relevant policy targets or historical trends; those under the maximum abatement scenario were identified based on expert judgment, scientific literature as well as applicability of the specific measure. For reference, livestock measure uptake under the BAU and abatement scenarios is summarized in Table I in Annex 4.

Crop and soil measures C1, C2 and C4 were assumed to be applicable in provinces and municipalities with lower NUE than the specific target levels in each jurisdiction (Table 3-6). Historical changes in water regime patterns in rice paddies (measure C3) referred to results reported by Zou et al. (2009) and Zhang et al. (2011). Baseline extension areas of high-efficient irrigation systems (C4), conservation tillage (C7) and straw returning (C8) correspond to explicit targets set in the National Agricultural Water-Saving Outline (2012-2020) (State Council, 2012b), the National Agriculture Mechanization Extension Plan (2011-2015) (MOA, 2011), and the Implementation Plan on the Comprehensive Use of Crop Straw during the 12th Five-Year Plan Period (NDRC, 2011a).

Table 3-4 Measure adoption under baseline and abatement scenarios

Measure No.	Historical or current adoption	Baseline adoption in 2020	Maximum feasible adoption in 2020	References or explanation
C1		Apply to 39% rice, 44% wheat, 55% maize, 100% greenhouse veg., 50% openfield veg. and 70% fruit fields.		See Table 3-6 measure C1
C2		Apply to 100% wheat and maize cropping areas.		See Table 3-6 measure C2
C3	Areas under F, F-D-F, F-D-F-M regimes were 16%, 77%, 7% in 1980s and 12%, 76%, 12% in 1990s.	Areas under F, F-D-F, F-D-F-M regimes are 8%, 76%, 16%.	Areas under F, F-D-F, F-D-F-M regimes are 8%, 0%, 92%	Zou et al. (2009) Zhang et al. (2011)
C4		Apply to 50% of cotton, greenhouse and openfield vegetable and fruit fields.*	Apply to 100% cotton, greenhouse and openfield vegetable and fruit fields.	See Table 3-6 measure C4. National Agricultural Water-Saving Outline (2012-2020)
C5	Limited	Limited	Apply to 50% rice, wheat and maize, 30% other upland crops (excluding beans), and 30% of vegetables and fruits.	
C6		30% of crops receive reasonable supply of organic manure.	80% of crops (except greenhouse veg.) receive reasonable supply of organic manure.	
C7	4.30 Mha (7.6% of wheat and maize areas) in 2010	20 Mha(34.8% of wheat and maize areas)	23 Mha(40% of wheat and maize areas)	National Agriculture Mechanization Extension Plan (2011-2015)
C8	28.5Mha (about 18 Mha of wheat and maize areas, straw retention on 60% of mechanized harvest areas)	22.5 Mha of wheat and maize areas (assuming straw retention on 60% of mechanized harvest areas†).	30.1Mha of wheat and maize areas (assuming straw retention on 80% of mechanized harvest areas).	National Agriculture Mechanization Extension Plan (2011-2015). Implementation Plan on the Comprehensive Use of Crop Straw during the 12th Five-year Plan Period
C9	Limited	Limited	Apply to 10% of rice, wheat and maize cropping areas.	

* According to the National Agricultural Water-Saving Outline, high-efficiency irrigations shall be installed on 22.5 Mha croplands (20 Mha new areas). We estimated that approximately 30% of cash crops shall benefit from this project.

† Areas with straw retention are highly dependent on crop harvesting mechanization levels, which were 64.5% for rice, 86% for wheat and 25.8% for maize in 2010, and are planned to reach 80% for rice and 45% for maize in 2015.

2.2. Maximum feasible mitigation potential from croplands

Based on information from Table 3-3 and Table 3-4 and taking into account measure interactions, the crop-weighted abatement rate, total additional area for application and the overall technical mitigation potential of each cropland measure are summarized in Table 3-5. We also distinguished mitigation potential arising from CH_4 and N_2O emission reductions and those achieved through carbon sequestration in soils. Mitigation potential of livestock measures is presented in Table J in Annex 4.

Table 3-5 Average abatement rate, additional application area and mitigation potential of cropland measures

Measure No.	Weighted abatement rate (tCO_2e/ha)	Additional application area (M ha)	Annual mitigation potential in 2020 ($MtCO_2e$)	Potential through emission reductions ($MtCO_2e$)	Potential through carbon sequestration ($MtCO_2e$)
C1	0.412	58.63	30.65	30.65	
C2	0.201	56.65	11.38	11.38	
C3	1.337	17.93	23.98	23.98	
C4	1.219	17.94	21.86	21.86	
C5	0.271	57.23	15.54	15.54	
C6	0.596	120.11	40.19	-5.77	45.96
C7	0.489	22.98	1.46	-0.72	2.18
C8	0.210	30.06	0.95	-0.11	1.07
C9	0.329	9.90	3.26	3.26	
Total			149.27	100.06	49.21

Source： Author's calculation and incorporated in Wang et al. (2014)

Table 3-5 shows that in 2020 under the maximum technical abatement scenario mitigation potential amounts to 149 $MtCO_2e$, representing 35% of BAU emissions (Figure 2-4). Equivalent emissions of 100 and 49 $MtCO_2e$ could be avoided from and stored in croplands, respectively. When only accounting the measures targeting CH_4 and N_2O (i.e. excluding C6, C7 and C8), abatement potential declines to 107 $MtCO_2e$ in 2020. Measure C6 on more efficient recycling of organic manure could be possibly applied to the largest additional area and offers the most significant potential, which is achieved exclusively by carbon sequestration in soils. Large amount of emissions could also be avoided through N fertilizer best management practices, collectively providing over 40% of cropland abatement potential. Direct reduce of N fertilizer rate (measure C1) presents the highest potential among emission reduction measures. The relatively low potential of conservation tillage and crop straw retention can be attributed to significant measure uptake under the BAU scenario due to policy enforcement, leaving limited scope for additional application.

3. Case study: mitigation potential from cereal production in China

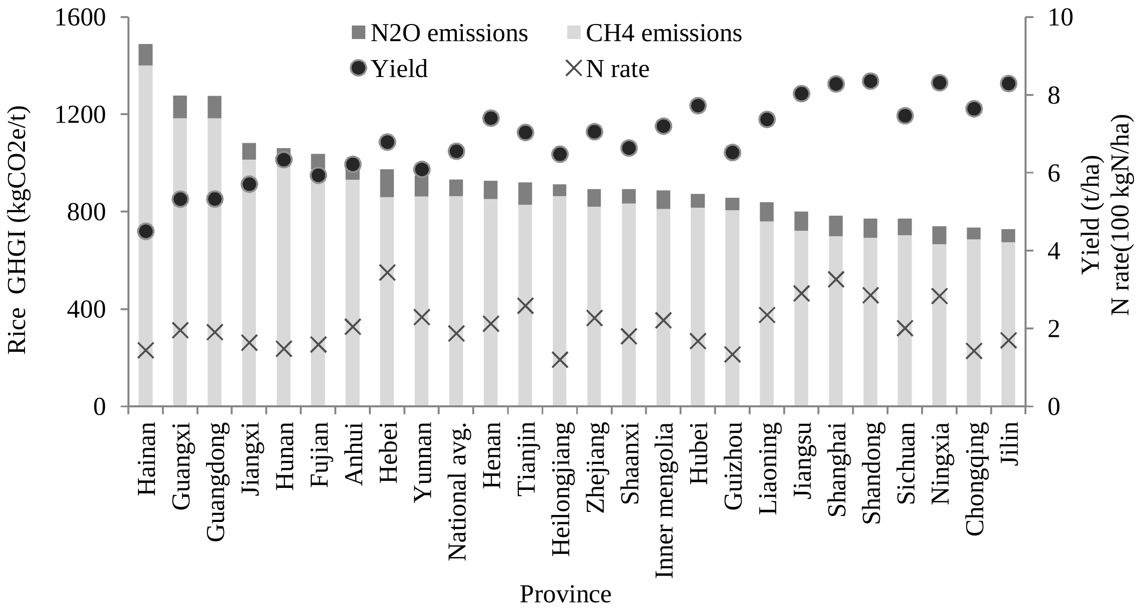
This section will first present the current and historical status of GHGI of cereal production at the provincial and regional level (section 3.1) as the basis for discussing the abatement potential from reducing synthetic N use at the provincial level (section 3.2). The provincial mitigation potential will be aggregated to generate the national total value (section 3.3).

3.1. Current and historical GHGI of cereal production at the provincial level

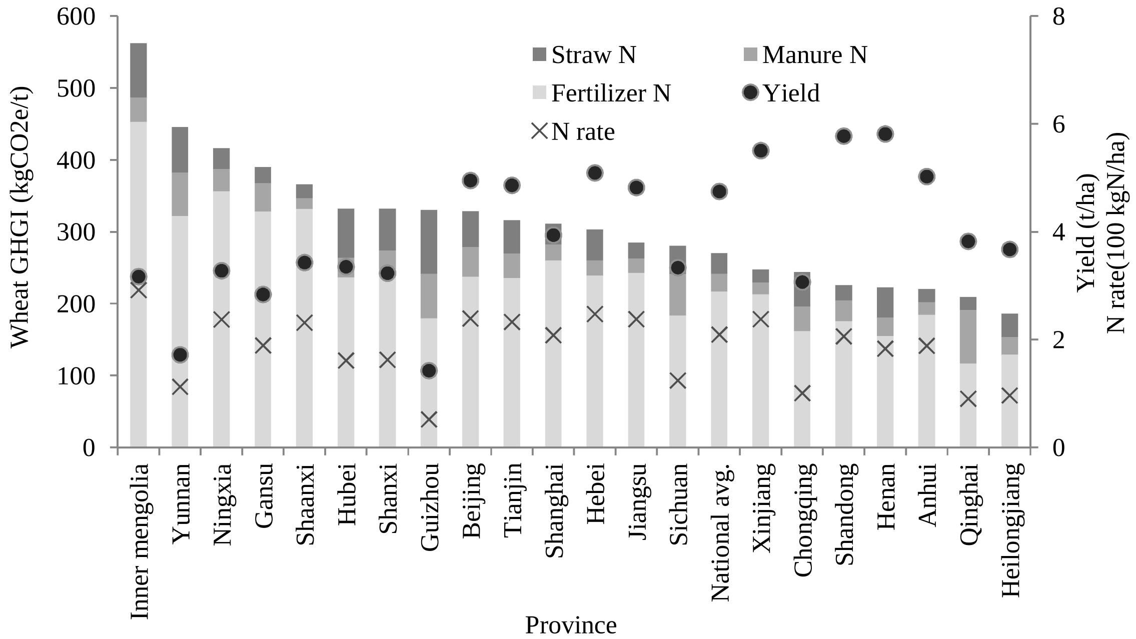
Following the same methodology used to quantify the GHGI of rice, wheat and maize production on the national scale (section 3.2 in chapter 2), similar evaluation is conducted for provincial-level GHGIs. Agriculture activity data were collected at the provincial level while emission factors and other parameters were average national values. In other words, data for $N_2O\text{-}N_{input}$ are province-specific and $Flux_{CH4}$ are region-specific, while other factors were held identical among provinces. Regions in China refer to northeast, north, northwest, east, south and central, and southwest China, each of which includes 3-7 provinces/municipalities.

GHGI of rice, wheat and maize at provincial scale in 2010

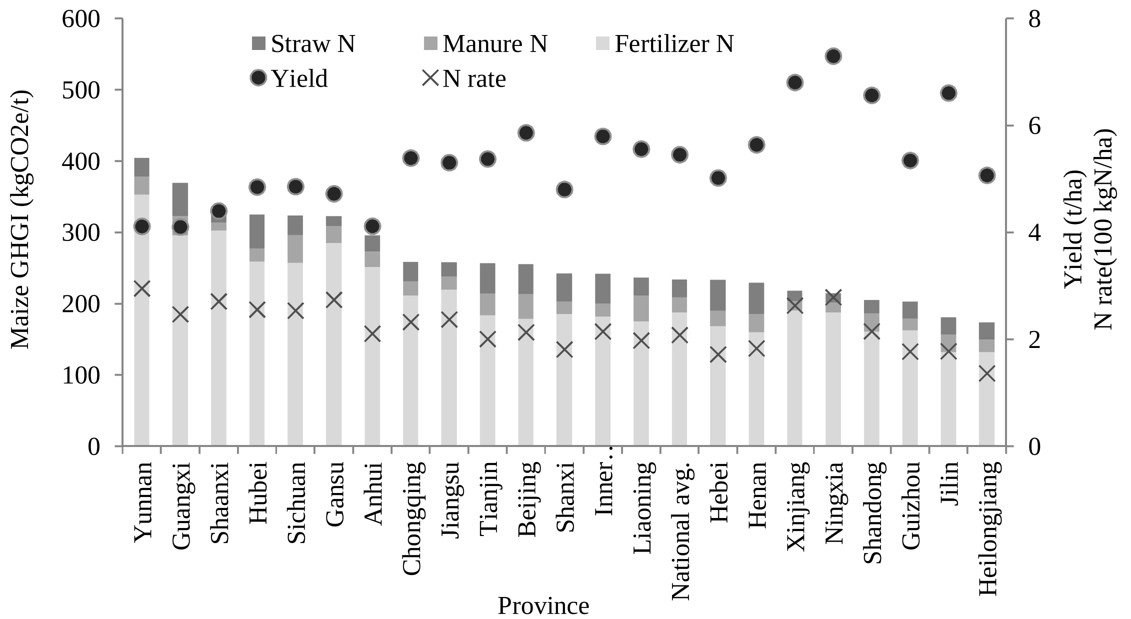
Figure 3-1 GHGI of rice (a), wheat (b) and maize (c) production in different provinces in 2010



(a)



(b)

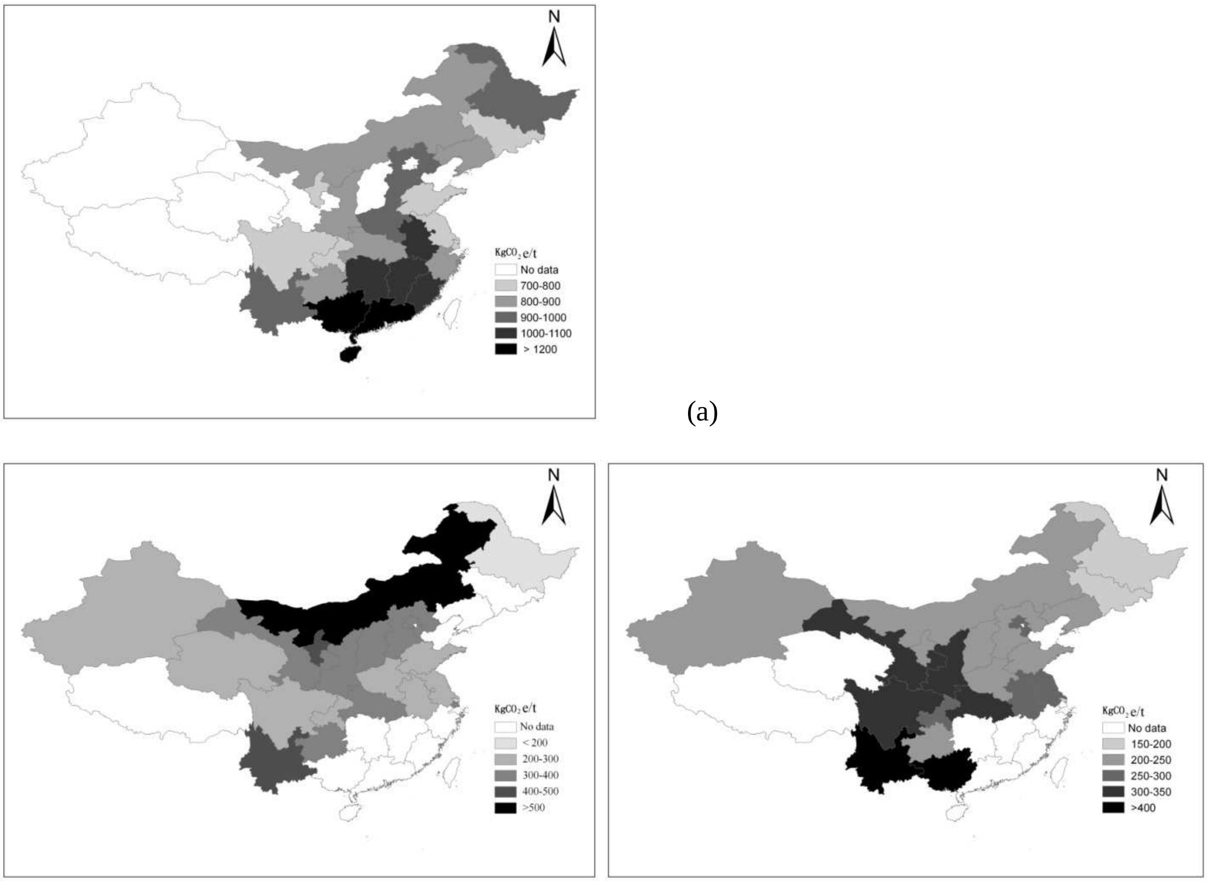


(c)

Note: the bars in (a) represent the contributions of N_2O and CH_4 to total GHGI; the bars in (b) and (c) represent the contributions of different N inputs to N_2O-derived GHGI.

Source: Results calculated by the author and incorporated in Wang et al. (2015)

Figure 3-2 Geographic pattern of rice (a), wheat (b) and maize (c) GHGI in 2010



(a)

(b) (c)

Source: Results calculated by the author and incorporated in Wang et al. (2015)

GHGI of rice production in 2010 ranged from 729 kgCO_2e/t in Ningxia Province to 1,488 kgCO_2e/t in Hainan Province, with a national average of 933 kgCO_2e/t (Figure 3-1(a)). There was no obvious relationship between GHGI levels and N application rates, the latter being the major source of N_2O emissions. For example, the Jiangsu Province in east China received 51% higher N application than national average in rice production but was moderate in GHGI (16% lower than national average). It is, however, evident that the estimated GHGI of rice production was negatively correlated with yield levels. There was a large provincial variation in GHGI (Figure 3-2 (a)) with the most carbon intensive provinces located in the southeast coastal areas due to the highest regional CH_4 flux (252 kg/ha) because of higher temperature and greater level of organic matter input (Zhang et al. 2011a). The low GHGI of rice production in the southwestern provinces (Sichuan, Chongqing, Guizhou and Yunnan) can be attributed to relative lower CH_4 flux (210 kg/ha) relative to other places (221-252 kg/ha). Among the six major rice producing provinces, which accounted for 56% of the national production, Jiangxi and Hunan had higher

GHGIs than the national average while Hubei, Jiangsu, Sichuan and Heilongjiang were below the national mean.

Large spatial variability of average GHGI of wheat (Figure 3-1(b)) and maize (Figure 3-1(c)) production could be observed among provinces. For example, producing one ton of wheat in Inner Mongolia emitted 3 times more N_2O than in Heilongjiang, attributable to significant differences in synthetic N input and wheat and maize yields between Chinese provinces. In general, synthetic N fertilizer made up at least 70% of total emissions and was therefore the primary emission contributor. Figure 3-1 (b and c) also shows that the trends of GHGI, which are affected by place-specific yield levels, were not necessarily consistent with those of per hectare N application rates. For example, although the N application rate for maize in Ningxia (279 kgN/ha) was 13% higher than in Guangxi (247 kgN/ha), a much higher yield in Ningxia (7.30 t/ha) than in Guangxi (4.10 t/ha) resulted in a lower maize GHGI in Ningxia. In contrast, high N rate and low productivity made Ningxia one of the most carbon intensive provinces for wheat cultivation.

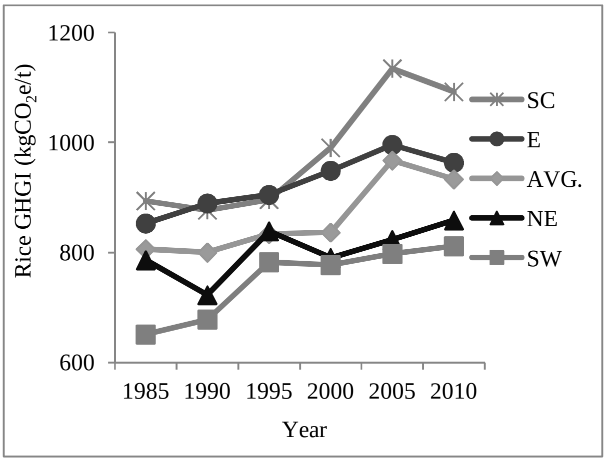
The geographic variations of GHG emissions per ton of wheat (Figure 3-2(b)) and maize product (Figure 3-2(c)) show both similarities and differences. In general, similar levels of GHGI can be observed for wheat and maize production (except for Ningxia), e.g. Yunnan was one of the most carbon intensive areas for both wheat and maize production in 2010. More N fertilizers were added to croplands in the northwest provinces to compensate poor soil fertility, resulting in elevated regional GHGI of wheat and maize production. The levels of maize GHGI converged to the range of 200-300 $kgCO_2e/t$ with obvious correlation with N rates and yields. Provincial discrepancies were more evident for wheat GHGI, implying that farmers were potentially more rational in determining the fertilizer amount for maize than for wheat. Among the five major wheat producing areas- Henan, Shandong, Hebei, Anhui and Jiangsu, which contributed about 74% of the national production, GHGI levels in Hebei and Jiangsu were superior to the national average. All major maize producing areas- Hebei, Jilin, Shandong, Henan and Heilongjiang, had lower GHGI than the national mean value.

GHGIs at the provincial level were further integrated to the regional scale for 2010 and compared with yields and SOC contents (Fig. A) to indicate regional GHGI reduction strategies (Annex 5).

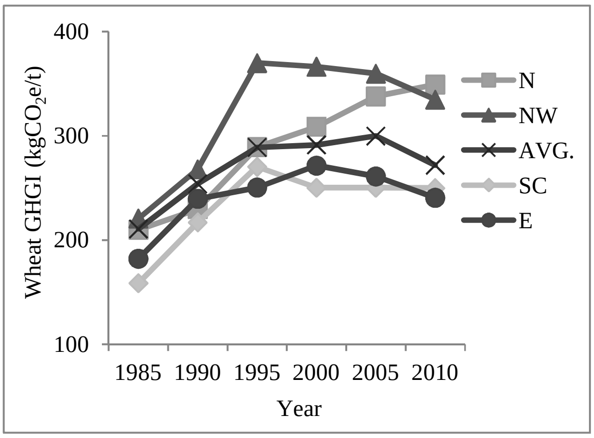
Historical trends of regional GHGI of cereal production

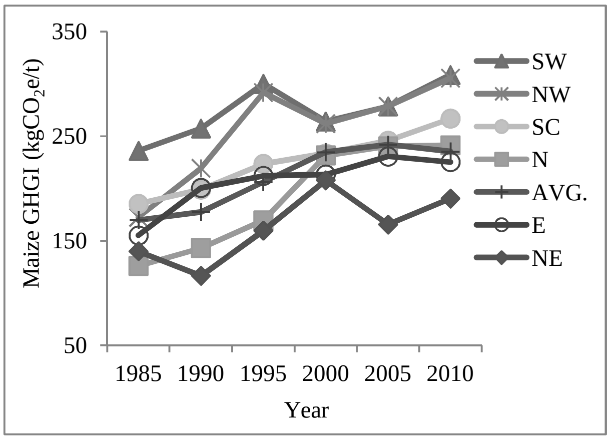
Nearly all regional GHGI of rice, wheat and maize production reached a higher level in 2010 relative to 1985 (Figure 2-3). For rice production (Figure 3-3 (a)), south and central and east regions have consistently been the most carbon intensive areas due to favorable climate conditions and greater level of organic matter application (Zhang et al., 2011a). In paralelle, rice paddies in eastern, southern and central China are found to have experienced the greatest SOC increase (Zhang et al., 2007; Pan et al., 2010). In contrast, lower level of crop residues, farm manure and green manure application enabled southwest to emitt least GHG in producing same amount of rice. As to the GHGI of wheat production (Figure 3-3b)), all regions except north China exhibited the same trends as the national average: sharp increase from 1985 to 2000 and stabilization or decrease thereafter. Consequently, reducing N rates should be advocated in northern provinces, confirming the findings of other experimental and theoritical studies (Ju et al., 2009, 2011). Maize GHGI evolution (Figure 3-3(c)) patterns were more diverse between geographic regions with northeast China having the least GHGI. The northwest has been characterized with the highest GHGI in both wheat and maize production.

Figure 3-3 Historical changes of provincial GHGI of rice (a), wheat (b) and maize (c) production



(a)





(b) (c)

Note: North(N) region includes Beijing, Tianjin, Hebei, Shanxi and Inner Mongolia; Northeast (NE) region includes Heilongjiang, Liaoning and Jilin; East (E) region includes Shanghai, Anhui, Fujian, Jiangsu, Jiangxi, Shandong and Zhejiang; South Central (SC) region includes Guangdong, Hainan, Henan, Hubei, Hunan and Guangxi; Southwest (SW) region includes Chongqing, Guizhou, Sichuan, Yunnan and Tibet; Northwest (N) region includes Gansu, Qinghai, Shaanxi, Ningxia and Xinjiang.

Source: Results calculated by the author and incorporated in Wang et al. (2015)

The obvious regional discrepencies observed in the evolution of cereal GHGI justify the necessity of conducting a provincial-scale estimation of abatement rate of fertilizer-related mitigation measures rather than taking a national-wide approach to take into account local circumstances and deliver more accurate estimates.

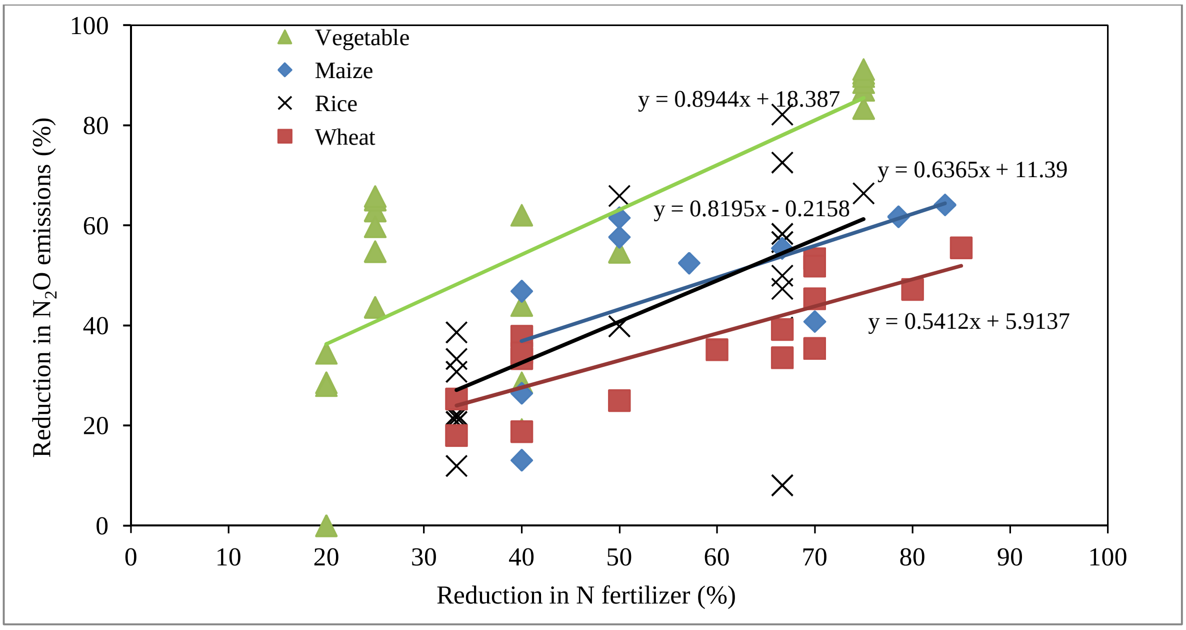
3.2. Abatement rate of N fertilizer management at provincial level

Prior to estimating the maximum potential from fertilizer N use reduction, it is important to determine an indicator to evaluate the NUE for different crops. There are four agronomic indices commonly used in China to describe NUE (Zhang et al., 2008): partial factor productivity of applied N (PFP_N, kg crop yield per kg N applied); agronomic efficiency of applied N (AE_N, kg crop yield increase per kg N applied); apparent recovery efficiency of applied N (RE_N, kg N taken up per kg N applied); and physiological efficiency of applied N (PE_N, kg yield increase per kg N taken up). This research endorsed the term PFP_N to describe NUE owing to the data availability and consistency with the GHGI calculation (PFP_N constitutes part of GHGI accounting for merely the part of synthetic N). The more the NUE is improved, the lower the GHGI will be.

We assume the optimal PFP_N to be achieved in two steps: first a direct reduction in synthetic N fertilizer use for certain crops in targeted provinces to raise regional PFP_N to 70% of the optimal levels (measure C1) and secondly adoption of other techniques such as better application time and placement to unlock the full potential. The regional optimal PFP_N were derived from scientific fertilization recommendations by Zhang et al. (2009). The current level of PFP_N was quantified using the same database for calculating GHGIs. The PFP_N in 2020 under the BAU scenario was estimated by dividing predicted crop yield by projected synthetic N rate. Changes in future N rate, crop yield as well as cropping area on the provincial scale was assumed to follow the national trend (Table E, Table 2-3): e.g. wheat yield improves by 0.8%, wheat area declines by 0.9%, and N application increases by 1.3% per year. As to the yield improvement under the abatement scenario, better synchrony in time and place between crop N nutrient demand and N supply could increase rice and wheat yield by 5% and maize by 8% based on a large number of on-farm demonstration trails (Zhang et al., 2012b).

Linear response of N_2O emissions to N fertilizer amount is generally observed (Zou et al., 2005; Mosier et al., 2006; IPCC, 2006) at low and intermediate fertilizer application rates. However, there is growing evidence that N_2O emissions increase abruptly at superoptimal level of N inputs indicating an exponential relationship between N addition and N_2O emissions (Grant et al., 2006; Zebarth et al., 2008; Hoben et al., 2011). These findings imply that improving NUE by avoiding N surplus will substantially reduce N_2O emissions by a greater proportion than the reduction in N rate. Given the phenomenal problem of N overuse in China, we decided to account for this non-linear relationship between N fertilizer reduction and N_2O emissions reduction based on database from site experiments (Nayak et al., 2014). The relationship shown in Figure 3-4 supported the estimation of abatement rates of measure C1-C4 on the provincial scale. Due to lack of sufficient experimental data from fruit fields, we used emission data from vegetable to represent fruits.

Figure 3-4 Relationship between reduction percentages of N fertilizers and N_2O emissions



The equation for rice is y=0.8195x-0.2158, for wheat is y=0.5412x+5.9137, for maize is y=0.6365x+11.39, and for vegetable is y=0.8944+18.387.

Source: Author

Annex 6 presents the N_2O abatement rate of measure C1, C2, C3 and C4 related to direct N use reduction. In rice paddies, highest per area N_2O abatement benefits are estimated to generate from Hebei, Shanghai, Liaoning, Jiangsu and Ningxia - provinces characterized by relatively high N rates. In the same vein, provinces which are sources of the highest N_2O abatement rate in wheat and maize are generally those with top rankings in terms of GHGI of wheat production (Inner mongolia, Ningxia, Shannxi and Jiangsu) and maize production (Yunnan, Shannxi, Gansu and Guangxi), respectively. The exception is wheat production in Jiangsu which is moderate in GHGI but has high abatement potential as a consequence of the serious problem of fertilizer N overuse in this province (Ju et al., 2009).

3.3. Mitigation potential from best N management practices

When determining the measure maximum uptake, the whole cropping areas in a province were supposed to be subject to a mitigation action when the provincial average PFP_N falls below the target despite of disparities of PFP_N within the province. As to mitigation potential from vegetable and fruit fields, we concluded from relevant literature (Ge, 2009; Jiao et al., 2010) that overuse of N fertilizer was phenomenal in nearly all greenhouse vegetable fields, and meanwhile we assumed about 50% of openfield vegetable areas received excessive N fertilizers 40% higher than crop demands. Regarding orchards, survey results (Lu et al., 2008; Zhang et al., 2012b) indicate that average N inputs rates were over 2.5 fold higher than fruits requirement in about 70% of Chinese orchards.

Table 3-6 summarizes the national average N rate, total chemical N savings, application area and overall N_2O emission reduction potential from implementing measure C1-C4. The nationwide or per hectare use of N fertilizer is projected to reduce by 20-42% relative to 2020 baseline levels. Since we adopted a conservative approach to estimate mitigation potentials, our results of around 30% cut in N application rates of cereal crops are inferior than the N use reduce suggestions by other researchers (Ju et al., 2009; Zhang et al., 2013). Under the abatement scenario, PFP_N of rice, wheat and maize will rise to 54.5, 34.7 and 46.5 kg grain/kgN respectively, which shall reach the world average level (42.5, 36.3 and 42.5) but are lower than the EU and US levels in 2006 (database from Heffer (2009)). This is justifiable since in average soil fertility is poorer in Chinese arable lands than in Europe or the US, requiring more fertilizers to reach same level of yields.

More than 30% improvement in PFP_N of the three major crops in China will lead to an abatement potential of 25.3 $MtCO_2e$, representing 10% of GHG emissions from rice, wheat and maize cultivation in 2010. Based on 1990s dataset, Huang & Tang (2010) concluded that N_2O emissions from rice, wheat and maize production could be reduced by 44 Gg/yr and 104 Gg/yr if NUE (used RE_N as indicator) is to be increased by 30% and 50%, respectively, which is equivalent to 13.1 $MtCO_2e$ and 31.0 $MtCO_2e$. Our results of 25.3 $MtCO_2e$ is comparable to this analysis result. It should be emphasized that the mitigation potential from best fertilizer management practices might have been underestimated since we did not take into account the manure and residue N supply in quantifying the optimal PFP_N.

Table 3-6 Application area and overall N_2O mitigation potential from measures C1-C4

	Average N rate (kg/ha)			Total N saved compared to 2020 baseline (kt)			National average PFP_N (kg/kg)			Application area (kha)		N_2O mitigation potential ($MtCO_2e$)	
	2010	2020 baseline	2020 abatement scenario	Measure C1	Measure C2-C4	Total saving	2010	2020 baseline	2020 abatement scenario	Measure C1	Measure C2-C4	Measure C1	Measure C2-C4
Rice	186	177	133	331	803	25%	38.9	38.9	54.5	26%	59%	0.75	1.84
Wheat	199	238	155	797	1030	35%	21.5	21.5	34.7	44%	98%	3.42	4.16
Maize	202	221	146	1258	1408	34%	28.4	28.4	46.5	55%	98%*	7.87	7.22
Greenhouse vegetable	671	656	379	350	635	42%	84.1	84.1	160.0	100%	50%	4.36	2.45
Openfield vegetable	268	262	210	244	573	20%	145.1	145.1	190.7	50%	50%	3.91	6.42
Fruit	492	565	350	831	1679	38%	22.6	43.3	73.4	70%	50%	10.34	10.66

* Maize cropping areas in Guangdong and Hunan account for 2% of national total; but the N rates were not reported in the statistical yearbook, relevant mitigation potential was therefore not accounted here.

4. Conclusion

Evaluation of the technical mitigation potential from Chinese croplands started with the screening of abatement options applicable to China's agricultural systems. Based on literature review and expert elicitation, we identified 9 mitigation measures that are currently available and acceptable, comparable to the agricultural development priorities and likely to generate important mitigation potential. These measures integrate a broad spectrum of techniques and practices that not only enable the reduction of N_2O and CH_4 emissions from production but also enhance the removal of atmospheric CO_2 through carbon sequestration in soils.

The abatement rate of each measure was concluded from meta-analysis results relying on documented evidence from experimental trails. However, we were unable to quantity the complete climate effects of some measures due to limited observations (e.g. SOC impacts of biochar amendment) while some were not China-specific values for the same reason (e.g. enhanced efficiency fertilizers). Therefore, this study was basically a country-wide exercise considering the challenge in data availability. Although we recognize that more regionalized abatement rates would merit better assessment, it appears difficult to extract these data at this stage. Further scientific research is needed to understand the applicability and opportunity of to implement the measures in the diverse agricultural systems, which constitutes part of a relevant research agenda in China. Nevertheless, our estimations of abatement rates (0.201-1.337 tCO_2e/ha) are comparable to those conducted by US EPA (2013), IPCC (2007) and Moran et al. (2011). Best N fertilizer management practices, combined with improved irrigation regimes, could deliver high per hectare abatement rate, especially for fruits and vegetables. More efficient recycling of organic manure, conservation tillage and straw addition are also important mitigation options to enhance the SOC contents.

There is great uncertainty concerning how largely the mitigation measure could be adopted in the target year. There are several measures never having been studied thoroughly on their regional applicability, so any assumptions relative to maximum scale of their adoptation would be variable to this research. To minimize such uncertainties we took a conservative approach in defining the uptake of technically feasible measures in 2020. For the same purpose, measure interactions were addressed primarily by aggregation of sub-actions and assignment of implementation priority to avoid possible overlapping of measures adoption and double counting. In total, our findings indicate an amount of 149 $MtCO_2e$ could be possibly mitigated from Chinese croplands, representing 35% of the BAU emissions. About 2/3 of the mitigation potential is predicted to come from CH_4 and N_2O emissions reduction while the rest 1/3 via carbon storage. Organic manure amendment, N fertilizer management in uplands and water regime improvements in rice paddies offer the highest overall mitigation potential.

An in-depth examination has been carried out to investigate the potential from best N management practices for cereal crops at provincial level. In general, high potential is found in those regions topped the rankings in GHGIs. An abatement potential of 25.3 $MtCO_2e$ can be delivered when there is a 30% improvement in PFP_N of the three major crops.

Wider life cycle mitigative impacts and CO_2 emissions related to fuel combustion from machine use were not considered in this study. According to Zhang et al. (2013), manufacture and transportation of one ton of N fertilizer in China shall emit 6.1 tCO_2e in addition to 5.8 tCO_2e N_2O emissions from cropland application. This implies that about 63.6 $MtCO_2e$ emissions could be cut annually in the industrial and transportation sector in case of full implementation of the series of best fertilizer management practices. Mitigation activities in agriculture can thus make considerable contributions to the fulfillment of China's climate commitment. Accounting for agriculture fuel emissions shall decrease the abatement rates of some measures, e.g. subsurface placement of fertilizers and straw retention, while increase those of conservation tillage.

Now that we provide some insights into how mitigation potentials can be applied across the range of biophysical conditions that characterize Chinese farming systems, the next question comes naturally: what's the relative abatement cost of implementing each measure and are agricultural abatement opportunities cost-effective compare with both a benchmark carbon price and abatement elsewhere in the economy.

Chapter 4 - Economic mitigation potential in Chinese croplands

The previous chapter has found which abatement measures are applicable to the farming systems in China and the technical mitigation potential that could be expected from these options. However, these technically feasible mitigation measures normally are differentiated in terms of their implementation cost to farmers. This raises question on how to draw the cost curves to identify the most effective measures. The selection on efficient mitigation options requires therefore the understanding of the cost-effectiveness of each measure in terms of cost per tonne of CO_2e abated. This economic analysis constitutes the second bloc apart from the technical potential in constructing a MACC which will show the economic availability of the mitigation options and information on whether agricultural mitigations worth pursuing compared to both the benchmark carbon price and abatement elsewhere in the economy. This is significant, since in allocating an emissions budget to a sector, a rational mitigation policy should normally prioritize the cheapest means of abatement by equalizing marginal abatement costs across sectors. Such information is also crucial to develop any market-based approach and offering low cost mitigation credits to any emerging carbon trading schemes. The development of emissions trading regimes in China may lead to an increasing scrutiny over the relative cost of emission reductions in all sectors of the economy.

This chapter will conduct an economic analysis of mitigation potential from Chinese agriculture, taking account the cost of applying the identified measures relative to a baseline of no additional mitigation activities. In the first place, a general overview of literature on MACC analysis is carried out to justify the approach we take to build a MACC for China's agricultural sector (section 1). Section 2 will elaborate on the method of how to quantify the additional benefits/costs that the adoption of the mitigation actions entails to farmers. Based on the cost-effectiveness results, section 3 will present the MACC results as well as the abatement scenarios. Section 4 will be dedicated to some discussions on the MACC results, including comparison with other agriculture MACCs, sensitivity analysis and significance and limits of this study. Section 5 concludes and points out next-step research.

1. MACC tool introduction and approach selection

In this section we present the several ways of constructing agriculture MACCs in the literature and explain the methodological choice we have made in our thesis.

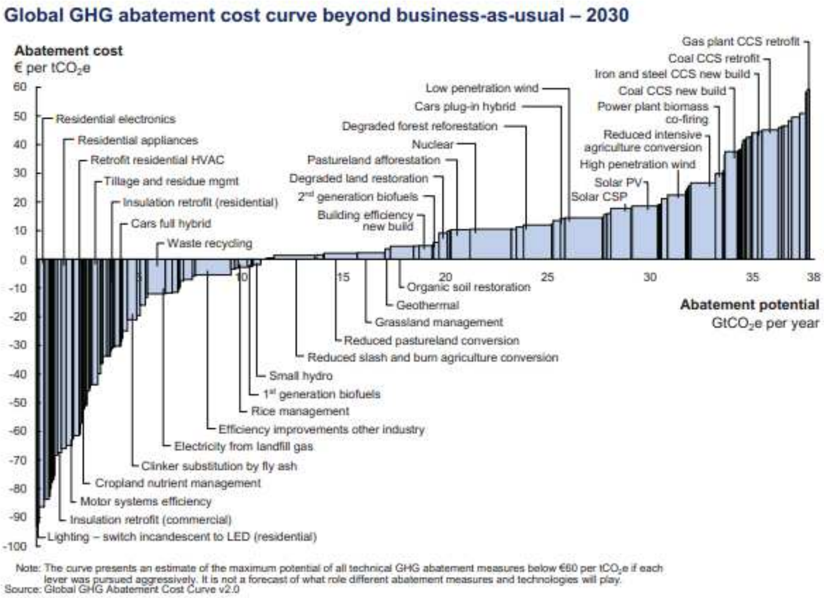
1.1. Literature review of existing agriculture MACCs

In order to identify and analyze cost effective mitigation options as well as quantify the overall potential that can be offered by a sector or a society, methodologies such as MACCs have been gradually developed over the past three decades. MACCs also enable comparison of the cost effectiveness of abatement options between different sectors of the society. MACCs have become a useful tool to assist policy makers in prioritizing mitigation options, especially with the release of a series of McKinsey&Co MACC reports (2009b, 2010). Advances in MACCs research has thereafter been accelerated and a range of GHG MACCs have been established for different sectors, including agriculture, at the global, regional, national, sub-national and local levels.

A MACC for GHG emissions is a graph that illustrates the relationship between the cost effectiveness of different abatement options and the total amount of GHG mitigation potential offered by these options in a given year. It reflects the expense associated with eliminating an additional unit of carbon and is upward-sloping: i.e. higher emission savings become increasingly expensive to achieve. According to Bockel et al. (2012), there are two types of MACCs graphs designed either as a histogram or a curve.

The histogram MACC represents the cost effectiveness and the mitigation potential of each individual abatement measure (Figure 4-1). Each bar represents a feasible abatement measure, differentiated by average implementation cost per ton of CO_2e emission reduced (height of bar), and quantity of emissions they can mitigate if the measure is fully applied to its technical potential (width of bar). The area (height*width) of the bar represents the whole cost of the action. The total width of the MACC shows the entire CO_2e savings available from all options. Measures below the x-axis are cost negative, i.e. removing emissions and saving costs; those above the x-axis entail implementation costs. Consequently, the biggest financial gains and emission reductions can be seen in the longest and widest bars under the x-axis, and conversely bars above the x-axis are the costlier measures. Policy therefore needs to focus first on the implementation of the former.

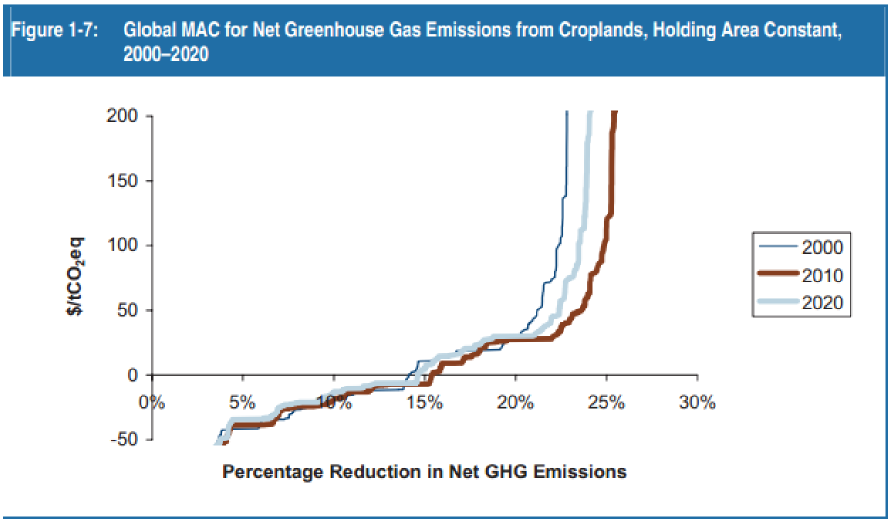
Figure 4-1 Illustration of a 'histogram' MACC



Source: McKinsey&Co (2009b)

MACC depicted as a curve (e.g. Figure 4-2) indicates the cost of abating the last unit of GHG emissions at a defined mitigation level (either as absolute abatement requirement or reduction percentage of the total emissions). The total abatement cost is represented by the integral of the area under the curve. The part of curve below the x-axis represents abatement opportunities at negative costs and the cost-effectiveness of mitigation options worsens moving along the curve from left to right.

Figure 4-2 Illustration of a 'curve' MACC



Source: US EPA (2013)

1.2. Choice of methodological approach for MACC construction

A MACC can be constructed following either a top-down or bottom-up approach. Vermont & De Cara (2010) provided a profound discussion on the two approaches and the underlined assumptions. Here we briefly describe some key aspects of the two approaches to justify the choice of the methodological method used to derive the MACC for China's agriculture.

A top-down analysis takes an exogenously determined emission reduction requirement and allocates it downward through modelling assumptions to conclude an overall abatement cost to the economy. This approach employs either microeconomic supply-side models (De Cara et al., 2005; Hediger, 2006; De Cara & Jayet, 2011; US EPA, 2013) or macroeconomic partial or general equilibrium models (IPCC, 2007; Schneider et al., 2007; Smith et al., 2008; Domínguez et al., 2009). In these models, simplified production functions are assumed to generally apply throughout the target sector. In agriculture supply-side models, a set of representative farmers are defined to maximize their gross margins or profits faced with technical and economic constraints. The abatement level can therefore be derived by stimulating farmers' decision when an emission tax or requirement is introduced. Prices are assumed exogenous without considering any possible market feedbacks; in particular we cannot capture the effects of increased costs on the demand-side. On the contrary, macroeconomic equilibrium models consider the influence of market responses on marginal mitigation cost in addition to the direct effects of abatement requirement on supply-side. The geographic coverage and scope is usually wider and abatement rate is often reported higher for a given carbon price using equilibrium models than that applying supply-side models; whereas the level of spatial disaggregation or resolution is generally lower (Vermont & De Cara, 2010).

In recent years, there has been a growing interest to apply engineering-oriented bottom-up approaches to analyze the abatement cost and potential of individual measures. This kind of MACC studies (Beach et al., 2008; Moran et al., 2011; Schulte et al., 2012; Pellerin et al., 2013) take into account heterogeneities in terms of abatement potential, applicability and implementation costs of mitigation options. Bottom-up MACCs are constructed by ranking mitigation measures by increasing cost per unit CO_2e abated against their associated abatements to obtain the cumulative mitigation potential.

A comparison of the two approaches (Vermont & De Cara, 2010) indicated that it is inappropriate to tell one approach is superior to another since some aspects are better addressed using models while others are better captured in engineering approaches. Top-down models are generally preferred for assessing the full effects of macroeconomic and fiscal policies for mitigation purposes (Bockel et al., 2012). The top-down approach is also practical for analyzing mitigation potential from sectors with relatively concentrated emission sources and high homogeneities in abatement technologies, e.g. power generation. Bottom-up MACCs account for the feasible choice of abatement options available to farmers and are able to reflect the substantial heterogeneities in abatement technologies as well as the biophysical extent and the different costs of applying these measures across diverse farm systems. This is the primary reason why a bottom-up approach was selected to construct the MACC for Chinese agriculture given the complexity and diversity of its agriculture systems and the predominance of small-scale farms. In addition, the large number of on-farm experiments throughout the nation allows for estimating detailed technical abatement rates representative of and specific to China. Finally, since supply-side models or macroeconomic general equilibrium or sectoral breakdown equilibrium models in China have not integrated carbon constraints modules, we were unable to perform a top-down agriculture MACC exercise.

Despites the various advantages of bottom-up agriculture MACCs, particularly information-rich in abatement options, there are key limits that should be bear in mind when interpreting the results and used for policy decision making. Compared with supply-side models, the effects of mitigation measures on the behaviours of farmers are poorly integrated in bottom-up MAACs. Studies that rely on engineering approaches are also incapable to reflect market feedbacks on prices since price evolution is pre-defined based on a set of hypothesis and independent of mitigation measures to be undertaken. In reality, equilibrium effects considering market responses of a mitigation instrument significantly affect abatement supply or total mitigation potentials (Vermont & De Cara, 2010).

2. Estimation of measure implementation cost

Implementation costs (expressed as ¥/ha for cropland measures) are estimated following Eqn (4-1), i.e. by evaluating per hectare measure-induced changes in yields, input costs (e.g. fertilizer, pesticide, seeds), investment, labor, machinery and irrigation costs, compared to conventional practices under the baseline scenario. Compared to top-down models, in which farmers are supposed to maximize their profits or margins facing certain constraints, the bottom-up approach seeks to evaluate the changes in input/output when farmers adopt a mitigation measure. Therefore, changes in production factors are preset and independent from each other in the bottom-up approach, rather than an optimization of fertilizer use, labour, land and other factors in top-down exercises.

$$C_t = \sum_i^I \Delta C_i = \sum_i^I (C_{Mi} - C_{Ci}) \qquad (4\text{-}1)$$

C_t is the annual net cost (2020 value) of measure implementation for farmers in 2020. i denotes the agriculture input/output items, e.g. fertilizer, pesticide, labor, yields, etc. ΔC_i defines the change in item i of measure implementation in monetary value. C_{Mi} represent typical benefits/costs of agricultural outputs/inputs when farmers adopt a mitigation measure and C_{Ci} is the conventional inputs/outputs without measure implementation.

Estimated costs in 2020 price then need to be converted to values in the benchmark year 2010. Such a process involves the notion of Net Present Value (NPV) which is used in capital budgeting to analyze the profitability of an investment or a project and is usually derived following Eqn (4-2).

$$\text{NPV} = \sum_{t=1}^{T} \frac{C_t}{(1+r)^t} - C_o \qquad (4\text{-}2)$$

C_o is the initial investment, which is assumed to be zero in this research since all investments, simply divided by their lifespan, are transformed into annual costs. r is the discount rate used to discount future cash flows to the present value and is a key determinant; here to convert measure lifetime costs to 2010 values, we adopted a nominal discount rate of 4.5% , equivalent to the five-year deposit rate of the Bank of China. T is the duration of the project, which is 10 years from 2010 to 2020.

2.1. Evaluating benefits/costs implications of adopting mitigation measures

To estimate the typical implementation cost of each identified measure on the national scale, China is simplified as a single farm using the national average costs/inputs data. Costs represent direct costs to farmers in complying with a measure, which is consistent with estimates of the abatement rate only accounting for on-farm emissions. Indirect and social costs/benefits are excluded from the analysis. The former include costs associated with changes in government subsidies and extension service improvement, implying that implementation of public policies is costless or farmers will change their behaviors according to their production margins without anticipating potential subsidy shift. Social costs refer to wider environmental impacts of implementing some measures (e.g. reduced water or air pollution).

A literature review and expert consultation was conducted to determine the on-farm implications and possible costs and benefits of performing mitigation actions compared with conventional/common practices. Aspects taken into consideration for each measure and major references are presented in Annex 7. For information, cost considerations on livestock measures are summarized in Table K in Annex 4. Yield improvement effects of integrated nutrient management measures were drawn from Zhang et al. (2012b) but modified in this study since baseline yields will already reach a higher level in 2020 than in 2010 (partially presented in Table E). Changes in agricultural inputs and production costs induced by measure implementation are summarized in Table 4-1(see Table L for relevant information about livestock measures). Again, please bear in mind that costs presented here do not account for the expenditure of policy enforcement and other hidden costs, which explain the absence of positive cost factors for certain measures, especially measure C1.

Table 4-1 Benefits/costs implications of cropland measure implementation

Measure No.	Target crops	Cost consideration factors (2010 level per hectare per cropping season)						Incurring frequency
		Fertilizer rate* and price	Labor (mandays)	Machinery	Irrigation	Other costs	Yield	
C1	Cereal crops	N rate: rice -15% wheat- 31% maize-16%						Cropping season
	Cash crops	N rate: greenhouse veg. -15% openfield veg. -10% fruit-15%						Cropping season
C2	Wheat	N rate: -20%	7.5				+5%	Cropping season
	Maize	N rate: -18%		¥225			+8%	
C3	Rice	N rate: -20%	15		-20%		+5%	Cropping season
C4	Vegetable	N rate: greenhouse -27%; openfield -24%. Nitrate-based fertilizer (10kg N) price:+60% higher	-15		-40%	Drip irrigation ¥3000; agri. film ¥1000	+10%	Cropping season
	Fruit	N rate: -30%. 17kgN price:60% higher	45				+10%	Annual
	Cotton	N rate: -33%. 17kgN price:60% higher	-30		-40%	Drip irrigation ¥3000; agri. film ¥1000; pesticide -30%	+10%	Annual
C5	All crops	N fertilizer price: 10% higher						Annual
C6	Cereal crops	N rate: rice -11% wheat- 10% maize -9%. Organic manure†: +1.6-2 t/ha at ¥500/t	7.5					Annual
	Openfield vegetable	N rate: -7%. Organic manure:+1.52 t/ha	7.5					
	Fruit	N rate:-11%. Organic manure:+5.16 t/ha	15					
C7	Wheat, maize		-30%	-20%		Seed +10%; pesticide+30%		3 years or 4 years
C8	Wheat, maize	+30kg/ha		¥300		Seed +10%; pesticide+30%		Cropping season
C9	Rice, wheat, maize	20t/ha at ¥1000 /t biochar	15				+10%	Every 5 years

* N rates here refer to those in target regions or balanced N application rates.

† Here the N content in typical organic manure fertilizers stands at 1.2%.

2.2. Forecast of future production input and input price

Values of typical agricultural inputs and outputs for average showcase farms across China in 2010 were obtained from the China Agricultural Products Cost-Benefit Yearbook (NDRC, 1998-2013) (see Table 4-2 taking rice as an example). Since the bottom-up approach doesn't rely on a macro equilibrium model accounting for market feedbacks, future prices of agriculture inputs and outputs have to be projected based on anticipated changing rates. Such price forecast for 2020 is therefore exogenous and independent of the levels of mitigation options taken, which is one of the principal limits of the bottom-up approach. Forecast of agricultural commodity/output prices in 2020 was based on the change rates simulated by the CAPSiM model and is presented in Table E in Annex 3. Maize price will see the largest increase during the period of 2010-2020 among the three main cereals driven by the demand in livestock feed.

Table 4-2 National average inputs (price) of rice production in 2010 and 2020

Item	Unit	Annual nominal increase rate 2000-2010	2010 condition	Assumed annual nominal increase rate 2010-2020	2020 condition
Direct material and service cost/0.067ha	¥	7%	303.93	4%	
1.seed cost	¥	8%	39.74	4%	52.75
2.fertilizer cost	¥	7%	110.94	3%	148.17
3.organic manure cost	¥	1%	9.65	0%	9.36
4.pesticide cost	¥	11%	22.39	5%	72.61
5.agri. film cost	¥	3%	2.34	1%	4.42
6.renting and operation cost	¥	12%	113.19	6%	254.21
machine renting and operation	¥	14%	84.94	7%	206.59
irrigation and drainage	¥	2%	19.08	1%	21.85
water cost	¥	0.4%	6.69	0.2%	10.14
animal power cost	¥	-3%	9.17	-1%	13.08
7.fuel and power cost	¥	22%	0.68	5%	
8.technical service cost	¥	-26%	0.02	-13%	
9.tool and material cost	¥	34%	3.40	17%	
10.maintenance and repair cost	¥	-0.7%	1.57	-0.4%	
11.other direct cost	¥	-43%	0.01	-22%	
Human labor cost/0.067ha	¥	6%	226.90	3%	380.20
1.equivalent family labor cost	¥	6%	206.27	3%	315.68
human input days	day	-6%	6.59	-3%	5.29
labor wage/day	¥	12%	31.30	6%	59.68
2.hiring labor cost	¥	8%	20.63	4%	64.52
human input days	day	-4%	0.34	-2%	0.49
labor wage/day	¥	12%	60.67	6%	130.50
Synthetic fertilizer price	¥/kg	7%	4.92	4%	6.95
Synthetic N fertilizer (pure nutrient) price	¥/kgN	6%	4.05	3%	5.35

Source: China Agricultural Products Cost-Benefit Yearbook 2011

Projection for the coming decade on agricultural production inputs followed the historical trends but the change rates for 2010-2020 are assumed to be half those over the period 2000-2010 for two reasons. First, average grain sale prices to 2020 are predicted to rise at half the rate of 2000-2010 (Table E). Second, energy prices, which are the key determinants of agricultural inputs prices, are anticipated to grow by 4-5% per year beyond 2010 compared with a 10.8% annual growth during 2000-2010 (IEA, 2012).

2.3. Illustration of cost calculation of the measure organic addition to rice paddies

We take the example of measure C6- more efficient recycling of organic manure to rice paddies, to illustrate the cost calculation process of measure implementation. The overall objective is to raise the proportion of organic manure in the total rice N supply to about 30%. In doing so, the inputs/outputs that would be altered include a reduction in synthetic N fertilizers, an increase in organic manure fertilizer and more labor inputs requirements for large quantity of manure application. The next step is to determine the level of alternation and convert them into monetary units.

It is essential to mention that the baseline for this measure is a scenario under which the previous five measures C1-C5 have already been implemented to avoid overestimation and account for measure interactions. This implies that the optimal level (133 kgN/ha) of synthetic N fertilizer rate served as the baseline rather than the 177 kgN/ha under the initial BAU scenario. N inputs from organic manure in 2020 is assumed constant as in 2010 which is 21kgN/ha. If organic manure supplies 1/3 of the total N inputs, there should be a source shift of about 19 kgN/ha from chemical fertilizers to organic manure fertilizers. Raw organic manure needs to be pre-treated and composed before being applied to fields, entailing material inputs and particularly farm labor inputs. These inputs are represented by the market price of organic manure fertilizer. Table 4-3 integrates the information for estimating the organic manure addition to rice paddies.

Table 4-3 Illustration of calculating cost of measure organic addition to rice paddies

Altered items	Per hectare change	Unit Price in 2020	Costs in 2020 (¥/ha)
Synthetic N fertilizers	-19 kgN/ha	5.35 ¥/kgN	-104
Organic manure	‘+19 kgN/ha (=1.62t organic fertilizer with N content at 1.2%)	660 ¥/t organic fertilizer (market price)	1066
Labor inputs	+7.5 man-days	130.50 ¥/day	448
		Total cost	1410

The average cost (in 2020 price) for adopting measure C6- rice is estimated at 1,410 ¥/ha (Table 4-3), which corresponds to 1,116 ¥/ha in 2010 price adopting a 4.5% nominal discount rate. Similar procedures were applied to wheat, maize, other cereals, vegetables and fruits to quantify the implementation cost of measure C6 for each type of crops, which were then weighted by the application area to conclude a measure average mitigation cost.

3. MACC results and abatement scenarios

3.1. MACC construction and results

Cost-effectiveness of cropland mitigation measures

The overall cost of a measure in 2010 prices is the negative of the NPV and is then divided by its abatement rate to calculate the cost effectiveness, i.e., average cost or gain of mitigating one tonne of CO_2e. An important hypothesis that should be bear in mind is that the marginal cost of a particular measure is constant since the whole China is considered one farm and generally national average data are used. The constant average costs are therefore a proxy of marginal cost.

Table 4-4 shows that the most cost-effective mitigation option is improved fertilization techniques in wheat and maize fields (measure C2). Such gains are achievable because revenues from increased wheat and maize yields shall dwarf the increase in labor and machine inputs. In the same vein, implementing measure C4 also brings benefits to famers since irrigation investments and additional labor inputs for split fertilization are estimated to be inferior to the economic gains of increased productivity of high-value cash crops. Despite more seed and pesticide inputs are necessary, reduced labor and machine requirements make conservation tillage (measure C7) an economic available option for mitigating climate change. Although more efficient recycling of organic manure to croplands (measure C6) also offers significant mitigation potential, substantial purchase costs for commercial manure fertilizer or labor requirements for manure composting may prevent its widespread adoption. Costs are also estimated to be high for crop residue incorporation (measure C8) owing to additional machinery use. The highest cost for mitigating one tonne of CO_2e comes from measure C9 of biochar addition owing to expensive biochar products and the large quantity to be applied. Similar information on cost effectiveness of livestock and grassland measures are illustrated in Table M in Annex 4.

Table 4-4 Cost and cost effectiveness of cropland mitigation measures

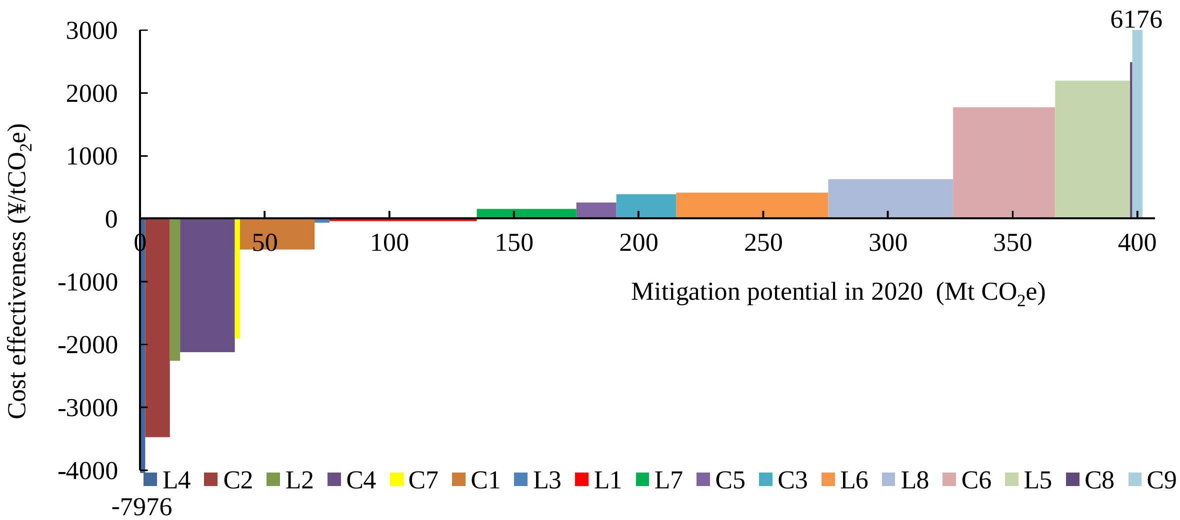
Measure No.	Cost in 2020 (¥/ha, 2010 price)	Cost effectiveness in 2020 (¥/tCO_2e, 2010 price)	Mitigation potential in 2020 (MtCO2e)
C1	-257	-491	30.65
C2	-698	-3475	11.38
C3	523	391	23.98
C4	-2586	-2122	21.86
C5	71	260	15.54
C6	594	1776	40.19
C7	-121	-1906	1.46
C8	79	2489	0.95
C9	2032	6171	3.26

Source: Calculation by the author and incorporated in Wang et al. (2014)

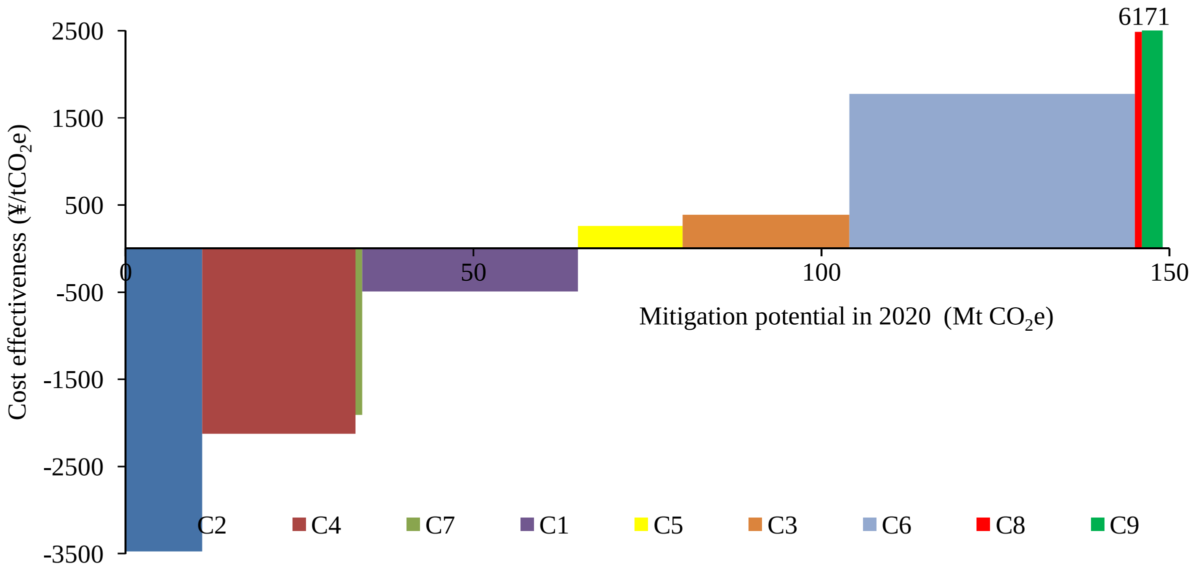
MACC for GHG emissions from China's agricultural sector

Based on data in Table 4-4 and Table M, the GHG emissions MACCs for China's agricultural sector as a whole (Figure 4-3 (a)) as well as for croplands (Figure 4-3 (b)) were constructed. Mitigation options are represented by bars on the plot in order of decreasing cost effectiveness on the x-axis and the bar width denotes the annual mitigation potential of the specific measure. This research concluded a full feasible technical mitigation potential of 149 $MtCO_2e$ from croplands, lower than the potential from livestock and grasslands (253 $MtCO_2e$). Among cropland options, measure C4-Fertilizer best management practices for cash crops is found to source the biggest financial gains and emission reductions as the longest and widest bars beneath x-axis. Measures at negative costs provide over 1/3 of the overall mitigation potential while the fulfillment of another 1/3 abatement potential does incur costs but at a price of less than 400 ¥ (50€) /tCO_2e and the rest mitigation potential are from costlier measures.

Figure 4-3 MACC for China agricultural sector (a) and cropland (b): maximum feasible abatement potential in 2020 (discount rate = 4.5%).



(a)



(b)

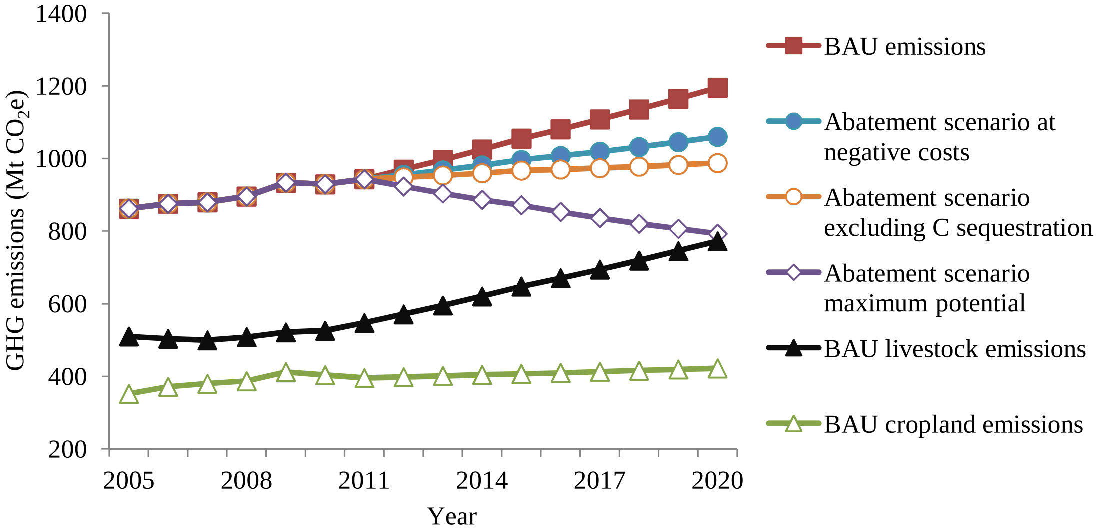
Codes refer to measures in Table 3-2: L4- Probiotics addition to the diet; C2- Fertilizer best management practices (Wheat&Maize) - Right time and right placement; L2- Purebred breeding of livestock; C4-Fertilizer best management practices (cash crops) - Right product, right time and right placement; C7- Conservation tillage for upland crops; C1-Fertilizer best management practices - Right rate; L3- Tea saponins addition to the diet; L1-Anaerobic digestion of manure; L7-Reduction of stocking rate - medium grazing intensity; C5- Enhanced-efficiency fertilizers; C3- Fertilizer and water best management in rice paddies; L6- Grazing prohibition for 35% of grazed grasslands; L8- Reduction of stocking rate - light grazing intensity; C6- More efficient recycling of organic manure; L5- Lipid addition to the diet; C8- Straw addition in upland crops; C9- Biochar addition.

Source: Author and information incorporated in Wang et al. (2014)

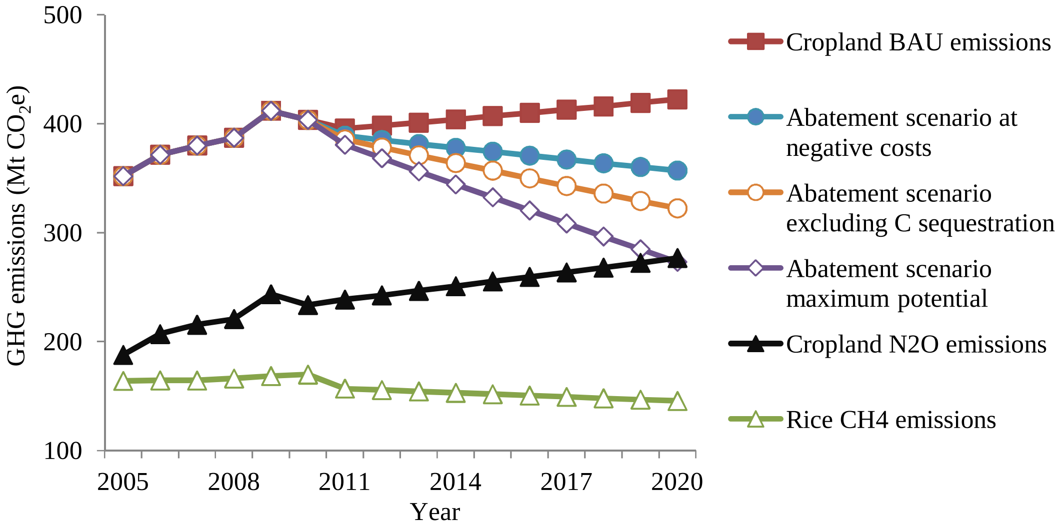
3.2. Abatement scenarios of emissions from China's agriculture

Three abatement scenarios (Figure 4-4) up to 2020 were drawn assuming measures adopted at a linear rate over time: 10% of total mitigation realized in 2011, 20% of total realized in 2012, 30% in 2013 and 100% in 2020. This assumption initially allows us to sidestep a range of possible policy scenarios and instruments incentivizing measure uptake. The maximum feasible mitigation potential scenario implies the maximum physical extent to which a measure can be applied and therefore encompasses the upper potential presented by Figure 4-3 from measures L4 to C9 for the whole agricultural sector. The abatement scenario at negative cost captures the potential from those cost-effective measures, i.e. from measure L4 to L1 in Figure 4-3. Under the abatement scenario excluding carbon sequestration, only the abatement potential from measures targeting CH_4 and N_2O emissions is taken into account.

Figure 4-4 Projected BAU and abatement emissions scenarios for the whole agricultural sector (a) and croplands (b)



(a)



(b)

Source: Author and results incorporated in Wang et al. (2014)

Figure 4-3 and Figure 4-4 show that under the maximum technical abatement scenario for 2020 emissions reductions from the China's agriculture amount to 402 MtCO_2e, representing 34% of BAU emissions. When only counting the measures targeting CH_4 and N_2O emissions, abatement potentials decline to 207 MtCO_2e in 2020, corresponding to 17% of baseline emissions. At national scale about 135 MtCO_2e emissions could be abated at negative costs, equivalent to 11% of baseline emissions in 2020. If fully implemented, these win-win measures result in savings of ¥ 125 billion (2010 price) for farmers. The results suggest that there is significant potential for win-win abatement avoiding emissions while providing financial savings. This analysis does not account for ancillary impacts such as reduced fertilizer production, government subsidies, or reduced environmental impacts. The analysis also shows that 191 $MtCO_2e$ (approximately 48% of the total potential) emissions can be realized at a carbon price less than 260 ¥ (32 €) per tCO_2e.

3.3. Regional disparities in MACC results

The cost-effectiveness levels presented in the MACC graphics are the national average around which fluctuates regional situations in terms of both the per hectare abatement rate and implementation cost. From the technical perspective, our research results on the GHGI of cereal production confirm the regional diversity in mitigation potentials. From the economic perspective, unbalanced provincial economic development implies high discrepancies in implementation cost among regions. For instance, the equivalent salary levels for hiring labors intend to be higher in the more prosperous and productive coastal zone than many interior areas, worsening the cost-effectiveness of certain measures in the coastal provinces. For the purpose of illustrating regional disparities of MACC results, we selected three typical provinces- Yunnan, Hebei and Jilin with high, medium and low GHGIs of crop production. Similar approach for estimating national average abatement rate and per hectare cost was pursued and regional data was mobilized whenever available. It is worth mentioning that, not only agriculture input prices (such as labor wages, fertilizer price) but also output/product prices (such as the wheat selling price) are different among regions with the former having greater regional variations.

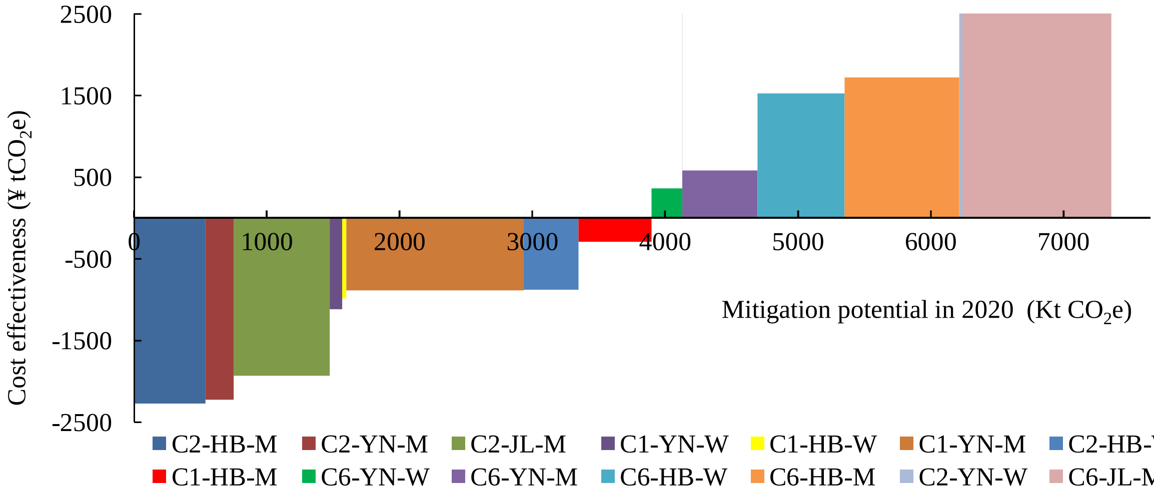
Table 4-5 and Figure 4-5 presents the cost-effectiveness and mitigation potential of measure C1, C2 and C6 for wheat and maize production in the three provinces. Although on the national average scale, measure C2 is more cost effective than C1 and C6, things are different when we descend to individual province and crop: farmers in Hubei implementing C2 for wheat would be economically better off than those in Yunnan applying measure C1. As one of the most-carbon intensive provinces in grain production and therefore offering largest room for GHGI improvement, Yunnan is more cost effective in directly reducing synthetic N fertilizers (measure C1). But the expense for matching fertilizer application timing and location to wheat growth needs (measure C2) will become positive which is negative at the national average scale or in Hebei. This is because benefits from N fertilizer savings and yields increase from low wheat productivity in Yunnan could not outweigh the increase in human labors despite that salary levels of hiring human forces are 33% lower in Yunnan than in Hebei. As to Jilin, both wheat and maize are not targeted by measure C1 but for different reasons: wheat is rarely grown in this province while high efficiency in maize (or relatively low GHGI) exempts it from measure C1. However, when it comes to measure C6 of applying more organic manure to croplands to replace part of synthetic fertilizers, Yunnan is the most cost-effective region given its higher utilization of organic fertilizers in the baseline and lower human wages than the other two provinces.

Table 4-5 Example of cost effectiveness of some measures at the regional level

	Measure C1		Measure C2		Measure C6	
Region	Wheat(W)	Maize(M)	Wheat	Maize	Wheat	Maize
Hebei (HB)	-982	-289	-877	-2270	1524	1723
Jilin (JL)				-1929		3416
Yunnan (YN)	-1117	-886	2595	-2226	365	584
National average	-939	-646	-644	-5107	1519	1811

Source: Author's calculation

Figure 4-5 Illustration of MACC for wheat and maize related measure C1, C2 and C6 in three provinces



Codes in the graphic: HB, YN and JL refer to the province of Hebei, Yunnan and Jilin, respectively, and W and M refer to wheat and maize.

Source: Author

We could find that the implementation cost of each measure differ substantially from one region to another and among crop types and it would probably be economically efficient to begin mitigation in regions with lower costs. The disparities of cost-effectiveness primarily originate from differences in GHGI and the application level of organic manure. Levels and prices of input and output including equivalent labor salary contribute to regional variations but to a lesser extent since their values are more closely distributed around the national average than the GHGI. This brings forward the needs for further regional and local studies which necessitate comprehensive information on the applicability of a specific measure and subsequent cost implications for local farmers.

4. Discussions and sensibility analysis

4.1. Comparisons with other MACC studies

Comparisons with international agricultural MACC studies

In the international literature on agriculture MACCs, measured as ¥(or €/$) per tCO_2e abated, abatement cost represents the typical cost for farmers/farms of complying a mitigation measure compared to the reference case, which generally incorporates public subventions but excludes private transaction costs. The inability of capturing wider social cost implications and behavior change considerations naturally leads to the existence of cost-negative/beneficial measures-reducing GHG emissions and enhancing farm profitability simultaneously. Most of the cost-beneficial mitigation measures mainly relate to adjustment techniques to increase production efficiencies, i.e. measures that maximize outputs with economies of inputs. Nutrient, especially chemical N fertilizers, management practices in croplands to increase NUE are highlighted across studies. Reduced tillage and better residue management are also identified as cost-beneficial options. Abatement potential from cost-beneficial measures range from 34% (this study) to as significant as 3/4 (Schulte et al., 2010) of total agriculture mitigation potential (see Table 4-6).

A review of MACCs literature also suggests that a set of measures are capable of delivering economically viable abatement below thresholds set by a reference carbon price. Grassland management practices generally fall under this type of inexpensive measures since their implementation is assumed to entail small changes to current practices and does not require significant capital investment. Moving rightwards along the x-axis, costs are climbing in excess of the defined carbon price to reduce one unit of GHG emission due to incremental requirements for significant financial investment without immediate payback, important human labors or specific inputs/additives. Lipid addition to the diet or livestock feed supplements are generally deemed as such cost-prohibitive measures.

Mitigation potential results from international MACC studies can be divided into three categories depending on the scope of accounting conventions (Table 4-6). The first category (I) quantifies emission reductions that could be credited to the agricultural sector in the national GHG emissions inventories. National inventories follow principally the IPCC methodologies and generally don't account for the expected mitigation arising from certain abatement actions. This is the case for actions promoting carbon storage in soils (e.g. conservation tillage) as well as for measures aiming at reducing emission factors through enhancing productivity (e.g. enhanced efficiency fertilizers). It worth mentioning that IPCC Guidelines do provide methods on how to account for changes in cropland soil C stocks but most countries opt for non-inclusion of soil C change. These mitigation opportunities are captured into the second category (II), which enlarges the mitigation perimeter by taking into account and therefore delivers higher potential than the category I. Finally, Life Cycle Analysis (LCA) could also be conducted to assess the overall impacts on climate change associated with each mitigation measure (category III).

Table 4-6 Comparison of agriculture MACCs results

Region	Reference carbon price (€/tCO2e)	Mitigation Potential (MP) — of total mitigation potential			Accounting Category — of baseline agriculture emissions			Reference
		MP at negative-cost	0<MP< carbon price	Carbon price< MP	I Accreditable to agriculture in national inventory	II Enlarged scope (e.g. inclu. carbon sequestration)	III Life-cycle Analysis	
China	48	34%	20%	46%	<17%	34%		This study
UK	45	55%	25%	20%		22%		Moran et al. (2010)
France	56	39%	49%	12%	10%	30%		Pellerin et al. (2013)
Irish	24	74%	19%	7%	6%	14%	17%	Schulte et al. (2012)
World	23	50%	37%	13%		13%		US EPA(2013)

Source: Compiled by the author

The most common approach for quantifying agriculture mitigation potential falls into category II and the results range from 13% to 34% of baseline emissions. Strictly following the method for national agriculture inventory, the French MACC study (Pellerin et al., 2013) concludes a cumulative national mitigation potential of 10 Mt CO_2e in 2030, accounting for 9.5% of agriculture emissions in 2010 (the baseline). Using the second category methodology, the potential rises to 32.3MtCO_2e, more than 3 times the result adopting the inventory approach. Under a feasible policy environment, the UK MACC reveals a combined (i.e. crop and livestock) sector total central abatement potential for 2022 (discount rate 3.5%) of 9.85 MtCO_2e, equivalent to about 22% of the 2005 GHG emissions in the UK. This result falls into the category II beyond the scope of inventory accounting. The Irish MACC analysis shows that the total maximum biophysical abatement potential amounts to 2.7 MtCO_2e per year by 2020, corresponding to 13.5% of the estimated 2020 baseline emissions. However, only 1.1 MtCO_2e could be recorded to the agricultural sector in the Irish GHG Inventory. Based on LCA analysis, the potential rises to 3.4 MtCO_2e per year. The US EPA (2013) estimate of total mitigation potential for the agriculture sector worldwide is the least significant in terms of its percentage (13%) relative to defined baseline emissions

However, such a comparison calls for precautions given the differences in scope, context, reference scenarios, modes of emission calculation, mitigation options encompassed, cost assumptions and the sensitivity of these studies. For example, compared to other work, the French MACC study claims to be more conservative which only considers typical mitigation options that are technically available and easy to implement such as fertilization, tillage and legumes practices. This choice may lead to mitigation and cost estimates more robust. The French

MACC also uses a static baseline of 2010 agriculture emissions while others generally adopt projected reference emissions for 2020-2030 based on agriculture activities forecast.

In short, the comparison allows us to conclude that mitigation potential from negative and medium cost accounts for a lower proportion of the total potential in China than in other jurisdictions/worldwide. Secondly, the existence of important negative-cost potential in our MACC results is not specific to China, but rather originated from the methodological limits inherent to the bottom-up engineering studies. Finally, with the overall feasible mitigation potential representing 34% the projected baseline emissions in agriculture, this percentage is the highest of all countries listed in Table 4-6, a significant part of which resulted from the massive reduction in synthetic N fertilizers in China where overconsumption is phenomenal.

Comparisons with domestic MACC studies for other sectors

From an economist's point of view, the most efficient approach in combating climate change is to tap the lowest cost mitigation potential across the full spectrum of economic sectors. In other words, the cheapest units of GHG should be abated first. It is therefore worth comparing the agriculture MACC results with those of other sectors of the economy in anticipation of an efficient economy-wide allocation of future reduction tasks. There are limited numbers of MACC studies for Chinese economy and the literature we refer to is mainly the report by McKinsey&Co (2009a) assessing GHG emissions abatement options in the five major sectors and the work by Xiao et al. (2014) for the building sector.

Table 4-7 Comparison of MACCs results across sectors

Sector	Horizon	Baseline emissions (MtCO2e)	Total Mitigation Potential(MP) (MtCO2e)	Total MP of baseline emissions	Percentage of MP at negative/ neutral-cost	Discount rate	Reference
Power generation	2030	5400	2800	52%	5%	4%	McKinsey&Co(2009a)
Emission-intensive industry	2030	4800	1600	33%	42%		McKinsey&Co(2009a)
Road transportation	2030	1800	600	33%	30%	4%	McKinsey&Co(2009a)
Buildings and appliances	2030	3200	1100	34%	70%	4%	McKinsey&Co(2009a)
Building	2030	2390	500	21%	66%	10%, 15%*	Xiao et al. (2014)
Agriculture and forestry	2030	1040	640(290 from agri.)	62% (26% for agri.)	41% (70% for agri.)	4%	McKinsey&Co(2009a)
Agriculture	2020	1195	402	34%	34%	4.5%	This study

* 15% for residential buildings and 10% for commercial buildings

Some aspects for comparison are summarized in Table 4-6 but the results should be regarded indicative rather than absolute given the different assumptions underpinning each study. For example, the discount rates used in these researches range from 4% to as high as 15%. The McKinsey&Co work is the most comprehensive in terms of covered sector but far less thorough and updated than the Xiao et al. (2014) research. Ten agriculture mitigation technologies were considered in the McKinsey&Co report, which concludes a less significant mitigation potential than the current study for the agricultural sector. We find that the proportion of mitigation opportunities relative to baseline emissions in agriculture is comparable to other sectors and cost-beneficial abatement is of considerable importance. This justifies the proposition to integrate agriculture into the national mitigation strategies to pick those “low-hanging fruits” in the agricultural sector.

4.2. Uncertainties and sensitivity analysis

All available scientific and technical data and information has been mobilized to conclude as accurately as possible the abatement potential and implementation cost for each measure captured in this MACC study. Still, such a process entails a set of uncertainties, some of which are inherent to the bottom-up MACC exercise while others are specific to the assumptions made in this study. Uncertainties arise from three aspects: the per hectare abatement rate, the feasible extent of measure adoption in 2020 and the unit cost of measure implementation. The first two uncertainty sources principally originate from technical obstacles impeding a more profound investigation of measure abatement potential and applicability, for example on the provincial scale. These uncertainties will reduce along with the progress in scientific research and technological evolution. Uncertainties in costs are mainly related to the assumptions made on changes in various inputs/outputs and price forecast. Although it is ideal to use sensitivity analysis in the face of key uncertainties or data discrepancies in all stages of MACC construction, here we focus on a sensitivity analysis to assess the robustness of cost estimates.

If we assume no alternation in input/output prices of agriculture production, in other words maintaining prices in 2020 identical as in 2010 after discounting, the overall cost shall increase but consequences are different among measures. Most measures see minor changes in their implementation cost, such as measure C1, C5, C6, C8 and C9. Measure C2 on managing the time and location of fertilization becomes more cost-effective by 57% while the per hectare cost of measure C7 on returning straw to uplands shall be 35% higher. Measure C4 (Fertilizer best management practices of cash crops) sees the largest change in cost-effectiveness from -2122 ¥/CO_2e to -248 ¥/CO_2e but remains cost-beneficial. The particular case is measure C3 (Fertilizer and water best management in rice paddies) which reverses its profitability from cost-prohibitive (391 ¥/CO_2e) to cost-beneficial (-179 ¥/CO_2e) in abating one tonne of CO_2e. This suggests that depending on the production aspects they intend to modify, mitigation measures are sensible to different price factors to varied extent.

We choose first to vary the equivalent salary level of human labors to examine its effects because Labor equivalent salary is an essential determinant given the phenomenon of increasing labour scarcity in rural areas. In the current study, when a measure entails additional labor forces, the per day hiring wage instead of equivalent family labor salary was used to account for the opportunity cost due to the insufficiency of work forces in rural China. A 5% increase in the labor wage will result in a 126% increase in the total implementation cost and a 10% increase leads to 256% higher costs. On the other side, if the labor wage decreases by 5% and 10%, overall cost will decline by 16% and 106%, respectively. C3 is the measure most sensitive to alternations in labor payment, the 10% increase in labor wage leads to a 30% increase in its implementation cost. If nominal labor wages grow at an annual rate of 2.4% relative to the 6% rate in research, adopting measure C3 will entail no additional burden to farmers. But such a low increase rate scenario is unlikely to take place given the gradually serious problem of labor scarcity in rural areas.

Since most measures target a modification in synthetic N fertilizer use, a change in chemical N fertilizer price would certainly affect the cost of each measure. During the timespan of 2010-2014, the N fertilizer price begun to decline after a peak in 2008/2009 owing to the overcapacity of N fertilizer plants and the government's various subsidies (more details in section 2.4 of chapter 5). If the declining trends persist, e.g. N fertilizer price (nominal value) remains stagnant from 2010 to 2020, measures (C1, C2, C3, C4 and C7) aiming at reducing the N use will become economically less attractive while C8 will be cheaper to adopt. C1 is most sensitive to variations in N fertilizer price since its mitigation cost is exclusively determined by the fertilizer price. We can conclude that higher N fertilizer prices will be more conducive to encourage the adoption of mitigation measures provided that farmers are rational and well informed on the optimal fertilizer rates and techniques. This means the government needs to shift from the traditional policies on heavy subsidies encouraging massive N fertilizer use. Otherwise, farm-scale use of chemical fertilizer use will not respond to an increase in fertilizer because of the low inelasticity of fertilizer demand price from past experience (Farquharson et al., 2010; Zhang, 2012).

Output price is another variable to be tested which will greatly affect the cost–effectiveness of those measures increasing yields. In the future, if output selling prices are 10% higher than those in the current study, the cost-effectiveness of C2, C3, C4, C6 and C9 will rise with C4 being the most sensitive of 24% change.

Finally, we test the effects of different private discount rates which reflect agents' perception of opportunity costs. When we vary the social discount rate from the current 4.5% to 3% (Bank of Saint Louis), the average cost shall increase by 7.8% while decreasing by 5.6% when a higher discount rate (5.7%) is taken, which is equivalent to the five-year average Loan Prime Rate defined by the Bank of China.

4.3. Significances and limits of MACC studies

Significance of MACC studies

This research work on MACC analysis represents the first attempt to derive a bottom-up evaluation of technical and economic abatement potential for the agricultural sector in China. The findings provide an overview of important low carbon options in agriculture and their cost effectiveness. Although low carbon development planning always involves more considerations than merely selecting the measures with the least additional costs or largest abatement potential, MACC results suggest an initial indication of priority interventions in the design of efficient policies. It also enables the comparison of economic availability of agriculture mitigation options with a benchmark carbon price and abatement in the other sectors of the economy. The investigation on economic potential is timely and potentially paves the way for identifying an agricultural contribution to national GHG reduction targets, either through carbon taxation or offsetting projects or eventually as part of other trading arrangements, which will be fully discussed in chapter 5.

The MACC construction itself suggests numerous research priorities in terms of tailoring practices to local biophysical conditions, thus allowing a more accurate estimate of measure cost-effectiveness. For example, straw returning modes vary significantly among regions, including direct incorporation, straw mulch and incorporation, burying into ditches, fast decomposition before returning and others. The Chinese government has already initiated programs to improve domestic research in the field of climate change mitigation and agriculture. For example, the ongoing research project “Integration and demonstration of key carbon sequestration and mitigation technologies in agricultural ecosystems” accredited by the Chinese Ministry of Science and Technology for the 12th Five-Year Plan period, aspires to identify appropriate mitigation measures for major cropping systems, to quantify abatement rates and to model mitigation potential at the regional level.

Limits of MACC analysis

The MACC study was conducted using a bunch of data, assumptions and experimental evidence which is currently limited and may imperfectly reflect the real biophysical heterogeneity in agricultural systems. These include the assumptions about baseline activity projections (including input and output prices), measure abatement rates, their spatial applicability, implementation costs and adoption levels. The MACC exercise aims to make these assumptions transparent and therefore provides a basis for on-going improvement of technical and economic mitigation estimates.

Since measure costs in this study represent typical average values across China, errors may source from two aspects. Firstly, on-farm practices under a mitigation measure are not unique, but rather should be tailored to local circumstances, thus entailing varying economic implications for farmers in different regions. Secondly, agricultural input prices and rural labor wages vary significantly among regions determined by local economic development levels as we show in section 3.3. The cost effectiveness of mitigation options will also change in response to factors such as R&D investment, learning effects, economies of scale and the indirect effects of non-GHG policy.

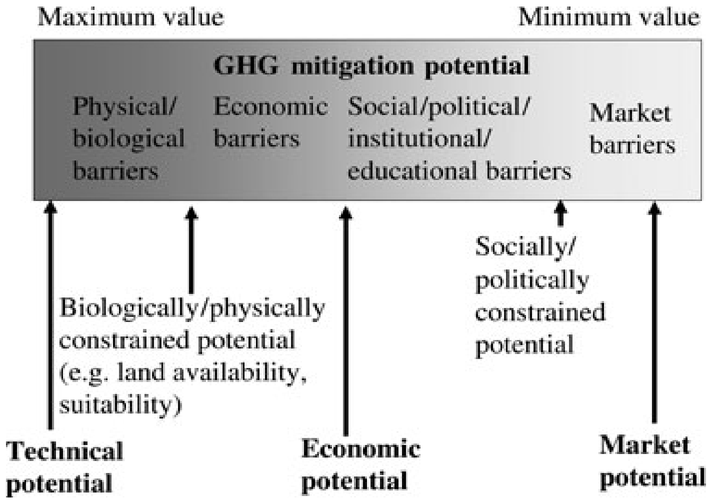
In addition, this study did not include a quantitative assessment of indirect and ancillary benefits and costs of identified measures. For example, more efficient use of N fertilizers and organic manure are positively correlated with water quality. On the other side, reversing farmers' traditional practice of high reliance on N input necessitates huge government investment and efforts in educating farmers and advancing rural extension service systems. Consideration of these benefits/costs shall largely alter the cost effectiveness of measures, but remains challenging to be quantified in monetary terms.

Barriers to measure adoption

The overall technical potential in the MACC graphic represents the total abatement that can be realistically achieved wherever biophysically possible. By setting a reference carbon price, the economic potential can be derived. Unlike the supply-side or equilibrium models, the engineering approach concludes some measures permit simultaneous cost and emissions reduction. The important share of abatement from negative-cost measures appears puzzling from an economist's perspective as we wonder why such profitable measures are not already implemented by farmers in the baseline. There are several possible explanations to why these apparently unrealized savings exist. This brings forward the notion of market potential which is the potential actually seen under current market conditions (Smith, 2012; Figure 4-6).

First, farmers have entrenched views on the links between inputs and yields (Wu et al., 2011) and are generally risk-averse faced with new technologies and practices. Second, given the small scale of Chinese farms, savings from rationalizing N application rates are perceived to be relatively insignificant by farmers, particularly when fertilizer prices are kept low by subsidies (Zhang et al., 2013). Third, increasing rural labor shortages raise the perceived opportunity cost of the time required for mitigation activities. Fourth, weak agricultural infrastructure and poor rural extension services are a hindrance to measure adoption. For example, although scientifically justified fertilizer recommendations have been developed for major crops and cropping systems (Zhang et al., 2009), the absence of good extension advice hinders information dissemination to millions of smallholder farms widely distributed with low levels of mechanization. In the same vein, the poor supply of artificial insemination services to livestock farmers can be attributed to large distances between farms. A solution for this would be the implementation of a tight grid of breeding farms to cover the whole country. Alternatively, the challenge of implementing more efficient and environmentally sound practices could be solved by the ongoing consolidation of agricultural land and more ambitious government investment in infrastructure. More detailed behavioural and institutional challenges are addressed in relevant mitigation (e.g. Moran et al., 2013; Zhang et al., 2013), some of which are specific to the structure of Chinese agriculture and its role in national policy on both food security and rural development. How a carbon pricing mechanism may be helpful to overcome these obstacles will be elaborated in chapter 5.

Figure 4-6 Relationship between technical, economic and market GHG mitigation potential



Source: Smith (2012)

5. Conclusion

This chapter moves from a technical potential to an estimate of feasible economic mitigation potential of GHG emissions from Chinese agricultural sector, with a focus on croplands. The methodological approach involves the use of a bottom-up engineering MACC, which offers a rational to combine biophysical and economic data, to reflect the mitigation costs. And it also allows aggregation of the mitigation potential arising from the application of a subset of cost-effective measures above a notional baseline level of activity, i.e., the BAU scenario.

The MACC results demonstrate that while the whole agricultural sector offers a maximum technical potential to cut 402 $MtCO_2e$ in 2020, a reduction of 135 $MtCO_2e$ is potentially available at zero or negative cost (i.e. a cost saving), and 176 $MtCO_2e$ (approximately 48% of the total) can be abated at a cost below a threshold carbon price of less than ¥ 260 (approximately €32) per tCO_2e. About half of the maximum potential is estimated to be realized through carbon sequestration in agricultural and grassland soils. In the arable sector, the full technical potential results in a cut of 149 $MtCO_2e$, of which 65 $MtCO_2e$ could be abated with gains to farmers. Best fertilizer management practices for both grain and cash crops are the largest source of cost-effective mitigations. Better manure recycling practices to croplands could also provide sizable mitigation but requiring higher economic compensation.

We also outlined the various assumptions underlying the MACC construction, which will inevitably introduce uncertainties to the MACC results. Different hypothesis on economic and social development levels would affect agricultural production levels and prices of its inputs/ outputs; therefore it could alter the cost-effectiveness of mitigation measures. More regional research is also needed in the future to fix the variables formed by the significant heterogeneities in Chinese agriculture systems and mitigation practices. Nevertheless, the MACC results conducted here can lead the first step to identify the best mitigation actions that should be integrated into the plan in priority for sustainable and low-carbon agriculture development. Furthermore, it calls for development of appropriate policies and instruments (including economic incentives) in a bid to accelerate the transformation of economic mitigation potential to realizable market potential, taking account of current agriculture policy environment and market conditions.

Chapter 5 - How can economic incentives trigger abatement in Chinese agriculture?

The constructed MACC reveals information on the abatement potential and cost-effectiveness of mitigation measures in the agricultural sector. Realizing the identified mitigation potential raises the question about which tool or combination of instruments should policy makers take to achieve overall efficiency. There are several instruments available in the economists' toolbox to integrate the cost of environmental damages into the economy. Economic instruments aim at addressing the negative externalities of excessive GHG emissions by putting a price on carbon. They can be either price-based such as carbon taxes or quantity-based such as tradable permits, generally called cap & trade or emission trading schemes (ETS).

A carbon tax is a tax levied on the carbon content of the products used in the economy to create incentives for lowering GHG emissions. The tax rate is set by the regulator and may be subject to adjustments over time. In a cap & trade system, the central authority fixes a cap on the amount of emissions and allocates or sells these emission permits to mandatory participants. Regulated entities under the scheme have to cover their emissions by a sufficient number of permits. Emitters with excessive emissions may buy permits from other sources that require fewer permits. Such demand and supply of emission permits therefore determines a carbon price on the market. A cap & trade scheme is often linked with an offset mechanism which allows covered entities to use credits generated by projects reducing or sequestrating emissions outside the scheme's perimeter, for compliance under the cap & trade scheme. In principle, offsets allow a transfer of knowledge and technology and unlock investments outside the scheme while the scope of possible mitigation options is expanded, thereby reducing compliance costs within the scheme. Project-based offsets also underpin the voluntary carbon markets.

This chapter will first present the fundamentals of using economic instruments in limiting GHG emissions and the practical experience of ETSs and carbon taxes, particularly agriculture's involvement (section 1). We will then look closely into China's move towards pricing carbon domestically and the negative implications of fertilizer subsidies in China (section 2). Section 3 will examine the specificities and difficulties of putting a price on carbon in the agricultural sector and propose possible solutions. Based on the policy context and sectoral challenges, we will put forward some suggestions on the possible ways to use market-based instruments to effectively mobilize mitigation potential in agriculture (section 4). Section 5 will summarize the propositions to reorient incentives of Chinese agricultural policies through the mechanisms of carbon pricing.

1. Pricing carbon in agriculture from theory to practice

1.1. Theoretical background on integrating environmental issues into the economy

The theory of using economic instruments to address environmental issues rests on the concept of externality, which was introduced by the English economist Arthur C. Pigou (1920) and underpins modern welfare economics. Environmental externalities refer to the uncompensated environmental effects of production and consumption on a third party outside the market mechanism. In the presence of negative externalities, private costs of production don't account for the social damage cost, leading to over-production or over-consumption of this product rather than achieving the optimum. Pigou argues that this problem can be effectively addressed by imposing a tax to the market activities entailing negative externalities; or in other words, polluters should pay for the damages they have created. As such, the market activity will automatically adjust to a new and healthier equilibrium. A carbon tax which addresses the negative externality of excessive GHG emissions hence falls under the classification of a pigovian tax.

In Pigou's theory, it is assumed that public authority is able to determine the marginal social cost of a negative externality to efficiently correct market failures. However, in practice, many economists challenged the possibility to accurately measure the social costs of any externality and determine the optimum output level. The effectiveness of direct government intervention was particularly questioned by the British economist Ronald H. Coase (1960). He advocates that negotiations and bargaining could lead to an efficient outcome without the need for a third party intervention regardless of the initial allocation of property as long as the property rights (including the rights to pollute) are well defined and the transactions costs are sufficiently low.

Coase also admitted that transactions costs, however, could not be neglected, and therefore, the initial allocation of property rights often mattered. There are generally two normative conclusions drawn from the Coase theorem. One is that property rights should initially be assigned to the actors with lowest cost to address the externality. Another is that the authority should minimize transaction costs so that misallocations of resources could be corrected in the cheapest way.

Coase's work laid the foundation for the concept of tradable pollution rights, considered as a kind of property rights. In an emissions trading or cap & trade system, the authority sets a total number of permits (the cap) and allocates or sells them to covered entities. Liable emitters are required to hold a number of allowances equivalent to their pollutions/emissions levels. The permits are allowed to be transferred and such market trading practices will determine the price of the permit. In theory, since entities have the flexibility to use least-expensive solutions to comply with their assigned obligations, a given level of abatement target can be achieved in the least-cost.

In later years, Demsetz (1964), Crocker (1966) and Dales (1968) gradually conceptualized the instrument of emission trading and proposed using this tool for air and water pollution control. Later on, cap & trade systems were further developed and formalized by Baumol & Oates(1971) and Montgomery(1972) and was first endorsed by government authorities in the USA as part of the US Acid Rain Program of the 1990 Clean Air Act. Since then emissions trading is increasingly being used as an environmental policy tool for pollution control.

The American economist Martin L. Weitzman, in his paper Prices vs. Quantities (1974), attempted to characterize the situations in which price-based policies or quantity-based policies would be proven more preferable under certain conditions or in a particular context. In a situation of ambiguity, Weitzman demonstrates that the best choice depends on the respective slopes of damages and abatement costs. In general, the advantage of a carbon tax lies in the visibility of the price that is known to affected agents and allows them to integrate the cost into their production/consumption decisions. But it is generally difficult for the government to access the data on abatement cost to determine an accurate tax rate. As a consequence, the environment target may not be guaranteed and it is hard to anticipate the effects on the rest of the economy. Another disadvantage of a carbon tax is the inflexibility to adapt to changing context. An ETS, on the other hand, guarantees the achievement of environmental objectives but the price is unpredictable since a carbon market is subject to changes in economic parameters which may disrupt the basic functioning of the market.

1.2. States of carbon pricing around the world

Although carbon trading versus carbon taxation is a much debated and documented issue, the two most prominent carbon pricing instruments (or a hybrid of them) have both been in operation around the world to achieve emissions reduction in a cost effective manner. The World Bank's State and Trends of Carbon Pricing 2014 (World Bank & Ecofys, 2014) reports that in 2013 about 40 national and over 20 sub-national jurisdictions worldwide have launched or plan to initiate carbon pricing initiatives. Together these carbon pricing schemes cover nearly 6 $GtCO_2e$, accounting for about 12% of the annual global GHG emissions. As more nations are taking concrete steps forward on putting a price on carbon, the share of GHG emissions covered by carbon pricing instruments is increasing, proving mitigation actions are underway despite sluggish progress at ongoing international climate negotiations.

Eight new carbon markets, i.e. California Cap & trade Program, Québec Cap & trade System, Kazakhstan Emissions Trading Scheme, five Chinese piloting ETSs entered into operation in 2013, increasing the total value of the world's ETSs to around US$30 billion. With a cap of 2,084 $MtCO_2e$ in 2013, the European Union ETS is still home to the largest carbon market in the world which started in 2005 and entered the Phase III in 2013. China now houses the second biggest carbon market with the operation of all the seven carbon trading pilots by July 2014 (more details in section 2). Other carbon trading programmes already put in place include the Switzerland ETS, the Regional Greenhouse Gas Initiative (RGGI) in nine Northeast and Mid-Atlantic states of the US, the Alberta Greenhouse Gas Reduction Program in Canada, the New Zealand ETS and several schemes in Japan. "While overall progress at the national level in China and the United States may take some time, it is remarkable that the world's two largest emitters are now home to carbon pricing instruments." However, on the other side, the Australian government's plan to repeal its Carbon Pricing Mechanism legislation as well as Japan, New Zealand and Russia's official statement to withdraw from the second commitment period of the Kyoto Protocol, are significant setbacks of progress on climate actions.

The other prominent economic instrument - carbon tax, has been introduced in some economies (particularly Northern European countries) at a national level for about twenty years, especially to regulate diffuse emissions (Elbeze & de Perthuis, 2011). Carbon taxes can be put in place alongside an existing carbon pricing instrument, such as in Ireland and France where carbon taxes aim to capture the emissions not covered under the EU ETS. Or, countries or sub-nations may choose to pursue a carbon tax instead of an ETS, such as South Arica. In most cases, when a carbon tax is imposed, other types of tax (generally energy or income) are lowered to maintain the overall tax burden (e.g., Denmark and Finland) with the exception of Sweden. In the meantime, entities that fall under the EU ETS are gradually being exempted from carbon taxes to avoid the potential loss of competitiveness due to double carbon pricing burden.

Carbon pricing initiatives are projected to gain more ground with additional markets planned to emerge in other regions such as South Korea where the ETS is due to start from January 2015. Brazil, Chile, Costa Rica, Mexico, North American Pacific Coast (Oregon and Washington), Russia, Thailand, Turkey and Ukraine are also considering introducing carbon pricing schemes. Some are moving steadily towards an ETS while others are assessing the effects of both cap & trade schemes and carbon taxes to come up with the most appropriate policy options.

1.3. States of carbon pricing in agriculture

Agriculture in cap & trade systems

ETS systems in operation or to be launched around the world all exclude agriculture with the only exception in New Zealand (NZ) where agriculture was originally anticipated to fully enter into the national ETS from 2015. Such an inclusion is justifiable since agriculture accounts for nearly half of NZ's GHG emissions. However, although agricultural participants are still required to report their emissions, the start date for surrender obligations for biological emissions from agriculture has been removed from the latest regulation (NZG, 2012). Acknowledging the difficulty in recording attributable emission reductions, the Government indicated "*biological emissions from agriculture will only incur surrender obligations only if mitigation measures are* *technologically and economically viable and the trading partners make more progress on addressing their emissions."*

Overview of agriculture offsetting projects

Credits issued from offsetting projects can be classified into two categories distinguished by the purpose of their use: the first is accepted by regulatory compliance markets (Kyoto and/or (sub-) national schemes) and the other is transacted on a voluntary basis for emission compensation. The CDM and the Joint Implementation (JI) under the Kyoto Protocol are the two standards used to certify emission reductions for international compliance offsets. The Carbon Farming Initiative (CFI) launched by the Australian government aims to encourage farmers and landholders to undertake agriculture and landfill emissions reduction projects which can be issued credits eligible for the nation's cap & trade scheme. The Alberta Offset system also gives access to agriculture and land use activities with livestock feed, CH_4 capture, soil tillage and other mitigation protocols under consideration. The Climate Action Reserve (CAR) and the American Carbon Registry (ACR) have shifted from voluntary to partial compliant standards since some project types have been recognized as eligible compliance offsets in the California's ETS. On the voluntary carbon markets, nearly all offset projects have adhered to a third-party standard for crediting and the most used standards are the Verified Carbon Standard (VCS), the Gold Standard (GS), the Chicago Climate Exchange (CCX), the CAR and the ACR (Peters-Stanley et al., 2013). We provide in Table 5-1 a summary of registered carbon offset projects related to agricultural activities and the average annual credits that were issued under each sub-sector.

As of May 2014, about 1,067 offset projects associated with agricultural activities had been registered under a carbon certification program/standard, with an average mitigation effect of 28 $MtCO_2e$ each year. Foucherot & Bellassen (2011) estimated that agricultural projects accounted for about 14% of total registered projects across all sectors but issued credits representing only 7% of the annual total. The 28 $MtCO_2e$ mitigation benefits delivered by these projects are minor compared to the huge abatement potential identified for the agricultural sector (IPCC, 2007).

Over 95% projects were developed in the three sub-sectors of bio-energy from agricultural residues, CH_4 recovery from livestock manure and soil carbon sequestration, which don't necessarily correspond to the scales of emissions of and mitigation potential from each sub-sector (e.g. improved N fertilizer use can deliver important N_2O mitigation benefits but are almost absent in the offset markets). The highest annual emission reductions issued from the soil carbon projects, exclusively under the CCX and JI, but the former of which has become inactive since 2013. CCX soil projects involve no-till practices and the conversion of cropland into meadows, and a sequestration rate (e.g. 0.5-1.5 tCO_2e/ha/yr for no-till) was assigned to quantify credits instead of field soil measurement. Bio-energies produced from agricultural residues (e.g. bagasse, rice husks, mustard-seed crop) come at the second place, with an average of 9.8 Mt

CO_2e delivered each year mainly by CDM projects. The third largest project type is livestock manure management, spreading among all standards, which mainly involves biogas generation while a few targeting manure composting treatment. Other types of projects account for a tiny proportion of the agricultural offset markets.

Table 5-1 Average annual credits issued (ktCO_2e) and number of agricultural projects registered under each standard (as of May 2014)

Sub-sector / Standards	Fertilizer use	Manure management	Bio-energy from agri. residues	Agro-forestry	Energy consump.	Soil carbon sequestration	Irrigation	**Sub-total**
CDM		4577 (246)	8776 (445)				0 (1)	**13353 (692)**
CDM PoA		0 (17)	18 (3)				0 (1)	**18 (21)**
JI		218 (4)	203 (9)		1222 (2)	4209 (7)		**5851 (22)**
CCX		388 (58)	651 (20)			6455 (65)		**7494 (143)**
VCS		216 (10)		0 (1)	27 (1)	0 (1)		**243 (13)**
CAR		252 (104)						**252 (104)**
ACR		6 (2)						**6 (2)**
GS		455 (18)	199 (4)					**653 (22)**
EDF China projects	78 (2)	44 (3)					138 (1)	**260 (6)**
Sub-total	**78 (2)**	**6155 (480)**	**9847 (481)**	**0 (1)**	**1249 (3)**	**10664 (73)**	**138 (4)**	**28131 (1025)**

Note: figures in the brackets represent the number of registered projects. Not all registered projects have been issued credits. Projects coordinated by the American EDF (Environmental Defense Fund) in China has not sought third-party certification; but were rather monitored and verified by domestic research institutions.

2. States of carbon pricing in China

2.1. Development of carbon emission trading schemes in China

From inception to implementation of domestic carbon markets

Until mid-2013, China was familiar with carbon trading only insofar as it had participated as a major supplier of international carbon offsets through the CDM framework but no domestic carbon pricing mechanism had ever been in place, with the exception of some voluntary initiatives. The past experience (section 3.2 of chapter 1) indicates that the potential of command-and-control regulatory approaches, which were until then extensively employed to deal with energy efficiency and conservation issues, had shown their limits. Hence, reaching for stronger and sounder commitments on emissions and energy consumption required additional policy measures, such as economic instruments which put a price on carbon to incentive emission reduction practices. Among the menu of possible policy instruments, a carbon trading scheme was appealing to the Chinese authorities in that it offered flexible options for companies in their compliance strategies while preserving some control for the authority, notably in incentivizing firms towards low carbon investments over the long term. Indeed, a domestic ETS has been eyed since 2009 when the NDRC first expressed its desire to test carbon trading through pilot schemes.

The 12th FYP is the first official document that explicitly pinpoints carbon trading as a central policy measure in economic restructuring and shift of growth pattern. In October 2011, the NDRC picked up five cities (Beijing, Chongqing, Shanghai, Shenzhen and Tianjin) and two provinces (Guangdong and Hubei) as the seven pilots to test carbon emissions trading (NDRC, 2011). The national government entrusted the local authorities with the responsibility to design their own ETS based on soft national guidelines so that various ETS plans could roughly converge on many design elements but there would be great leeway left for specific details to accommodate regional circumstances. This diversity in features also allows for the pilots to provide feedback on different design elements and on how carbon pricing affects regional economies. Drawing on these local experiences, the national government should be better prepared to design the features that a potential national ETS ought to contain, should such a roll-out come to life.

Current status of development of regional ETSs

As of August 2014, all seven ETS pilots have started operation. Annex 8 briefly summarizes the market design features, some of which may vary widely, reflecting diverse circumstances and priorities in the localities where ETS are implemented. Here we debrief major pillars of ETS features and highlight differences in market design across pilots. For more details on the market design and development, readers can refer to Jotzo (2013), Quemin & Wang (2014), World Bank & Ecofys (2014) and Zhang et al. (2014).

Coverage. Akin to existing ETSs around the world, pilots account for only CO_2 emissions with the exception in Chongqing where the six GHGs are covered. ETS pilots in China cover emissions from fuel use and combustion in the power sector and (heavy) industries (cement, iron, steel, etc.). Apart from these traditional sectors, other covered sectors vary greatly between schemes but what is really interesting is the inclusion of buildings or transport in some places. How and to which extent it is done will be interesting to follow. Public and commercial buildings are to be included in Beijing, Shenzhen or Tianjin. With the exception of the Shanghai ETS which includes aviation, transport is generally excluded from the scope for now but may be included in the future. While some pilots like Tianjin/Chongqing/Hubei concentrate on a limited number of energy and industry sectors, almost all sectors of the economy are capped in the Shenzhen ETS. In Tianjin, Chongqing, Guangdong and Shanghai the inclusion threshold is 20 ktCO_2e per year while cut-offs are lower in Beijing and Shenzhen, so that, on the whole, more companies are enrolled in the two pilots when compared to the size of each market. It is suggested that a potential national scheme is more likely to resemble that of Hubei covering only industry and electricity/heat entities with the highest inclusion threshold (120 ktCO_2e/yr) rather than the innovative Shenzhen. Agriculture is not in all the seven ETS piloting programmes.

Emission Caps. Despite that national and regional climate/energy targets are expressed in intensity basis, the NDRC requires pilots to put absolute caps on emissions. Determining an absolute cap from intensity targets is quite a challenge in itself since different production projections must be made for each sector. Disclosed caps cover roughly 35 to 60 percent of each region's emission totals. Beijing and Guangdong are the only two pilots that enforce explicit absolute emission reductions. For instance in Beijing firms in the manufacturing and service sectors will see their allocation shrink each year, from 98% of 2009-2012 emissions in 2013 to 94% in 2015. Caps are not always given and when known the figures often lack accuracy and local governments have yet to disclose their calculation methods.

Allowance allocation rules & revenue use. In general pilots have opted for free allocation based on grandfathering for most permits. Although historical emission periods used to calculate these allocations differ across pilots, they all only span a few years (2009-2012 at most) due to limited available data. When specified, benchmarks are used to treat the case of new entrants. Last but not least in Shenzhen allocation is partly output-based. While liable companies are generally endowed with permits on an annual basis, there is only one one-off allocation for the whole pilot phase in Shanghai and potentially in Shenzhen so that liable firms can smooth their use of permits over the whole period. Another interesting feature is early reductions rewards: in Shanghai firms can earn extra allowances for early actions while in Hubei 20% of all free permits are granted on an early-reduction basis.

Auctioning is often mentioned as a complementary allocation method whose share should increase over time. For the time being though only Guangdong requires 3% of its annual cap to be auctioned and has completed several sessions of auctioning. Participation at auctions is mandatory since liable firms are required to first buy the 3% of allowances set to be auctioned before being endowed with the remaining 97% of free permits. The government said it intended to bring the revenue from auction for general financial management. The Hubei provincial government also auctioned 2 million credits set a price of 20 yuan per permit two days ahead of the official launch of the ETS. The interesting case is Shanghai which held an auction on June

30th -the final day for allowances surrender, to help those in short of quotas acquire their missed allowances.

New entrants' reserves amount to about 6% (or 20 Mt) and 2% of the overall cap in Guangdong and Shenzhen, respectively. Other pilots have yet to disclose how they intend to allocate permits to new entrants. In case of closure or relocation of activity, compliance of past activity should be completed and, when specified, 50% of the following year allowances shall be taken back so as to minimize the incentive for old plants to receive allowances and then shut down to sell permits. In the same vein, quotas shall be reallocated when activity changes – whether reduced or increased – beyond a predefined range.

Price management mechanisms. Generally speaking pilots have chosen to manage price volatility through a variable permit volume meaning that the authority can intervene in the market by selling more (if prices spikes) or buying back permits (in case of oversupply). It involves the creation of both an allowance reserve by setting aside a share of the cap and a monetary fund dedicated to market intervention. The size of the fund is not given, except in Hubei, nor is its origin, except in Shenzhen where auction proceeds are explicitly dedicated to this purpose. In the same vein, the size of the allowance reserve, or the limit of governmental intervention as a share of the cap, is not always detailed, as in Beijing or Shanghai. In Tianjin, 15% of the annual cap is set aside in the reserve. Similarly in Guangdong, 18 million quotas of the annual cap are moved to the reserve. In Shenzhen, lastly, the reserve is made up of 2% of the annual cap plus leftover from auctions and government-purchased allowances.

Banking is always allowed during the pilot phase, except in Hubei where both banking and borrowing are explicitly proscribed. Symmetrically to banking, borrowing allows liable entities to use allowances from future compliance periods in advance. Explicit borrowing is forbidden in all pilots, but the exception is Shanghai where participants have already been endowed with free permits for the whole pilot phase (one one-off allocation for 2013-2015).

Offsets. Allowed offsets include credits issued from the national scheme (i.e. the CCER to be fully discussed in section 2.2) as well as locally approved offset projects in certain places. Generally speaking, compliant entities are allowed to meet up to between 5% and 10% of the emissions obligations with government-approved offsets. On top of that a certain amount of CCERs must originate from the region where the ETS is located in. For the moment, Guangdong, Hubei and Beijing require at least 70%, 100% and 50% of offset compliance to stem from local projects. In addition to the quantitative and location restrictions, some pilots also imposed qualitative limits on offset types. Beijing rules out certain types of projects owned by liable entities but allows local carbon credits from energy conservation and forestry projects (i.e. credits not seeking CCER accreditation). Chongqing excludes hydropower from eligible project types. At the time of writing, only a few advance purchases of future CCER have occurred. The latest trade to date has been settled at around 15-20 yuan per offset, which stands around the allowance prices.

Quemin & Wang (2014) estimated that the maximum CCER demand to be around 55 Mt for the 2013 compliance. For the seven schemes taken together, for the 2014 and 2015 compliances, the annual allowed amount of usable CCERs would add up to a maximum of 85 Mt. Given the number of projects that could apply for CCER issuance, future supply is foreseen to be huge in comparison with annual demand.

Monitoring, Reporting, Verification (MRV) standards and enforcement. A specific feature of Chinese MRV system is that the liability falls on the compliance unit and not the installation level as in the EU ETS. Local DRCs are entrusted to formulate MRV guidelines for their ETSs which may differ from one another. At the time of writing, Shanghai and Shenzhen have released guidelines on emissions quantification and reporting based on ISO 14064-1: 2006 terms, and Shenzhen also published the verification guidelines. In parallel, NDRC is currently developing a national electronic reporting and verification system. In a move to strengthen both emissions database and reporting means, in several pilots, non-compliant firms are also required to report their emissions. The inclusion thresholds for mandatory reporting only are lower than those for direct liability under the cap. Enforcement measures and fines in case of fraud or non-compliance are summarized in Annex 8 and vary across pilots. Roughly speaking non-compliant firms must pay a fine of about three times the average market price for each missing allowance and will see their following year allocation be reduced by one time the missing amount, or more.

As of mid-July 2014, five of the seven pilots have completed the first year compliance. All entities in Shanghai have fulfilled their obligations while two firms in Guangdong have not surrendered sufficient allowances. The number of non-compliance emitters is four each in Tianjin and Shenzhen while compliance information has not yet been disclosed in Beijing.

The special case of the power sector. In 2011, power and heat generation roughly accounted for half of China's CO_2 emissions. China's electricity production is heavily coal-fired, accounting for 66% of its total installed generation capacity and 77% of its total electricity production in 2011. Since electricity price is fixed and power dispatch regulated by the government in China, encompassing the electricity sector in the ETS may well be the greatest challenge of all but it is also essential in nature and feasible. Although the optimal scenario would be to completely repeal the current regulation so that decisions could be made as a response to the price signal, this option deviates from the authority's main focus on electricity price stability and supply security. Nevertheless, carbon pricing can be introduced ahead of full open-up of both price and dispatch regulations and still be effective. As the pilots have opted for, the best option in such a context is the upstream coverage of all power plants while other sectors may be covered downstream. Given that electricity prices are fixed and no carbon cost pass-through is feasible double counting is likely to occur for electricity-related emissions since both power generator and large electricity consumers are liable.

2.2. Regulatory conditions of domestic offset markets

Different from the laisser-faire approach taken for local governments testing carbon trading programs, the central government has unified at the national level the procedures for generating project-based offsets. In June 2012, the NDRC released the interim VER Rules (Measures for management of Voluntary Emission Reductions Transactions in China) establishing the regulatory guidelines for domestic project-based offset market. In doing so, the NDRC intends to centralize the management of offset credits eligible for compliance use in the 7 ETS pilots so that it might ease and boost ETS development. Credits respecting these rules are labeled CCERs, standing for China Certified Emission Reductions. The entire approval process closely resembles that of the U.N. CDM mechanism, with the NDRC being the counterpart of the CDM Executive Board in that it oversees the development of methodologies, projects registrations and hosts the a national registry. China-based producers of CER offsets are given a possibility to convert their U.N. approved yet unsold credits into CCERs via a re-registration process. In contrast to the allowances issued during the pilot phase, CCERs could be transacted across borders and are likely to have a higher long-term value because they could be transition into a national program.

All projects seeking CCER accreditation must use methodologies approved by the NDRC, which can be those adapted from existing CDM methodologies or newly submitted. As of August 2014, three forest methodologies and one grassland management methodology have been approved by the NDRC. Four kinds of projects are eligible to request for registration with NDRC: (*i*) new projects using methodologies registered with the NDRC; (*ii*) CDM projects already approved by the NDRC (acting as the designated national authority (DNA) in the CDM process) but not yet registered with the CDM EB; (*iii*) CDM projects which had previously generated emission reductions prior to registration with the EB, e.g., pre-CDM credits (should be without CER issuance); (*iv*) registered CDM projects yet not issued CERs. Similar to CDM process, project validation prior to registration and verification before requesting for CCER issuance is mandatory and should be performed by NDRC-accredited third party (ies). For the moment, CCER transactions are restricted to the seven trading platforms (which also host allowances trading in the seven pilots) recognized by the NDRC. More details of the VER rules and development of CCER projects can be found in Annex 9.

Regarding the participation of agriculture and forestry in the VER market, 4 out of the 5 new non-CDM methodologies target emissions reductions from forestry (forestation, bamboo forestation and improved forest management) and land use (sustainable grassland management). As of August 2014, among the 285 projects having entered or completed the “validation” process, there are about 20 rural household biodigesters projects which all fall under the project type (*iii*) claiming pre-CDM credits and 2 forestry projects- one in Guangdong intending to request CCERs through carbon-sequestration by afforestation.

2.3. Carbon tax discussions in China

Implementing a carbon tax was obviously another option on table at the early stage of the debate on the most suitable economic instrument to cope with climate challenges in China. The government's concern is that an additional tax generally encounters public opposition because firms are often adamant that they would be better off if they can manage their own liabilities in markets. In the meantime, because using a quantity control instrument ensures a certain environmental target reached, in practice, ETSs are generally preferred to taxes. A traditional argument in favor of levying a carbon tax was that China did not have the market fundamentals to underpin a carbon market. Therefore, whereas an effective ETS was quite a challenge to implement, putting a carbon levy on emissions presented itself as a rather straightforward fallback option for it is a policy relatively easy to handle which the government was already familiar with. That is why in parallel to the inception of the ETS programs, the government continues to give some thoughts as to whether to introduce a carbon tax in China. The MOF, MEP and NDRC have initiated relevant research studies as early as 2009 and released reports that gave different suggestions in terms of tax rate, introduction period, taxpayers, use of tax revenues and other aspects (see Annex 10), but agreed on the limited impacts that a carbon tax would generate on GDP growth and positive incentives signaled to emissions reductions actions. In May 2013, the carbon tax was listed as one of the environmental taxes proposed in the "PRC Environmental Protection Law (draft version)" submitted by the MEP. In short, there are still strong disagreements among relevant ministries on the best carbon pricing mechanism in China with NDRC backing ETS while the MOF and the MEP supporting tax.

It is conventionally not be desirable to include small and diffuse emissions sources into an ETS for it would significantly raise the transaction costs, in particular when the level of reliability of MRV cannot meet ETS standards. Carbon equivalent taxes can instead be used to capture these mitigation opportunities. Furthermore, a carbon tax can be very appealing when there are other tax schemes already in place that can be used to channel the new levy. This is especially relevant in the case of transport to impose upstream liability on fossil fuel distributors. At the international level, there is a consensus building up around the idea of joint and complementary use of carbon trading and tax (hybrid policy scenario). While the market would deal with big emissions sources above a certain threshold, the levy would cover small and harder-to-reach emitters, such as buildings, SMEs, transport, etc.

In theory economists suggest the carbon tax base to be as broad as possible to stimulate wider emission mitigation practices and minimize the risks of unintentional distortion. However, in practice, existing carbon tax systems all exclude non-CO_2 emissions from agriculture given the technical complexity in the accurate MRV of agricultural emissions and subsequent high implementation costs as well as considerable lobbing power from farmers' associations (Elbeze & de Perthuis, 2011).

2.4. Subsidies for synthetic fertilizers is a negative carbon tax

In developing countries, it is common for governments to use subsidies to maintain agricultural materials affordable to farmers. Being a key input affecting crop production, fertilizers often receive particular attention. In China, it is estimated that about ¥ 55.255 billion (US $8.1 million) of fertilizer subsidies were granted by the government in 2010/2011, which represent about 10-30% of the total fertilizer cost (IFA, 2012). Before 2006, subsidies had been exclusively allocated to fertilizer producers, largely through preferential electricity use and fertilizer transportation prices, partial exemption from value added tax (VAT) and the fertilizer reserve subsidy. For example, the fertilizer industry has been exempted from the electricity price hikes for all industries and commercial entities in 2006 and enjoys a lower railway transportation charge rate. The VAT exemption was extended to the entire synthetic fertilizer products in 2006. The NDRC has also initiated a fertilizer reserve subsidy program to stabilize fertilizer supply and reduce shortage at peak seasons. Starting from 2006, the central government begun to shift some subsidies for fertilizer producers to farmers embedded in the 'general agricultural input subsidy' package. Direct payments to farmers were initiated for the purpose of offsetting the negative effects of increases in input prices on grain production.

The core objective of fertilizer subsidies in China is to keep fertilizer price affordable to farmers to ensure national food security and support domestic fertilizer production. This was justifiable in the past given the scarce availability of high-quality farmland and the low land productivity in China. Indeed, the increasing use of inorganic fertilizers (particularly N fertilizers) has substantially boosted both the grain and other crop yields in the second half of the 20th century, but it has also driven China to become the largest consumer of synthetic fertilizers in the world, accounting for about 1/3 of the global consumption. However, there has been minor improvement in grain yields compared with higher rate of N fertilizer application in the past decade, leading to diminishing fertilizer use efficiency and contributing to serious environmental problems, including water quality deterioration in lakes and rivers through N run-off and leaching, soil acidification and GHG emissions. As suggested by Hoffmann (2011), subsidies of agricultural inputs and many such kind payments and usually generate perverse incentives, leading to overuse of chemical fertilizers, pesticides and fuel and encouraging land degradation. If not adjusted appropriately, these subsidy policies will discourage or even be impediments to sustainable development in agriculture.

The purpose of a carbon tax is to encourage less carbon intensive modes of production or consumption via internalizing the negative externalities of excessive GHG emissions. In China, our research results on the historical trends of GHGI of crop production show that overapplication of N fertilizers to the Chinese croplands did not lead to a continuous increase in yields; on the contrary, resulted in a range of environmental issues and heavy budget burden for both the government and rural householders. Therefore, in the sense of encouraging N overuse and contributing to excessive N_2O emissions, the fertilizer subsidies could be regarded as a negative carbon tax. The following equation (Eqn (5-1)) was formulated to estimate the rate of this negative carbon tax in China for the year of 2010. Given their dominant marketplace in China, urea, monoammonium phosphate (MAP), potassium chloride (KCl) were used to represent the general nutrient contents of N, P, K fertilizers, respectively. Explanation, unit and data value for each variable in Eqn (5-1) are summarized in Table 5-2.

$$\begin{aligned} Rate_{CO2e} &= \frac{Rate_N}{EF_{CO2e\text{-}N}} \\ Rate_N &= \frac{Subsidy}{Pdt_{fert}} \cdot Content_N \\ Pdt_{fert} &= \frac{Pdt_N}{Content_N} + \frac{Pdt_P}{Content_P} + \frac{Pdt_K}{Content_K} \end{aligned} \qquad (5\text{-}1)$$

Table 5-2 Explanation, unit and values of variables for estimating the negative carbon tax rate

Variable	Explanation	Unit	Value
$Rate_{CO2e}$	Negative carbon tax rate	¥/tCO_2e	
$Rate_N$	Subsidy rate per unit N	¥/tN	
$EF_{CO2e\text{-}N}$	Emission factor of N input to CO_2e	tCO_2e/tN	5.34 (Table 2-2)
Subsidy	Total fertilizer subsidy in 2010	¥ billion	55.255
Pdt_{fert}	Total production of fertilizers (physical quantity)	Mt	66.19
Pdt_N	Production of N fertilizers (pure nutrient as N)	Mt	45.21
Pdt_P	Production of P fertilizers (pure nutrient as P_2O_5)	Mt	17.01
Pdt_K	Production of K fertilizers (pure nutrient as K_2O)	Mt	3.97
$Content_N$	N nutrient content (N)	%	46%
$Content_P$	N nutrient content (P_2O_5)	%	64%
$Content_K$	N nutrient content (K_2O)	%	55%

The data on fertilizer production was extracted from the Fertilizer Industry 12th Five-Year Development Plan (MITT, 2012). Taken as a whole, producers and farmers received about ¥192 subsidy for manufacturing and applying one ton of N fertilizers, which translates into a negative carbon tax of 36 ¥/tCO_2e (about 4.2 €/tCO_2e) in 2010.

3. Difficulties and specificities of pricing carbon in China's agriculture

Our MACC results show that there are sizeable mitigation opportunities offered by agriculture, a significant number of which can be realized at negative or low costs. Consequently, in absence of transaction costs, the substitution of high cost abatements measures in the sectors currently covered or to be included in an ETS by low-cost mitigation options in agriculture would improve the cost-effectiveness of the whole system. This substitution could be reached by either including agriculture in the ETS or by developing offset schemes. However, the high transaction costs arising from the diffuse nature of agricultural emissions and mitigation actions, the difficulties in accurate emission and abatement measurement and verification as well as the concerns about food production are key impediments to agriculture's inclusion in an ETS or the viability of agriculture offset projects. These hurdles correspond to the hidden cost not captured in MACC studies or the barriers identified in section 4.3 in chapter 4, explaining the existence of unrealized abatement at negative costs in the MACC graphic. The following sections will give an analysis of these obstacles and possible solutions to overcome these obstacles to release agriculture potential via the effective use of economic instruments.

3.1. Diffuse nature of agriculture emissions and mitigation actions: the need for aggregator

GHG emissions sources are highly diffuse determined by the fragmented feature of agriculture activities. For example, even in NZ where farm scales are relatively large, there are still about 41,000 farms around the country, not to mention the predominance of extremely small-scale farms in China with each on average possessing less than one hectare of land. What is worse, the limited land is generally segmented into 2 or 3 plots, increasing the difficulty of management. The government therefore needs a substantial budget for programs and initiatives to pass the right information on fertilizer application and practices to the millions of households in rural areas. This constitutes a large part of the hidden costs which were not targeted in the MACC analysis, giving rise to the appearance of negative cost measures.

A carbon pricing scheme can be introduced here to reduce the overuse of fertilizers and encourage better management practices under the condition that the price signal can be effectively passed to farmers with a moderate economic burden for government. Many economists suggest the use of carbon taxation to regulate diffuse emissions such as in the transport and housing sectors. This principle would also apply to the agricultural sector with a taxation of the content of CO_2 incorporated in the inputs used by farmers.

When it comes to carbon trading, the diffuse nature makes agriculture an inappropriate candidate for ETS coverage. Cap & trade schemes often start with sectors accounting for a large share of emissions and emissions concentrated in relatively limited entities to pursue economic efficiency and relieve administrative burdens. The reporting of emissions and allocation of quotas to each individual farm would imply considerable administrative burdens in terms of cost and time for both farmers and the system authority. To resolve this challenge, an aggregator or representative organization should be designated to represent individual farmers. This is the case in the NZ ETS where farmers are not required to directly participate in the scheme, e.g. reporting and monitoring emissions and surrendering permits. Rather, with some exemptions, meat processors, dairy processors, fertilizer manufacturers and importers, and live animal exporters are engaged as agriculture participants in the ETS. The fragmentation of agriculture systems also explains why the average scale of agricultural project is relatively small in terms of expected credits relative to projects in other sectors (Larson et al., 2011).

A suitable aggregator is also pivotal to render agriculture offset projects economically viable. It is impossible for an individual farmer to initiate an offset project since the costs of setting up, validating, monitoring and verifying agricultural projects tend to overwhelm the value of issued credits. Project aggregation therefore provides a solution to project developers working with hundreds of millions of smallholder farmers on fragmented croplands, especially in developing countries. An excellent example is the French alfalfa drying JI initiative which promotes alfalfa drying by natural sunshine prior to dehydration process to save energy use and partially replaces fossil fuel by biomass energy to power the alfalfa drying oven. As the project developer and aggregator, the Luzéal cooperative, a French pre-wilting cooperative, was highly active and engaged in project organization and coordination. The EDF agricultural projects would also not be successful without the support from relevant local government agencies (e.g. the provincial agricultural extension center in the biogas project) that played the role of project aggregator.

To further address this challenge, the Program of Activities (PoA) under the CDM provides a promising framework since component project activities (CPAs) with the same goal are allowed to be added in an approved PoA without undergoing the entire application process as a stand-alone CDM project. Given the streamlined registration and verification processes, the transaction costs could be substantially saved. In addition, full scalability is possible since additional CPAs can be added at any point in the life of the PoA without requiring additional approval from the CDM EB. This is particularly attractive for agricultural projects given the diffuse nature of the emission sources and mitigation actions. However, only 21 agricultural PoA have successfully registered under the CDM with only one using biomass being issued credits. The main obstacle is the unsolved DOE liability issue on the CPA level, i.e. the transfer of responsibility to DOE for CPA inclusion and corresponding consequences makes DOEs reluctant to validate POAs and include new CPAs (South Pole, 2010).

3.2. Difficulties in accurate measurement and verification of emission and abatement

For a sector to effectively participate in an ETS or to render offset credits credible, its emissions need to be reported and monitored accurately in an economically efficient way and mitigation actions must be verifiable. However, it is challenging to meet these prerequisites since on-farm agricultural emissions are generated through biological processes and may vary a lot depending on climate, soil and water conditions as well as management practices. For instance, section 3.1 of chapter 2 illustrated that estimates of N_2O emissions from various agricultural systems are affected by N application rate, soil and water conditions and other factors. This challenge is also seen in the process of calculating representative abatement potential from mitigation measures (section 1.2 of chapter 3) that mobilized a wide range of experimental data.

Agricultural emissions can be quantified using a bottom-up or a top-down approach and the two kinds of methods present both advantages and limits. On the one hand, albeit transparent, straight-forward, practical and verifiable, applying a top-down approach to compute agriculture emissions by prescribing an homogenous emission factor to diverse agriculture activities can not reflect heterogeneities at the farm level, and therefore is not able to provide incentives to farmers adopting better practices. On the other hand, if bottom-up approaches are used, i.e. farmers need to choose the most appropriate emission factors or conduct soil measurement for more accurate reporting, the administrative burden will be too big for farmers. In the NZ ETS case, the authority has decided to move from a bottom-up to a top-down approach for calculating agricultural emissions in the latest regulation amendment. As such, the allocation will be provided on an output intensity basis, implying that a participant's allocation will vary only with output levels. In addition, the NZ government acknowledged the difficulty in recording attributable emission reduction and consequently removed the entry date for surrender obligations for agricultural biological emissions from the new regulation.

Regarding the offset market, agriculture projects are primarily implemented in the areas of managing methane emissions from manure or using agricultural residual matter as fuel sources, where estimation of emission reductions is easy to conduct with relatively low uncertainty. For instance, the emissions avoided from livestock manure management correspond to the quantity of gas captured by the methaniser. In contrast, it is more complicated and costly to measure N_2O emissions as a result of fertilization. The EDF precise fertilization project used the IPCC default emission factors combined with the N fertilizer use difference between baseline and project scenarios. However, simplification of the calculation method may raise concerns about the accuracy of mitigation effects and negatively affect buyer's confidence. Moreover, a conservative approach is generally required since agriculture offset activities of smallholder projects are subject to a range of influencing factors and uncertainties. Consequently, only 61% of credits have been issued compared to the expected emission reductions described in the Project Development Documents (Foucherot & Bellassen, 2011), further worsening the financial balance of agriculture projects.

If robust MRV presents high transaction costs for agriculture's participation in carbon markets, the best option to introduce a price signal is maybe to pursue carbon taxation, the effectiveness of which does not rely on accurate MRV. Or, MRV processes need to be performed in a more efficient way to reduce the transaction costs, which means using an appropriate indicator for either determining the allowances in a cap & trade system or setting baseline and quantifying offset credits. To this end, using the GHGI performance (section 3.1 of chapter 3) as the benchmark for quota-setting combined with an appropriate aggregator will allow the authority to control the overall agriculture emissions without directly working with each farm. Adoption of standardized approaches for the determination of baselines and additionality based on the GHGI could to a great extent simplify the MRV procedure to make agricultural offset project viable.

3.3. The specialty of soil carbon storage

In agriculture an important proportion of mitigation potential is estimated to stem from cropland soils via carbon sequestration. However, carbon sequestration has not been recognized for the purpose of calculating national emissions inventories or are underrepresented in the offset markets (except CCX), attributable to the difficulty of soil carbon calculation and the reversal risk of carbon storage.

There are existing approaches for soil carbon testing but are not able to simultaneously meet the standards of accuracy and cost-effectiveness, since soil carbon contents are locally specific and it would be very costly to carry out sampling measurement to conclude statistically credible accounting of SOC change. An integrated approach to landscape-level SOC accounting and verification is not currently available to reasonably reflect the climate benefits of various improved soil management practices. If direct measurement is used to monitor carbon stock change, the main difficulty lies in designing an efficient sampling regime to represent soil conditions at the field scale given the high spatial variability of SOC (FAO, 2011). This will lead to an enormous increase in sampling size and implementation cost to reach certain accuracy. Another option is to apply activity-based or biogeochemical process models, but such an approach is demanding in robust research underpinning the establishment of a credible basis. The CCX opted for certifying the practice with predefined sequestration rate rather than measuring or modeling actual carbon stock improvement. It therefore has lower requirements for additionality.

A special issue of sequestered carbon lies in the inherent risk of non-permanence. Prior sequestration, which may already have been verified, credited and sold as offset credits, may be released back to the atmosphere through unintentional occurrences (e.g. fire, flood, etc.) or intentional factors (e.g. project proponents choosing to discontinue the project activity). Addressing this issue revolves around distribution of the reversal risk and responsibility for the cost of mitigating the risk to make carbon sequestration projects attractive. Under the CDM, agricultural soil carbon is not recognized while temporary crediting is the approach retained for A/R projects. However, in the voluntary markets most standards, including the Australian CFI, adopt the ‘buffer tool’ approach, where each sequestration project is mandated to deposit a certain number of credits to the common buffer pool managed by the Standard operator before credits are issued. Engaging third-party insurance for managing the reversal risk is also under consideration.

4. Suggestions on pricing carbon in China's agriculture

From the MACC analysis, we estimated that both the technical and economic mitigation potentials are significant in Chinese agriculture. However, such potential is currently far from tapped, particularly the negative and low-cost mitigation potential technologies. The extreme overuse of synthetic fertilizers is a good demonstration that farmers don't take into account environmental benefits (including climate change) in production practices despite mitigation actions also improving productivity. To alter the behavior of farmers towards more sustainable and low-carbon production modes, introducing a carbon price into agriculture seems to be a good option since Chinese farmers are probably more responsive to a carbon price signal than the state power companies which are highly regulated. Farmers are more liberalized to adjust their farming practices taking into account the changes in cropping-related factors and therefore more incline to incorporate mitigation aspects into their decisions. In the exploration of appropriate market-based instruments, how to address the specificities of the agricultural sector (as discussed in section 3) should be key considerations.

4.1. Restructure fertilizer subsidies

Considering the diffuse nature of agriculture emissions and the challenges of meeting a robust MRV, imposing a carbon tax may be more preferable to a cap & trade system in pricing carbon in the Chinese agriculture. However, an emission tax, either applied to the production of agricultural products on-farm or consumption-based, would incur a considerable administration burden and evoke public aversion, making it hard to implement. On the contrary, the government's various subsidies on synthetic N fertilizers play an important role in stimulating fertilizer excessive use and related GHG emissions, acting as a negative carbon tax at an equivalent rate of 36 ¥/tCO_2e (section 2.4).

It is obvious that there is an urgent need for China to restructure the subsidy policies on N fertilizer production and use to incentivize more environmentally-friendly and yield-beneficial practices. The principle here is not to remove government subsidies to support rural development and food production, but rather to reform the subsidy structure to pursue low-carbon agriculture development. For reference, our MACC analysis indicates a total saving of at least 10 million tons of N fertilizers in the case of full implementation of mitigation measures C1, C2 and C4 on fertilizer use. Zhang et al. (2013) points to a possible reduction of 17.2 million tons in total agriculture N fertilizer use in China in 2010.

It is obvious that the government's heavy subsidies consolidate farmers' reliance on "high input, high yield", therefore, lowering the subsidies on synthetic fertilizers is itself a positive political signal to change the fertilization behavior of farmers who already have interest to do so for all measures at negative costs in the MACC. In addition to making farmers more rational on their fertilization practices, as much as ¥ 1.92 billion government expenditures could be saved from N fertilizer subsidies if 10 million tons of N fertilizers overuse are to be avoided. This amount is substantial and can be redistributed to finance agricultural development in a more efficient way; in other words, to help remove the institutional and social barriers (as discussed in section 4.3 of chapter 4) or to compensate the hidden cost preventing the realization of mitigation potential at negative or moderate costs in the MACC.

One possible use of the economies of grants is to improve the infrastructures and technical extension services for agriculture development. Secondly, given the increasing shortage of 'full-time' farmers in rural area, the development of professional service groups appeared prominent as suggested by Zhang et al. (2013). Such groups can provide contracted services including fertilizer application, machine use and irrigation to farmers working in cities and towns so that they do not need to come back to countryside in critical cropping periods. Finance programs could be intended to foster the creation and promote the development of such professional groups to make contracting service economically beneficial than the opportunity cost of farmers' quitting works in cities. In addition, it is more efficient to educate and inform such groups on best fertilizers application practices and they have greater motivation and access to purchase machinery for subsurface application and adjust the fertilizer application time. Thirdly, part of subsidies on synthetic fertilizers could be replaced by those aiming to improve the utilization of large quantity of livestock manures which is so far much less subsidized, or to cover the additional costs of enhanced efficiency fertilizers. Finally, parts of the government expenses could be redirected to crediting mechanisms in agriculture to reduce the various transactions costs to make agriculture offsets viable or to help prepare the inclusion of agriculture in a pilot ETS that we also going to discuss.

4.2. Up-scaling carbon crediting schemes

Although agriculture is not excluded from Chinese offset regimes, past experiences (section 1.3) indicated the difficulty for stand-alone or individual projects developed following the traditional procedure to deliver significant impacts. To minimize the high transaction costs (as discussed in section 3), the most plausible and potential option is to explore the possibility of scaled-up or programmatic approaches using standardized baselines for carbon crediting in agriculture. A new channel of finance through carbon compensation could help popularize cost-beneficial measures in the MACC graphic and make those with moderate costs economically viable to farmers.

By referring to the CDM PoA or engaging enough number of householders under a bundled project, simplified processes shall accelerate agriculture's access to carbon offset markets. The PoA is particularly relevant to aggregating emission reductions from smallholder farmers in

China since further activities don't have to demonstrate additionality or be individually validated. In terms of project coordination, we recommend local agriculture or environmental administrative agencies as the aggregator for PoA or up-scaled projects. The reasons are pretty the same as suggesting them as the entry point to an ETS (will be discussed below in section 4.3) for their unique advantages in both project organization and implementation. As a matter of fact, implications of local agricultural administration bureaus are indispensable whoever the project aggregator (such as NGOs or carbon professionals) is. For political reasons, a carbon project could not be successfully carried out by other project developers without assistance from local government. It would, therefore, be convenient to have them act directly as the project aggregator to incentivize project implementation with other participating entities providing expertise on carbon finance. As one example, in the agriculture offset projects initiated by the US EDF, Shannxi Provincial Agricultural Technology Extension Administration Center and Sichuan Provincial Environmental Protection Office served as project aggregators and played a key role in projects' success.

In the sphere of technical procedures, the Chinese authorities need to simplify the application process for agricultural projects. A core element in making agriculture projects attractive (see section 3.2) is to adopt streamlined process to determine baselines. In this regard, this PhD research outcome on the national and regional GHGI of crop production contributes to the exploration of large-scale benchmarks baselines. Assuming that all eligible farmers in a country/city are grouped in a carbon offset project, its abatement performance could be evaluated against the historical (generally the past 2-3 years) GHGI of this place or the current province-average GHGI. Using the merit of GHGI as the performance evaluation index shall stimulate mitigation and yield improvement simultaneously. In addition, as demonstrated in our research, it is not complicated to gather the information needed for GHGI calculation. Furthermore, our MACC results could also assist the identification/selection of mitigation measures that are most locally applicable, both in terms of abatement potential and economic benefits. When it comes to the management of carbon reversal risks, the prevailing approach among the third-party standards to depose a part of carbon sequestration credits into a common reserve could be well adopted.

Apart from using standardized baselines, an offset program administrator is also suggested to give large flexibility in or have least requirements for demonstrating additionality in certain cases given the "non-regretfulness" feature of agriculture mitigation activities. Another key issue is to shift the liability from DOEs for the inclusion of further activities under the current CDM rules to project aggregators to accelerate projects validation/verification. Indeed, project aggregators have more leeway to ensure the performance of abatement actions.

To achieve payments to smallholder farmers on a larger scale, crediting schemes need to be embedded in government-led programmes and be established on a regional or sectoral basis using intensity as the merit for performance evaluation. The REDD experience combined with our attempts on GHG intensity estimates has provided some useful guidance on how to develop a robust and simplified baseline. Setting crediting schemes on a regional basis can dramatically reduce carbon-related transaction costs and provide flexibility to scale up mitigation activities. To a larger extent, the nationally appropriate mitigation actions (NAMAs) can serve as another field for channeling carbon finance to agriculture mitigation through sectoral crediting. In fact, PoA building blocks can be highly coherent with NAMA target trajectory especially the baselines in the sense that a NAMA crediting program for agriculture can be regarded a regional or national sectoral PoA project. A NAMA framework can further simplify the whole process since benchmark baselines can be introduced on a provincial scale to facilitate project coordination and environmental integrity, especially the DOE liability for CPA wrong inclusion which can be controlled by a national authority. In addition, as mitigation and adaptation are generally inseparable, NAMA crediting proposals could be aligned with national adaptation plan to maximize synergies and co-benefits.

On the demand side, agriculture credits issued from up-scaled or PoA projects could have three main destinations. First, they could be defined eligible for compliance use in the current pilot ETSs or in the future national scheme. Secondly, they are also attractive to companies willing to buy credits for voluntary carbon compensation owing to the high co-benefits (e.g. adaptation, poverty alleviation and sustainable development) of agriculture mitigation projects. Thirdly, large state-owned companies under the national energy-efficiency programs could buy agriculture offsets instead of paying the penalties for missing their emission reduction targets. Such mechanisms create a channel for finance flow from cities to the countryside, and from industry to agriculture, and are therefore well aligned with the ongoing discussion on eco-compensation in China.

4.3. ETS pilot covering the agricultural sector

In line with the specificities of agriculture emissions and challenges in MRV, it is probably more complicated to include agriculture into a cap & trade scheme than initiating offset programmes in this sector. This is the case in the current carbon markets where bottom-up initiatives on agriculture offsets have been proliferating across the world despite of the limited scale while agriculture is nearly absent in cap & trade systems. Still, in the meantime of up-scaling crediting schemes in agriculture, it might be worth contemplating the benefits of extending the coverage of a pilot trading scheme to agriculture and how could this be practically possible.

Two advantages merit particular considerations in mandating agriculture in a GHG cap & trade system. First, a cap & trade system would be more effective to send the price signal than crediting schemes as the former generally attract ampler attention and engage larger scale of participants. Secondly, sending a carbon price signal to agriculture through the extension of an ETS scope will accelerate the financial flows from industry and energy sectors to the countryside if actors opt to pursue the abatement at the least marginal cost. This is justifiable since the full realization of the cost-beneficial abatement identified in our MACC points to a total savings of as much as ¥125 billion per year for farmers in 2020. In a period when the government is keen to identify effective approaches to achieve the objective of "cities support countryside and industries compensate agriculture", an ETS presents a preferable tool due to its efficiency in directing financial resources towards the cheapest marginal abatement in the system.

In spite of these advantages, as discussed above high transaction costs remain the greatest challenge to incorporating agriculture into any ETSs. Ancev (2011) estimated that unit transaction cost in agriculture is likely to be more than 2.5 times higher than in the sectors currently covered in the EU ETS, principally due to the small scale of agricultural entities to be included in the scheme. That is why none of the ETS pilots in China directly regulate agricultural emissions under the cap & trade systems as elsewhere in the world.

However, if there are ways to make cost-savings from agriculture abatement outpace the costs, a pilot cap & trade scheme covering N_2O emissions from synthetic fertilizer use may be worth experimenting. It is preferable to start such a pilot in one of the current seven pilot regions where ETS infrastructure has been gradually put in place to avoid the fixed costs related to the creation and establishment of the market. We would suggest such a pilot to be started in Guangdong as this province is more open-mined to accept new concepts. We propose mandating the local agriculture bureau/agencies on the county-level as the participating entities in the scheme, responsible for receiving and surrendering allowances and communicating with the ETS management authority. There are several advantages in designating county-level agricultural bureaus as the aggregator to consolidate the small-scale farms in their administrations. First, local agriculture administrations are the hub of information related to agriculture production including cropping, fertilizer use, yields, land management and others; so they are the best candidate to interact with the ETS authority to alleviate the administrative burden. Secondly, familiar with local agricultural conditions and having frequent contact with farmers, local agricultural bureaus are better placed to advocate mitigation measures to farmers. The ultimate objective is to lower the overall carbon intensity of the county and therefore they are not obliged to allocate allowances to each individual rural household. They could achieve the objective by enforcing the implementation of programmes for low-carbon development in agriculture. Or, they can (and know how to) pick up representative farmers to apply some cost-beneficial measures at the initial stage to show the positive effects to have them accepted by more farmers. Thirdly, with the expectation of receiving rewards from the carbon market, these bureaus may be more likely to promote sustainable management practices among farmers.

Regarding the cap setting and allowance allocation, the GHGI is suggested as the benchmark. The information for calculating the GHGI is relatively easy to collect at the county level. Each county could be required to report their historical GHGI of crop production following the methodologies used in this study. To set the cap an objective on GHGI needs to be defined, which can be in the form of proportional increase of the historical GHGI or the proportion of optimal intensity reflecting the perception on efficiency improvement. The optimum level of GHGI can rely on our research results we concluded in the purpose of quantifying provincial mitigation potential from N-use measures. Another component in determining the cap using an intensity benchmark is the quantity of crop production. The cap can be set ex-post when the production level is known relative to industrial entities, as farmers are less capable of predicting crop productions facing numerous and varied climate risks. This means the allowances that the aggregator will receive are determined based on the actual level of production rather than be allocated ex-ante. Such a method avoids the difficulty in projecting accurate production levels. Transaction activities can take place in the next-cycle of ETS operation: counties having better performance in minimizing the carbon footprint of agricultural activities could sell their quotas to the other regions or to energy/industrial entities within the same scheme, facilitating financial transfers from industrial sectors to agriculture.

5. Conclusion

As suggested by Kahrl et al. (2010), achieving sustainable intensification and addressing fertilizer challenge in China's agriculture requires a rethinking and reorienting of public service support to this sector and an exploration of funding schemes to support these services.

Lack of clear and firm political signal is one of the reasons giving rise to the existence of significant negative-cost mitigation potentials in the MACC results. Therefore, as a first step, we suggest integrating climate change into agriculture policy-making to deliver a clear message on the government's move towards sustainable low-carbon agriculture. We recommend enhanced emphasis in national policies to highlight the role of the agricultural sector in GHG reduction targets and the long-term decarbonization of society.

In the exploration of funding mechanisms, there is no single approach to tackling climate mitigation in agriculture and a range of economic mechanisms are needed in addition to regulatory regimes. Market incentives can be in the form of grants, subsidies, levies, carbon taxes, cap & trade scheme or more broadly, payments for environmental services. In pondering on the suitable mix of financing mechanisms the key considerations are scaling up of funding and delivery mechanisms, reducing transaction costs and the effectiveness of reaching small-holder farmers. The government's various subsidies on synthetic N fertilizers could be reasonably considered a negative carbon tax (at an equivalent rate of 36 ¥/tCO_2e according to this study) because of their encouraging effects on fertilizer overuse and GHG emissions. We therefore recommend the reform of the current subsidy structure to save government expenditure for improving rural infrastructure and extension service, catalyzing professional service groups, promoting the use of organic manure and lowering the transaction costs related to carbon crediting /emission trading schemes in agriculture.

When it comes to the domestic carbon markets, all the Chinese ETS trials currently exclude agriculture from their mandatory regimes and it is unlikely for a national carbon market, if implemented after 2016, to regulate agriculture emissions directly. However, including agriculture into one of the ETS pilots with GHGI performance as allowance allocation criteria is worth trying as agriculture offers significant cost-effective mitigation potential compared to energy and industry sectors. In addition, assigning an emission cap to aggregators such as local agriculture administration bureaus would enforce and encourage them to carry out mitigation actions with potential rewards from participating in the ETS system. Although it is ideal to mandate agriculture into a pilot ETS to stimulate mitigation actions to the largest extent, such an inclusion requires a lot of efforts in local capacity building to get administration bureaus acquainted with ETS procedures and trained on how to manage allocated allowances as well as to perform trading activities in the market. Given the heavy workload and the government's cautious attitudes towards involving agriculture in mitigation caps, it is more practical to start with the offset approach to get local ETS authority and agricultural administrations prepared. In line with the objective of minimizing transaction costs while preserving the environmental integrity and ensuring food security, we suggest using local GHGI as the benchmark baselines to upscale agriculture mitigation efforts. Scaled-up approaches are relevant for bundled projects, PoA programs and sectoral crediting, the latter of which could also constitute the core element of a NAMA financing and crediting proposal. In any of these crediting schemes, saved government budget from reforming N fertilizer subsidies can be partially directed to compensate the high transaction costs of initiating emission trading or performing carbon projects in agriculture.

Whichever the market-based approach or a combination of approaches undertaken to tap the vast mitigation potentials offered by agriculture, a robust MRV framework needs to be gradually improved to foster confidence in payments for land-based mitigation and carbon sequestration.

General Conclusion

This thesis is structured around the core research question of whether it is worth integrating agriculture into China's climate change mitigation strategy and consequently how a carbon price signal could be effectively introduced in this sector. The key objective is to estimate the mitigation potential available in agriculture and determine the cost-effectiveness of agriculture abatement options. The research results will help policy makers make better use of economic instruments to unveil the vast mitigation potentials offered by Chinese agriculture.

Based on a review of current climate policies in China, chapter 1 shows that agriculture has not been incorporated into the national mitigation strategy. Domestic mitigation efforts have continuously centered on the energy, industry and forestry sector. However, given the size and urgency imposed by the climate challenge, it is crucial that mitigation potentials in all sectors should be tapped fully to achieve a successful transition to a low carbon economy. Agriculture should also contribute to the collective endeavors since it is the primary source of N_2O and CH_4 emissions arising from both land cultivation and livestock breeding practices. In chapter 2, we estimated the baseline emissions under the BAU scenario for 2020 based on emission quantification methodologies and the forecast of agriculture activities. We are able to conclude that, without additional mitigation incentives, agriculture emissions will continue to climb in the near future and higher growth rates will be observed in emissions related to livestock production as compared to croplands. We then try to investigate the opportunities that croplands could provide to constrain climate change while safeguarding national food security, generating two main research results from a technical and economic perspective.

The first result of this thesis is to provide a complete assessment of the overall technical mitigation potential from cropland emissions (chapter 3). In doing so, a comprehensive database from hundreds of field experiments across China is mobilized to inform the abatement rates of the nine identified cropland mitigation measures. The results are therefore representative of Chinese conditions and exhibit the climate benefits of adopting a mitigation measure against the conventional practice. An investigation on the additional area available for measure implementation is also carried out to see the extent to which an abatement measure could be adopted against the baseline. Taking into account measure interactions, the overall technically feasible mitigation potential from Chinese croplands is estimated at 149 $MtCO_2e$, representing 35% of the BAU emissions. Our results highlight the significant potentials of measures like organic manure amendment, synthetic N fertilizer management in uplands and water regime improvements in rice paddies. Particular attention is dedicated to the GHGI variability of crop production among provinces and the implications for differentiated mitigation challenges and opportunities from best N management practices faced by each region.

The second result of this thesis is an assessment of the cost-effectiveness of each mitigation measure for the purpose of constructing a MACC for the Chinese agricultural sector (chapter 4). Given the diversity of agricultural systems and in an attempt to maximize the large extent of data collected, we apply a bottom-up engineering approach in the MACC derivation. Implementation costs are estimated by evaluating the changes in production inputs/outputs for farmers when complying with a mitigation option. We are then able to conclude the cost-effectiveness of each option which represents the national average cost of avoiding one tonne of CO_2e. The MACC results demonstrate that about 1/3 of the technical potentials from agriculture could be achieved at zero or negative costs and about half of the technical potentials are available at a cost below a threshold carbon price of less than ¥260 (approximately €32) per tCO_2e. High cost-effectiveness is particularly observed in synthetic N fertilizer management practices for both grain and cash crops, attributable to the current excessive and improper use of N-containing fertilizers.

MACC results suggest an initial indication of priority interventions in the design of efficient policies. This is particularly relevant for those measures identified as ‘win-win’ in the MACC graphic, which, according to rational economic theory, should have already been adopted voluntarily by farmers since they save costs or increase revenues in addition to limiting emissions. The authorities therefore need to identify and remove any barriers to the realization of win-win potentials. Compared with other sectors of the economy, agriculture could provide significant CO_2e abatement at competitive costs. Incorporating agriculture into the national mitigation strategy is therefore worth pursing since a rational mitigation policy should normally prioritize the cheapest means of abatement by equalizing marginal abatement costs across sectors.

Finally, the thesis analyzes the conditions and possible approaches of introducing a carbon price into Chinese agriculture. Chapter 5 shows that in China, applying economic instruments to control GHG emissions have only been brought into attention in the past few years, but this doesn’t impede the country’s fast progress towards a domestic carbon price. China is now the second largest player in the global carbon markets with the launch of its seven pilot ETSs. Similar to international experiences, agriculture is not included in the sectors covered by the pilot cap & trade systems, and is underrepresented in offset markets. The high transaction costs arising from the diffuse nature of agriculture emissions and the difficulties in consistently measuring and reporting emissions are key obstacles. Facing these challenges, we suggest that the urgent need in China is to reduce the negative carbon tax rate. This means reforming the current fertilizer subsidy systems to create a right price signal to encourage reasonable use of synthetic fertilizers. Despite the relatively high requirement on MRV, carbon market mechanisms may also merit some trials since they provide efficient channels for financial flows from industry and energy sectors to climate-friendly production systems in rural areas. Regarding project-based offsetting, up-scaled crediting programs such as the PoA initiative are recommended, which may prepare the ground for a possible pilot ETS covering agriculture. In both cases, local agricultural administrations are probably the best candidate for aggregators while the use of GHGI is highly proposed as either the benchmark for allowances allocation in a pilot ETS or the standardized baseline in large-scale carbon crediting schemes. If China could come up with innovative methods to integrate agriculture into domestic carbon markets, this example would greatly accelerate the process of putting a carbon price on agriculture at the international level.

The main contribution of this thesis is that it is the first attempt to derive a bottom-up evaluation of technical and economic abatement potentials for the agricultural sector in China. The findings suggest important points of policy intervention using market-based instruments. There are several limitations of the study, which themselves suggest numerous further research areas.

Firstly, this study is principally performed on a national scale so that China, as a whole, is assimilated as one showcase farm. However, both mitigation potentials and cost implications of an abatement measure will be quite variable among regions, depending on local ecological conditions that farmers are operating under, the types of farming systems, the degree to which mitigation and productivity improvements can be obtained as well as the socio-economic environment. The investigation of regional GHGI of crop production indicates significant variations of mitigation potentials and the examples on regional implementation costs also illustrate the differences in measure cost-effectiveness among provinces. A mitigation measure could be cost-effective in one region but may be cost-prohibitive when implemented in another region. These variations could be reflected in building regional MACCs, from which a national MACC would be aggregated. The construction of a regional MACC could replicate the methodological approach applied in this study but using local data. However, due to time constraints and limited data availability, regional variations are not taken into account except for measures involving synthetic N use reduction. In the future, more detailed regional work merits further investigation since such information would be helpful in assisting decision marking at the regional level.

Secondly, similar to other MACCs, this study also reveals important potentials from win-win mitigation measures which simultaneously reduce emissions and save costs. The win-win effects are particularly highlighted in those measures related to improved synthetic N fertilizers management. This is due to the limits in considering other wider social costs and the simplification of assuming farmers being rational in production decision making. Nevertheless, in reality, farmers are usually risk averse to uncertainties induced by natural hazards. As a result, they are reluctant to part with their traditional belief in ‘high input, high yield’ and to take on novel management practices. The government’s heavy subsidies of synthetic fertilizers further consolidate farmers’ reliance on excessive fertilizer use in crop production. The suggested subsidy reform will help to send the right signal on rational fertilizer use. The principal is not to reduce the support to agricultural development; but rather to redistribute the public funds to provide farmers with better information on fertilizer use to overcome the various barriers to the fulfillment of win-win measures. In addition, a part of the public funds could be dedicated to support assurance systems for compensating production losses caused by natural disasters or harvest volatility. In addition, Moran et al., (2013) suggests that more contributions from psychology, cultural evolution and behavioural economics would help in designing more effective policies to send win-win messages.

Finally, land cultivation, livestock activities and related emissions are treated independently in this thesis. There are limited considerations of the interactions between cropland, livestock and grassland mitigation actions. However, in reality, the two kinds of farming practices are highly correlated and the extent of interdependencies will be much affected by the expected shift in nutrition intake toward meat and dairy products. This implies that, for example, maize and soybean areas and productions will continue to expand and larger amounts of livestock manure will be produced. How to effectively design integrated production systems and accelerate the better recycling of organic manure should be a research priority in the future.

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Annex 1: Selected values for estimating N_2O emissions from croplands

Table A Selected values for estimating N inputs to croplands from animal manure

	Non-dairy cattle	Milk cows	Sheep (goats)	Horses	Asses	Mules	Pigs	Chicken	Rabbits
$Frac_{Grazing}$*	17%		35%						
N_{rate}	0.34	0.47	1.27	0.46	0.46	0.46	0.5	0.82	
TAM	319	350	29	238	130	130	50†	2	
Nex	39.6	60	13.4	40	21.8	21.8	9.1	0.5	8.1
$Frac_{Loss}$	40%	40%	67%	50%	50%	50%	35%	50%	50%
Days_alive‡							158	180	105

* Data in this table represents the national average.

† IPCC default value for Asia is 28. Here we adopted 50 according to Chinese conditions.

‡ Days_alive of chicken is the weighted number of broiler chicken (65 days) and hens (352 days), which account for 60% and 40% of chicken population, respectively.

Note: Annual number of head slaughtered was collected for pigs, hens, broiler chicken and rabbits with average breeding days standing at 158, 65, 352 and 105, respectively (MOA, 2001-2011). As for other types of animals, annual stock number was used.

Table B Selected values for estimating N inputs to croplands from crop residues

	R_{ST-GR}	N g/kg	R_{BG-AG}
Rice	0.9	9.1	0.13
Wheat	1.1	6.5	0.17
Maize	1.2	9.2	0.17
Potato	0.5	25	0.05
Soybean	1	21	0.13
Cotton	3	12.4	0.2
Oils	1.7	13.5	0.17
Vegetable	0.5	2.5	0.25

Table C Proportion of aboveground straw residue returned to land in 2010

	North	Northeast	East	Southeast	Southwest	Northwest	National average
Rice	75%	33%	25%	77%	39%	19%	39%
Wheat	100%	48%	37%	100%	58%	28%	57%
Maize	67%	29%	22%	68%	35%	17%	35%

Note: North region includes Beijing, Tianjin, Hebei, Shanxi and Inner Mongolia; Northeast region includes Heilongjiang, Liaoning and Jilin; East region includes Shanghai, Anhui, Fujian, Jiangsu, Jiangxi, Shandong and Zhejiang; South Central region includes Guangdong, Hainan, Henan, Hubei, Hunan and Guangxi; Southwest region includes Chongqing, Guizhou, Sichuan, Yunnan and Tibet; Northwest region includes Gansu, Qinghai, Shaanxi, Ningxia and Xinjiang.

Table D National average proportion of aboveground straw residue returned to land

	2005	2010	2015	2020
Rice	29%	34%	35%	36%
Wheat	42%	49%	51%	52%
Maize	26%	30%	31%	32%
Potato	18%	21%	22%	22%
Soybean	45%	52%	53%	55%
Cotton	12%	14%	15%	15%
Oils	17%	20%	21%	22%
Vegetable	5%	6%	6%	6%

Annex 2: Detailed quantification methods of GHGI of cereal crops

$Flux_{N2O(Direct)}$ is estimated using Eqn (A), Eqn (1-2) and Eqn (1-3).

$$Flux_{N_2O(Direct)} = \frac{Emissions_{N_2O(Direct)}}{CA} = SN_{rate} + \frac{F_{AW}}{CA_{eqv}} + \frac{\sum_i F_{CR(i)}}{\sum_i CA_i} \qquad (A)$$

$$CA_{eqv} = a \bullet CA_{veg} + b \bullet CA_{fruit} + CA_{other}$$

SN_{rate} represents per hectare synthetic N fertilizer application rate (kgN/ha). CA_{eqv} denotes the equivalent cropping area (kha).CA_{veg}, CA_{fruit} and CA_{other} are the cropping areas of vegetables, fruits and other crops (excluding vegetable and fruits), respectively (kha). a and b is the ratio of organic manure received by vegetable fields and fruits compared with other crop lands, respectively. 4 and 5 are assigned to a and b since survey results (Huang & Tang, 2012; Zhang et al., 2013) show that vegetable and fruit fields generally receive 4 and 5 times, respectively, more organic manure than cereal cropping lands in the 2000s. i denotes crop type (rice, wheat, maize). i denotes crop types (rice, wheat or maize).

Since N application rates of the tree major cereals are only available for the year of 2005 and 2010 at 5-year intervals, Eqn (B) is formulated to estimate N application rates in a given year.

$$SN_{rate(i)j} = SN_{rate(i)2005} \bullet \frac{SN_{ratej}}{SN_{rate2005}} = SN_{rate(i)2005} \bullet \frac{TN_j}{TCA_j} \bullet \frac{TCA_{2005}}{TN_{2005}} \qquad (B)$$

$SN_{rate(i)j}$ is the synthetic N application rate of crop i in year j in a province (kgN/ha). i denotes the crop type (rice, wheat, maize) and j denotes year. $SN_{rate(i)2005}$ is the N rate of crop i in 2005(kgN/ha). SN_{ratej} and $SN_{rate2005}$ denote the crop-wide average N rate in year j and 2005, respectively (kgN/ha). TN_j and TN_{2005} are the provincial total synthetic N consumption in year j and 2005(kt). TCA_j and TCA_{2005} represent the total cropping area in year j and 2005(kha).

Annex 3: Past and predicted future agriculture activities

Table E Past and predicted future agriculture activities (crops)

Crops	Cropping area(kha)			Production (kt)			Yield (t/ha)			Price (¥/kg)			
	2010	2020*	Annual change	2010	2020*	Annual change	2010	2020*	Annual change	2010	2020	Original CAPSiM annual change*	Adjusted by inflation (+2%)†
Rice	29,873	25,612	-1.5%	195,761	176,823	-1.0%	6.55	6.90	0.5%	2.36	3.02	0.70%	2.5%
Wheat	24,257	22,099	-0.9%	115,181	113,260	-0.2%	4.75	5.13	0.8%	1.98	2.46	0.40%	2.2%
Maize	32,500	35,361	0.8%	177,245	221,882	2.3%	5.45	6.27	1.4%	1.87	3.13	3.50%	5.3%
Sweet potato	3,545	3,923	1.02%	14,834	18,910	2.46%	4.18	4.82	1.42%				
Potato	5,205	5,118	-0.17%	16,307	17,981	0.98%	3.13	3.51	1.15%				
Other coarse	6,108	5,356	-1.31%	12,037	11,781	-0.21%	1.97	2.20	1.11%				
Soybean	8,516	8,223	-0.3%	15,083	16,549	0.9%	1.77	2.01	1.3%	3.87	5.46	1.70%	3.5%
Cotton	4,849	5,168	0.6%	5,961	7,503	2.3%	1.23	1.45	1.7%	24.77	26.28	-1.10%	0.6%
Oils	13,890	14,613	0.5%	7,106	8,757	2.1%	0.51	0.60	1.6%	5.25	8.5	3.10%	4.9%
Sugar	1,905	1,837	-0.4%	14,199	15,297	0.7%	7.45	8.33	1.1%	0.45	0.68	2.30%	4.1%
Total vegetable	19,000	19,040	0.0%	650,994	785,748	1.9%	34.26	41.27	1.9%	1.56	2.21	1.70%	3.5%
Greenhouse vegetable‡	3,553	3,560	0.0%	162,749	196,437	1.9%	45.81	55.17	1.9%	1.98	2.81	1.70%	3.5%
Openfield vegetable‡	15,447	15,479	0.0%	488,246	589,311	1.9%	31.61	38.07	1.9%	1.42	2.01	1.70%	3.5%
Fruit	11,544	11,668	0.1%	128,652	176,712	3.2%	11.14	15.14	3.1%	3.54	4.72	0.90%	2.9%

* Future cropping area, production, yield and agricultural price change (with variations among years) were direct modeled results of CAPSiM.

† Since inflation is not an element considered in the CAPSiM model, here we adjusted price variation rate by assumed annual inflation at +2% (+2.1% during 2001-2010).

‡ CAPSiM model gives information on total vegetable; here we split into greenhouse and openfield vegetables to facilitate subsequent mitigation potential analysis. We assume that greenhouse vegetable accounts for 18.7% and 25% of total vegetable cropping area and production, respectively, from 2005 to 2020 (Wang et al., 2010).

Table F Past and predicted meat production and livestock numbers

	Production annual growth rates (%) [*]						Livestock population (1000 heads)		
	NBS (2000-2010)	CAPSiM results (2010-2020)	OECD-FAO (2010-2020)	USDA (2010-2020)	FAPRI (2010-2020)			2010	2020
Beef	2.4%	4.8%	1.5%	1.7%	2.5%		Non-dairy cattle	92,063	147,617
Milk	15.8%	5.0%	3.6%		5.0%		Milk cows	14,201	23,095
Mutton	4.2%	3.8%	2.1%			Stock population	Sheep+goats	280,879	407,711
							Horses	6,771	6,771
							Asses	6,397	6,397
							Mules	2,697	2,697
Pork	2.5%	2.5%	2.1%	2.1%	2.8%		Pigs	666,864	853,203
Poultry	3.4%	3.3%	2.5%	2.8%		Slaughter population [†]	Chicken		
Eggs	2.4%	2.0%					(Poultry: hens=1:1)	11,005,780	14,297,441
							rabbits	454,455	740,259

[*] Database of livestock products in the CAPSiM model are not completely in consistent with those in the China Rural Statistic Yearbooks, so we use the in. Population of horses, asses and mules is assumed to be stable according to historical trends and rabbit population shall grow by 5% annually.
[†] Use slaughter population for pigs, chickens and rabbits since they are alive for only part of a complete year before slaughtering.

Annex 4: Mitigation potential from the livestock sector

Projecting GHG emissions from the livestock sector under the BAU scenario

An attempt to validate these data assumptions revealed a disparity between our GHG emissions estimates from enteric fermentation and manure management, and those produced for the China national inventory (NCCC, 2012). Since assumptions underlying the latter cannot be publically accessed, this study assumed a percentage increase of the baseline emissions from 2005 (stated by the national GHG inventory, NCCC, 2012) until 2020, which was observed in our estimation. As such, livestock GHG emissions are projected to reach 742 Mt CO_2e in 2020, an increase of 51% compared to 2005 levels (NCCC, 2012)

Table G Selected livestock and grassland mitigation measures and target species

No.	Measure	Explanations	Target species
L1	Anaerobic digestion of manure	Implementation of on farm anaerobic digesters for storing livestock manure residues and converting some of the organic content to CH_4. CH_4 can be burned to produce heat or electricity for the livestock farm or sold to other consumers.	Cattle, dairy cows, pigs, poultry
L2	Animal breeding	Breeding techniques like artificial insemination of domestic livestock with high quality semen from breeding stock will generate a trade-off between decreasing rumen CH_4 production and improved feed intake, milk production, weight gain and production efficiency. This measure does not consider cross breeding.	Indoor - cattle, dairy cows, pigs, sheep, goat
L3	Tea saponins addition to the diet	Tea saponins are plant secondary compounds that are available in highly concentrated form in waste by products of tea production. Adding tea saponins to the diet of livestock is considered to increase the productivity while reducing rumen CH_4 production.	Indoor - cattle, dairy cows, sheep and goat
L4	Probiotics addition to the diet	Probiotics are commonly used in Chinese aquaculture industry but the application is uncommon for terrestrial livestock. Adding probiotics to the diet modifies the rumen ecosystem and thereby reduce the CH_4 production as well as improve the animal productivity and immune response.	Indoor - cattle, dairy cows, sheep and goat
L5	Lipid addition to the diet	Adding polyunsaturated fatty acids to the diet of livestock can effectively reduce the CH_4 production through suppression of rumen protozoa and inhibition of methanogens in the rumen and increase the productivity of the animal.	Indoor - cattle, dairy cows, sheep and goat
L6	Grazing prohibition for 35% of grazed grasslands	Grazing ban is a common technique in grazing systems for improving degraded grasslands. This measure considers a ban of 35% of the total grazed grassland in China. While the vegetation type is recovering, the dry matter production is improving. The grass will not be cut and thus grass residues can enter the soil to improve the soil organic matter content and increase the carbon sequestration rate.	Grazing - cattle, dairy cows, sheep and goats
L7	Reduction of stocking rate - medium grazing intensity	Chinese grasslands are usually overgrazed. This measure considers a stocking rate reduction to a medium intensity. While the grassland condition is improving, the dry matter production of the grasslands would increase by 10%. The grassland utilization rate is reduced to 50% and thus the higher amount of organic material entering the soil will increase the carbon sequestration rate.	Grazing - cattle, dairy cows, sheep and goats
L8	Reduction of stocking rate - light grazing intensity	This measure considers a light grazing intensity on Chinese grasslands. As a result the grassland utilization rate is reduced to 35% and the dry matter production increases by 3%. Similar to L9, the carbon sequestration rate increases due to a higher organic matter input to the soil.	Grazing - cattle, dairy cows, sheep and goats

Table H Mitigative effects and stand-alone abatement rates of livestock mitigation measures

Measure No.	Mitigative effects N_2O	CH_4	SOC	Abatement rate (per year) Cattle (%/hd)	Dairy cow (%/hd)	Pig (%/hd)	Sheep (%/hd)	Goat (%/hd)	Average (%/hd)	Grassland (tCO_2e/ha)	Anaerobic digester (tCO_2e/ digester)
L1	+	+									2
L2		+		-11	6	4	8	8	4		
L3		+		12	15		17	17	15		
L4		+		-0.2	0.3		1	1	1		
L5		+		8	6		4	4	4		
L6	+	+	+							1.07	
L7	+	+	+							0.7	
L8	+	+	+							0.88	

Treatment of measures interaction in the livestock sector

All three grassland (L6–L8) and dietary mitigation options (L3–L5) are mutually exclusive. Lacking more detailed data, we assume that grazing controls or intensities are implemented in approximately 1/3 of the total grazed grassland in China. Applications of multiple feed additives have no additive effect on emissions or productivity. Hence, multiple dietary mitigation options will not be applied simultaneously. To avoid double counting, an equal application of each of the 3 dietary mitigation options is assumed; i.e. all livestock receive only one feed additive.

Table I Livestock measure adoption rates under baseline and abatement scenarios

Measure No.	Historical or current adoption	Baseline adoption in 2020	Maximum feasible adoption in 2020	References or explanation
L1	33% of total 120 M possible farm-scale anaerobic digesters	66% of total possible farm-scale anaerobic digesters	33% of total possible farm-scale anaerobic digesters	NDRC (2007)
L2	Limited	most common for beef and cow but practically non-existent for goat farms	20% of beef and dairy cattle, 30% of sheep, 60% for goat	Waldron et al. (2007)
L3	Very limited	Very limited	10% of livestock since tea saponins are not sufficient available	Expert opinion
L4	10% of terrestrial livestock	Increasing adoption rate	50% of livestock	Wang et al. (2008) Beijing Shennong Agricultural Consultancy. (2013) Research Report on Feeding Probiotics Industry in China
L5	Limited	Limited	70% of livestock	Expert opinion
L6	In 2010, 40% of Chinese grassland is under grazing ban, suspended grazing, or rotational grazing.	In 2010, 60% of Chinese grassland is under grazing ban, suspended grazing, or rotational grazing.	33% of grazing grassland	18th formal announcement of the strategic objectives of the sustainable development of Chinese grassland (in Chinese) Ministry of Environmental Protection of People's Republic of China (2005 – 2011) Report on the State of the Environment of China (in Chinese) Brown et al. (2008)
L7	Limited	Limited	33% of grazing grassland	18th formal announcement of the strategic objectives of the sustainable development of Chinese grassland (in Chinese) Ministry of Environmental Protection of People's Republic of China (2005 – 2011) Report on the State of the Environment of China (in Chinese)

Table J Average abatement rate, additional application range and mitigation potential of livestock measures

Measure No.	Weighted abatement rate		Additional application	Annual mitigation potential in 2020
	(tCO2e/ha)	(CO_2e reduction in %/SU†)	(M ha)	($MtCO_2e$)
L1	2*		‡	58.66
L2		4.1	‡	4.4
L3		15.4	‡	5.53
L4		0.6	‡	1.09
L5		14.3	‡	30.76
L6	1.067		56.98	60.78
L7	0.705		57.85	40.77
L8	0.877		57.85	50.72
Total				252.71

* Per anaerobic digester

† Sheep unit (SU) is a standard unit to compare different animal species. The conversion equivalence is sheep: 1, goat: 0.9, cattle: 5, dairy cow: 7, pig: 0.8. It is only an approximate simplification and normally applied in grazing systems. Hence the costs/SU should be interpreted with caution.

‡ Here livestock numbers refer to Table F.

Table K Explication of and references for livestock measure implementation cost estimation

Measure No.	Explications	Major references
L1	The investment cost for an anaerobic digester on farm scale is about 3250 Yuan but a subsidy between 800 and 1200 Yuan is provided. The annual benefit of running a digester is estimated to be 500 Yuan. We assume that one anaerobic digester is operational for 15 years and a relative high failure rate of 8% of new constructed digesters due to immense maintenance and technological short comings	MOA (2007a) NDRC (2007) Zhang et al. (2012) Han et al. (2008)
L2	Costs for high quality genetic material, artificial insemination and administration are 20 Yuan, 40 Yuan, and 20 Yuan per animal, respectively (the costs are adjusted to current prices). Due to the low success rate more than one artificial insemination has to be done for one animal. The milk production and body weight will increase by 1% each year.	Waldron et al. (2007) Zhang and Beckman (2008) expert judgement†
L3	A sheep unit that is fed with 1g concentrated tea saponins per day shows increased milk production, body weight, and wool/cashmere production of 3%, 4%, and 4%, respectively. The feed intake increases by 2%. The costs are at ¥125/Kg.*	expert judgement†
L4	A sheep unit that is fed with 1g probiotics per day shows increased milk production and body weight of 6%. The feed intake increases by 5%. The costs are ¥50/Kg.*	Musa et al. (2009) expert jugement†
L5	A sheep unit that is fed with 40g poly unsaturated lipids per day shows increased milk production, body weight and wool/cashmere yield of 4%, 2%, and 2%, respectively. The costs are at ¥15/Kg.*	expert judgement†
L6 L7 L8	The cost assumptions for herders are based on farm surveys in Inner Mongolia. A simple model was generated that estimates the DM availability under different grazing intensities and hence the additional costs for supplementary feeding. Costs for machinery and labour input are based number of animals and area for hay making. We assume that the livestock is freely grazing. Thus, no costs a generated by grazing livestock.	Farm questionnaires by the Inner Mongolia Agricultural University.

*Additional management costs of ¥2/animal apply for purchasing, transporting, feeding the feed additives.

† Since there is a gap in Chinese Scientific literature for the required information, we consulted several Chinese experts on their judgment of impact on yields and costs. The results presented here are the mean of all assumptions.

Table L Cost considerations of livestock measure implementation

Measure No.	Target animals	Cost consideration factors (2010 level per sheep unit per year)				Application rate
		Investment costs (per year)	Administration cost	revenue	Yield increase (per head)	
L1	Cattle, dairy cows, pigs, sheep, goat, , poultry	¥3250	not available	¥500/year		Every 15 years
L2	Indoor - cattle, dairy cows, pigs, sheep, goat	¥60/head	¥20/head		1%	Annual
L3	Indoor - cattle, dairy cows, pigs, sheep, goat	¥1/head	¥2/head/year		3-4%	Daily
L4	Indoor - cattle, dairy cows, sheep, goat	¥18/head	¥2/head/year		6%	Daily
L5	Indoor - cattle, dairy cows, sheep, goat	¥219/head	¥2/head/year		2-4%	Daily
L6	Grazing - cattle, dairy cows, sheep, goats	*	*		1%†	Annual
L7	Grazing - cattle, dairy cows, sheep, goats	*	*		10%†	Annual
L8	Grazing - cattle, dairy cows, sheep, goats	*	*		3%†	Annual

* We assume free grazing on pasture which is most common in Chinese grassland systems. Additionally, we do not assume construction of new warm shed since the Chinese government increases the housing capacities strongly each year. Therefore, only costs regarding additional feeding and running housing facilities are applied.

† Increase of DM production /ha based on Patton et al. (2007).

Table M Unit cost and cost effectiveness of livestock mitigation measures

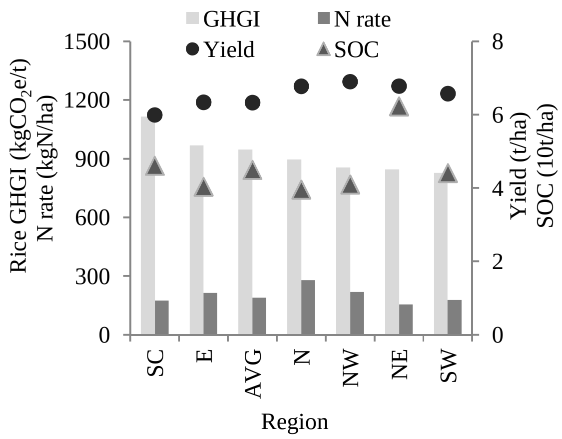
Masure No.	Cost in 2020 (¥/ha, 2010 price)	Cost in 2020 (¥/SU, 2010 price)	Cost effectiveness in 2020 (¥/tCO_2e, 2010 price)	Mitigation potential in 2020 ($MtCO_2e$)
L1	-500*		-32	58.66
L2		-29	-2571	4.4
L3		-3.4	-56	5.53
L4		-17	-7079	1.09
L5		109	1950	30.76
L6	300		281	60.78
L7	45		64	40.77
L8	283		322	50.72

* Per anaerobic digester.

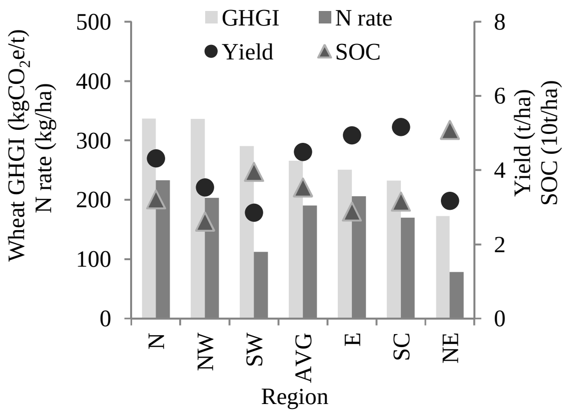
Annex 5: GHGI at regional level in 2010 and implications for mitigation strategies

The GHGI, yield and synthetic N rate of rice, wheat and maize cultivation as well as the SOC content at the regional scale in 2010 are illustrated in Fig. A. In general, the southwest had lowest cereal yields, albeit second highest SOC after the northeast. Conversely more N fertilizers were added to croplands in northwest provinces to compensate poor soil fertility, resulting in elevated regional GHGI of crop production. Fig. A reveals that yield levels do not necessarily correspond to local SOC status, since productivity is also influenced by climate, precipitation and other factors. In this regard, regional strategies to minimize GHGI and improve soil fertility should accommodate local climatic, soil and water conditions and management practices. For example, in the northwest measures improving SOC density (e.g. conservation tillage) should be favored to enhance soil fertility and land productivity. In intensive cropping systems in east and north China where over-fertilization is prominent, more efficient use of N fertilizer can allow N rates to be cut by 30 to 60% without sacrificing crop yields (Ju et al. 2009). Although the northeast was the least carbon intensive region in cereal production, this came at the expense of net carbon losses, especially in Heilongjiang Province (Pan et al. 2010; Yu et al. 2012), thus calling for better management practices to sustain soil fertility in this region.

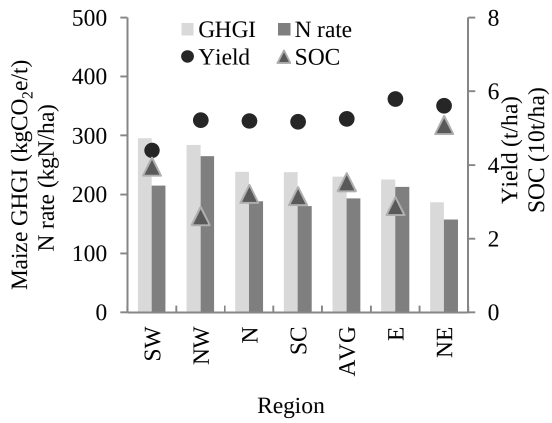
Fig.A GHGIs of rice (a), wheat (b) and maize (c) production in different regions in 2010 and their relationship with yield, N rates and SOC content.



(a)



(b)



(c)

Note: NE, N, NW, E, SC, SW and AVG refer to northeast, north, northwest, east, south and central, southwest China, and national average, respectively.

Source: Results calculated by the author and incorporated in Wang et al. (2014)

Annex 6: Estimates of N_2O abatement rates of mitigation measures C1-C4

	BAU scenario						Mitigation measure C1				Mitigation measure C2 and C3				
	N rate		Yield		PFP_N		Target PFP_N (70% of optimum)	N rate reduce	N_2O emission reduction	Abatement rate	Optimal PFP_N	Yield	N reduce quantity	Abatement rate	Current N rate
	2010	2020	2010	2020	2010	2020									
Provinces	(kg/ha)		(t/ha)		(kg/kg)		(kg/kg)	(%)	(%)	(tCO_2e/ha)	(kg/kg)	(t/ha)	(kt)	(tCO_2e/ha)	(kg/ha)
Rice															
Tianjin	259	246	7.04	7.40	28.3	30.0	35.0	14.2	11.4	0.08	50.0	7.8	0.8	0.13	155
Hebei	344	328	6.79	7.14	18.7	21.8	35.0	37.8	30.7	0.28	50.0	7.5	3.8	0.12	150
Inner mengolia	221	211	7.23	7.60	33.4	36.1	35.0				50.0	8.0	4.3	0.12	160
Liaoning	235	224	7.38	7.76	31.4	34.6	52.5	34.0	27.6	0.17	75.0	8.1	22.5	0.09	109
Jilin	170	162	8.29	8.72	48.9	53.9					75.0	9.2	22.8	0.09	122
Heilongjiang	120	114	6.47	6.80	54.0	59.5					75.0	7.1	42.0	0.04	95
Shanghai	326	311	8.28	8.71	24.8	28.0	36.1	22.5	18.2	0.16	51.6	9.1	6.0	0.15	177
Jiangsu	290	277	8.03	8.45	27.7	30.5	36.1	15.4	12.4	0.10	51.6	8.9	119.9	0.14	172
Zhejiang	227	217	7.06	7.42	31.1	34.3	36.1	5.1	4.0	0.02	51.6	7.8	44.0	0.13	151
Anhui	205	195	6.22	6.54	30.4	33.5	38.9	13.9	11.2	0.06	55.6	6.9	86.1	0.10	123
Fujian	159	152	5.94	6.24	37.3	41.1					50.5	6.6	16.3	0.05	130
Jiangxi	164	156	5.71	6.00	34.9	38.5					50.5	6.3	88.8	0.07	125
Shandong	285	272	8.35	8.78	29.3	32.3	35.0	7.7	6.1	0.05	50.0	9.2	7.6	0.15	184
Henan	213	203	7.40	7.78	34.8	38.4	35.0				50.0	8.2	20.9	0.09	163
Hubei	168	160	7.73	8.12	46.1	50.9					55.0				160
Hunan	148	141	6.34	6.66	42.8	47.2					56.5	7.0	60.5	0.04	124
Guangdong	191	182	5.33	5.60	28.0	30.8	35.3	12.7	10.2	0.05	50.5	5.9	71.0	0.10	117
Guangxi	196	186	5.32	5.60	27.3	30.0	35.3	15.0	12.1	0.06	50.5	5.9	76.8	0.10	116
Hainan	144	137	4.50	4.73	31.2	34.4					50.5	5.0	10.7	0.09	98
Chongqing	143	136	7.65	8.04	53.6	59.1					50.0				136
Sichuan	201	192	7.47	7.85	37.5	41.0					50.0	8.2	47.0	0.06	165
Guizhou	134	127	6.52	6.86	48.8	53.8					50.5				127
Yunnan	229	219	6.09	6.40	26.8	29.3	35.3	17.2	13.8	0.09	50.5	6.7	42.6	0.11	133
Shanxi	180	172	6.64	6.98	37.1	40.7					50.0	7.3	2.7	0.06	147
Ningxia	283	270	8.31	8.74	29.3	32.3	35.0	7.6	6.0	0.05	55.0	9.2	5.8	0.19	167
Nation average	186	177	6.57	6.90	23.2	38.9		7.3		0.08		7.2	803.0	0.08	133
Wheat															
Beijing	239	286	4.95	5.34	20.5	18.7	24.9	24.9	19.4	0.33	35.6	5.6	3.2	0.18	158

Tianjin	233	278	4.86	5.25	20.7	18.9	24.9	24.2	19.0	0.31	35.6	5.5	5.6	0.18	155
Hebei	248	296	5.09	5.50	20.6	18.6	24.9	25.4	19.6	0.34	35.6	5.8	129.5	0.19	162
Shanxi	163	194	3.24	3.50	19.9	18.0	24.9	27.8	21.0	0.24	35.6	3.7	24.5	0.12	103
Inner mengolia	292	349	3.19	3.44	10.9	9.9	14.9	33.9	24.3	0.50	21.4	3.6	28.9	0.19	169
Heilongjiang	97	115	3.67	3.96	38.4	34.4					35.6				115
Jiangsu	238	284	4.82	5.20	20.3	18.3	27.1	32.4	23.4	0.39	38.7	5.5	97.4	0.16	141
Anhui	189	225	5.03	5.42	26.8	24.1					35.6	5.7	141.1	0.29	160
Shandong	206	246	5.77	6.23	28.1	25.3					35.6	6.5	203.1	0.28	184
Henan	183	219	5.81	6.27	32.0	28.7					35.6	6.6	163.3	0.18	185
Hubei	161	193	3.35	3.62	20.8	18.8	24.9	24.6	19.2	0.22	35.6	3.8	35.3	0.12	107
Chongqing	101	120	3.07	3.31	29.0	27.5					35.6	3.5	3.5	0.11	98
Sichuan	124	148	3.34	3.60	26.9	24.3					35.6	3.8	49.5	0.19	106
Yunnan	113	135	1.72	1.86	15.1	13.8	24.9	44.6	30.1	0.24	35.6	1.9	7.8	0.06	55
Shanxi	232	276	3.43	3.70	14.9	13.4	24.7	45.7	30.6	0.50	35.2	3.9	41.8	0.13	110
Gansu	189	226	2.84	3.07	15.0	13.6	24.7	45.0	30.2	0.40	35.2	3.2	27.7	0.10	91
Qinghai	91	108	3.82	4.12	44.0	38.1					35.2				108
Ningxia	238	284	3.28	3.54	13.8	12.5	24.7	49.4	32.7	0.54	35.2	3.7	7.4	0.12	105
Xinjiang	238	284	5.51	5.94	23.2	20.9	24.7	15.2	14.1	0.24	35.2	6.2	58.9	0.20	177
Nation average	199	238	4.75	5.13	23.9	21.5		15.2		0.35		5.4	1030.5	0.19	155
Maize															
Beijing	213	233	5.86	6.79	25.2	29.1	32.7	11.1	18.5	0.25	46.7	7.3	8.5	0.19	157
Tianjin	201	220	5.37	6.22	26.6	28.3	32.7	13.5	20.0	0.26	46.7	6.7	8.7	0.17	144
Hebei	172	188	5.02	5.81	29.3	30.9	32.7	5.6	15.0	0.17	46.7	6.3	144.1	0.16	134
Shanxi	181	198	4.80	5.56	26.6	28.1	32.7	14.2	20.4	0.24	46.7	6.0	68.6	0.15	128
Inner mengolia	214	235	5.80	6.71	27.0	28.6	32.7	12.6	19.4	0.27	46.7	7.2	137.7	0.19	155
Liaoning	198	216	5.57	6.44	28.2	29.8	33.7	11.6	18.8	0.24	48.1	7.0	104.9	0.17	145
Jilin	178	194	6.61	7.65	37.3	39.4					48.1	8.3	76.5	0.21	172
Heilongjiang	136	149	5.06	5.85	37.1	39.2					48.1	6.3	80.7	0.17	131
Jiangsu	237	259	5.30	6.14	22.5	23.7	32.6	27.5	28.9	0.44	46.6	6.6	20.9	0.17	142
Anhui	211	230	4.11	4.76	19.6	20.7	32.6	36.7	34.7	0.47	46.6	5.1	29.6	0.13	110
Shandong	215	235	6.56	7.60	30.7	32.3					46.6	8.2	195.9	0.38	176
Henan	183	200	5.64	6.53	30.8	32.6					46.6	7.0	160.3	0.32	151
Hubei	256	280	4.85	5.61	19.0	20.0	32.6	38.5	35.9	0.59	46.6	6.1	23.9	0.16	130
Guangxi	247	270	4.11	4.75	16.7	17.6	32.6	46.1	40.7	0.65	46.6	5.1	21.0	0.13	110
Chongqing	232	254	5.39	6.24	23.3	24.5	32.2	23.7	26.5	0.39	45.9	6.7	24.6	0.18	147
Sichuan	254	278	4.86	5.62	19.4	20.2	32.2	37.1	35.0	0.57	45.9	6.1	64.7	0.16	132
Guizhou	177	194	5.35	6.19	30.5	31.9					46.6	6.7	43.1	0.32	143

Yunnan	295	323	4.11	4.75	13.9	14.7	31.7	53.6	45.5	0.86	45.3	5.1	56.6	0.14	113
Shanxi	271	296	4.40	5.09	16.3	17.2	32.8	47.7	41.7	0.73	46.9	5.5	50.1	0.14	117
Gansu	274	300	4.73	5.47	17.3	18.3	32.8	44.4	39.6	0.70	46.9	5.9	31.5	0.15	126
Ningxia	279	305	7.29	8.44	26.6	27.7	32.8	15.7	21.4	0.38	46.9	9.1	15.4	0.23	194
Xinjiang	263	288	6.82	7.89	26.2	27.4	32.8	16.5	21.9	0.37	46.9	8.5	40.8	0.22	182
Nation average	202	221	5.62	5.86	26.9	28.4		15.6		0.40		6.8	1408.0	0.21	146

	BAU scenario				Mitigation measure C1				Mitigation measure C4					
	N rate 2020	Yield 2020	Area 2020	PFP_N 2020	Target PFP_N (10% or 15% increase)	N rate reduce	N_2O emission reduction	Abatement rate	Optimal PFP_N	Yield increase by 6% or 10%	Abatement rate (N rate reduce)	Abatement rate (EF change)	Total Abatement rate	Current N rate
Crop Type	(kg/ha)	(t/ha)	(kha)	(kg/kg)	(kg/kg)	(%)	(%)	(CO_2e /ha)	(kg/kg)	(t/ha)	(CO_2e /ha)	(CO_2e /ha)	(CO_2e /ha)	(kg/ha)
Greenhouse vegetable	656	55.2	3,560	84	97	15	31.8	1.225	160	60.7	0.936	0.440	1.376	379
Openfield vegetable	262	38.1	15,479	145										210
N overuse area	315		7,740	121	133	10	27.3	0.505	200	41.9	0.389	0.440	0.829	209
Normal area	210		7,740	181										210
Fruits	565	24.5	11,668	43										350
N overuse area	678		8,168	36	42	10	31.8	1.266	70	26.0	1.079	0.748	1.827	371
Normal area	301		3,501	81										301
Cotton	237	1.5	5,168							1.6	0.463	0.440	0.903	

Annex 7: Considerations for measure implementation cost estimation

Measure No.	Considerations	Major references
C1	National average reductions in N rates were aggregated from those in target regions (see Table in Annex 7 measure C1) and are presented in Table 3-3.	Zhang et al. (2009) Zhang et al. (2012b)
C2	More labors are required for the additional wheat topdressing.	Zhang et al.(2009)
	Increased machine inputs for deep fertilizer placement for maize cultivation.	SAIN(2012a)
C3	More labor inputs for the additional topdressing; irrigation costs saved thanks to improved irrigation regimes in rice paddies.	Liu et al.(2006) Zhang(2012b)
C4	Reductions in N fertilizer rates were aggregated from those in target regions (See Table in Annex 7 measure C4) and the national average is shown in Table 3-3. We used Calcium Ammonium Nitrate (total nurient≥34%, total N=26%) to represent nitrate-based fertilizers and its application rate stands at 450kg/ha for vegetable and cotton and 750kg/ha for fruit. High-efficient irrigation systems allow for labor savings from vegetable and cotton productions; more labors are required for split application of fertilizer in fruit production. Subsurface drip irrigation system costs compromise ¥ 15000 initial investment and installation cost (lifespan=10years) per hectare and annual maintenance and renewal cost of smaller diameter polytube at ¥1500/ha and film input at ¥1000/ha, while labors and pesticides (cotton) and irrigation costs will be saved.	Yang et al. (2005) Huo et al. (2011) Zhang et al. (2012b)
C5	Used NI CDC to represent additional cost of enhanced-efficiency fertilizers: in general DCD is applied at rates equivalent to 5% of N nutrient (w/w), the price of DCD is about ¥ 10000/t.	Bai et al. (2012) Liu et al. (2013)
C6	Material and labor inputs for manure composting and disposal are represented by the market price of organic manure fertilizer. More labor inputs are needed for large quantity of manure application	Huang et al. (2010)
C7	Long-term no-till could lead to excessive soil surface compaction, weed spread and pest infestation. It is recommended that deep loosing should be carried out every 3-4 years. Increased seed and pesticide costs are attributed to crop residues return to lands.	He et al. (2006) Lv et al. (2010) Wang et al. (2010)
C8	Increased machine cost is for straw mulching following harvest. Additional N fertilizers should be added to accelerate fresh straw decay. Large amount of straw is likely to affect seed emerging and encourage weed growth and pest infestation, therefore seeding rates need to be increased.	Jiang et al. (2006) Liu et al. (2009) Tian et al. (2011)
C9	Biochar price is represented by the straw pyrolysis product from Sanli NewEnergy Company, Henan, China. More labors are required to apply large amount of biochar. Per tonne biochar price is considered constant thanks to technology improvement. Domestic experts suggest applying biochar every 5 years since single application can provide beneficial effects over several growing seasons in the field.	Major(2011) Zhang et al.(2012a) Pan (2012)

Annex 8: Comparative table of market design among pilots

Pilot ETS system		Beijing	Chongqing	Guangdong	Hubei	Shanghai	Shenzhen	Tianjin
ETS Structure	Start Date	28-Nov-2013	19-June-2014	19-Dec-2013	2-Apr-2014	26-Nov-2013	18-Jun-2013	26-Dec-2013
	Total GHG Emissions	103 MtCO_2e (2010)	125 MtCO_2e (2010)	510 MtCO_2e (2010)	952 MtCO_2e (2010)	211/240 MtCO_2e (2010)	83.4 MtCO_2e (2010)	134 MtCO_2e (2010)
	Issued cap	60 Mt each year	125 MtCO2e for 2013, then subject to annual reduction of -4.13%	388 MtCO2(350 allowances+38 reserve)	324 MtCO2 each year (298 allowances+26 reserve & auctions)	About 150 MtCO_2 for 2013	ab. 100 MtCO2 total for 2013-15 ab. 30 Mt issued for 2014. 2013 surplus of ab. 10% of allocation.	80 MtCO_2 each year
	Trading center	CBEEX	CCEEX	CEEX in Guangzhou	HCEEX	SEEEX	CEEX in Shenzhen	CTEEX
	Allowance name	BEA	CQA	GDEA	HBA	SHEA	SZA	TJEA
	Guidance regulation	Interim Measures for the Administration of Carbon Emissions Trading in Chongqing	Interim Measures for the Administration of Carbon Emissions Trading in Chongqing	Interim Measures for the Administration of Carbon Emissions Trading in Guangdong	Interim Measures of Hubei carbon emissions trading	Interim Measures for the Administration of Carbon Emissions Trading in Shanghai	Provision of Carbon Emissions Trading Management of Shenzhen	Interim Measures for the Administration of Carbon Emission Trading in Tianjin
Coverage	GHG covered	CO_2 (direct and indirect)	6 GHGs (direct and indirect)	CO_2 (direct and indirect)	CO_2 (direct and indirect)	CO_2 (direct and indirect)	CO_2 (direct and indirect)	CO_2 (direct and indirect)
	GHG emissions cov.	49%	39.5%	40%	35%	57%	54%	60%
	Sectors	Electricity providers, heating sector, manufacturers (automobile, cement, petrochemicals) and major public buildings (health, education, banking, …)	Production of electrolytic aluminum, ferroalloys, calcium carbide, cement, caustic soda, iron and steel.	Power, cement, steel, iron, petrochemicals). Textile, non-ferrous metals, plastic, paper may be included later. Transports and buildings (public, commercial) construction are part of the newly released regulation (from March 1st).	13 sectors: power plants and industrial companies (iron and steel, cement, chemicals, automobile, manufacturing, nonferrous metals, glass and paper)	16 sectors: industrial sectors (electricity, iron & steel, petrochemical, non-ferrous metal, chemical, building materials, textile, pulp & paper, rubber, chemical fiber), other sectors (aviation, ports, railway, commercial, hotel and financial sector buildings).	Almost all sectors. 26 sectors for now. Including industrial companies, building sector and electricity generators. Future transport inclusion under consideration.	Iron and steel producers, chemical facilities, power and heat generators, oil and gas exploitation, civil buildings.

	Liable entities & Mandatory reporting	About 490 entities Threshold: 10ktCO2/yr (average of 09-11) Mandatory reporting and voluntary participation. Threshold: 2k tce/yr energy consumption.	242 entities Threshold:20 ktCO2/yr (any year of 2008-2012), and new installation after 2010: 20 ktCO2/yr	184 liable entities with > 20 ktCO2/yr (any year of 2011-2014) New regulation: Industry > 10 ktCO2/yr, Non industrial sectors: with > 5 ktCO2/yr. Transport: threshold TBD Mandatory reporting when > 5 ktCO2/yr,	Province's 138 biggest emitters Threshold: 120k tCO2e/yr (any year of 2010-2011). Mandatory reporting Threshold: 8ktce of energy consumed/yr.	191 companies Threshold: 20 ktCO2/yr (any year of 2010 or 2011) for industrial companies; 10 ktCO2e/yr for other sectors. Mandatory reporting for about 600 firms. Threshold: 10 ktCO2/yr.	635 city's biggest companies (2013). Threshold: 5 ktCO2e/yr to be lowered to 3kt in 2014. 197 large buildings. Threshold: 20,000m2 for public buildings and 10,000m2 for state office buildings. Mandatory reporting. Threshold: emissions between 3-5 ktCO2e/yr. + other specific firms and buildings.	114 entities Threshold: 20 ktCO2/yr (any year since 2009) Mandatory reporting for carbon intensive industries and civil buildings with > 10 ktCO2e/yr (steel, iron, power, heating, (petro) chemicals).
	New entrants and activity change	Entities with emission change of > 5 ktCO$_2$/yr or >20% are liable to request allowance change.	Compliance obligation in case of closure.	New entrants reserve (20Mt). New project (including capacity extension or reconstruction) with > 10 ktCO2/yr should purchase all quotas prior to operation. Quota reallocation for activity change, reduction and closure.	21.43% of the cap is set aside for new entrants	In case of closure or displacement of activity, compliance obligation is due and 50% of following-year allowances after obligation shall be taken back.	Reserve (2% of total cap). New fixed-asset projects with over ¥ 200 million investment should submit emission evaluation report. In case of closure or displacement of activity, compliance is due and 50% of following-year allowances shall be taken back.	Compliance obligation in case of closure.
Allocation	Compliance Period	One year	One year	One year	One year	One year	One year	One year
	Trading Period	2013-2015	2013-2015, with backdating for 2013	Phase I: 2013-2015 Phase II: 2016-2020 Phase III: post 2020	2014-2015	2013-2015	2013-2015	2013-2015
	Form of allocation	Free allocation: 99.9% in 2013 to 99.5% in 2015 for coal-fired plants, 98% to 94% for manufacturers. Absolute reductions for manufacturing and service. Small amount reserves may be auctioned.	Free allocation only, based on historical emissions, linearly decreasing with time (-4.13% per year). It means it is the first China's ETS to explicitly enforce a declining emissions trajectory as from 2014.	Free allocation of 97% in 2013-2014. Free allocation contingent on the purchase of 3% of individual cap at auctions. This share will rise to 10% in 2015 and 50% in 2020 for power generators, with possibility to complete purchase on 2ndary market. Actual auctions: 29 Mt/ year.	Mainly free allocation (97% of 2010 emissions, to decline by 1% a year) 7.8 Mt auctioned each year. Minimum price set at 20 yuan.	Free allocation only. Auctioning under consideration. Early action over 06-11 rewarded with additional permits.	At least 90% of cap is freely allocated. Auctioning, fixed-price sell will be complementary methods. Auctioned quota should be <3% cap. Absolute emissions growth limited to 10% by 2015 compared to 2013 levels.	Free allocation. Auctioning or fixed-price sell may be used only in case of large market price fluctuation, and generated revenue shall subsidize emission reduction activities.

	Allocation mechanism for free allowances	Free allocation based on 09-12 emissions (industry) or carbon intensity (power), corrected by a sector-specific factor, declining with time. New entrants' allocation based on benchmarks.	Free allocation based on the highest annual emissions of 2008-2012,	For P1: mainly grandfathering based on 2010-12 emissions, considering sectors' characteristics.	Based on 2010 emissions. One third is held back until firms report emissions: only power generators received half of their endowment, i.e., only 193 Mt have been issued yet.	One-off free allocation for 2013-2015 based on 2009-2012 emissions, growth considered. Whenever possible (electricity and aviation), benchmarks will be used.	Free allocation based on firms' 2009-11 historical emissions, performance and future activity level. New entrants' allocation based on benchmarks.	Free allocation based on 2010-11historical emissions (existing entities) and benchmarks (new entrants).
Flexibility	Banking & Borrowing	Banking allowed within pilot phase. Borrowing forbidden. Only spot trading allowed.	x	Banking allowed within pilot phase (P1). Borrowing forbidden. Only spot trading allowed.	No banking allowed. Annual surplus permits will be cancelled.	Banking allowed within pilot phase. Borrowing forbidden.	Only spot trading is allowed. Both banking and borrowing forbidden.	Banking allowed within pilot phase.
	Offsets & Credits	Up to 5% of CCERs are allowed for compliance obligation. At least half of used CCERs must originate from local projects (except certain types of projects owned by liable entities). Local carbon credits from energy conservation and forestry are also eligible	Only CCERs from local projects (excluding hydro) allowed for up to 8% of compliance obligation.	CCERs allowed for up to 10% of compliance obligation, 70% of which must stem from local projects.	Only CCERs from projects located in Hubei (Forestry projects incl.) Up to 10% of compliance obligation.	Up to 5% of annual compliance obligation.	Up to 10 % of CCERs are allowed for compliance obligation.	Up to 10 % of CCERs are allowed for compliance obligation.
	Price management & Cost Containment	Auction or government buying back permits from the market. Holding serve for macro control (not exceed 5% of cap)	x	Market price adjustment quota reserve (18 Mt). Minimum price for auctions set at 60 yuan. Necessity to buy auctioned permits to use and trade free allowances.	Holding reserve for macro control (8% of cap). A 100 million yuan reserve is planned for the market intervention. Price floor at auctions (20 yuan).	Holding reserve under consideration for market control, including government buy/sell in the market.	Allowance reserve (2% of total allowances + rest allowance of auction+ government's purchase) to control price.	Government buy/sell in the market.

Compliance	MRV	Yearly emissions reports submitted by April 15. Verification by accredited 3rd party required and report submission before April 30. Allowance allocation by June 30 and compliance by June 15. Compliance information dissemination before end July.	Compliance before June 20 2015 or Phase I obligation. Compliance before June 20 2016 for Phase II obligation.	Allocation on July 1 each year. Compliance before June 20. Verification by a third party is required.	Yearly emissions reports submitted by February 28, and verification report submitted by a third party before April 30. Yearly allowance allocation before June 30. Compliance before end May.	Following-year monitoring report submitted by December 31. Yearly emissions report by March 31 and verification report by a 3rd party submitted by April 30. Annual compliance between June 1 and June 30.	Yearly emissions report submitted by March 3 and verification report submitted by a third party before April 30. Yearly allowance allocation before May 31. Compliance by June 30. Not allowed to use the same verification agency for 3 consecutive years. Compliance information dissemination before end July.	Following-year monitoring report submitted by November 30. Yearly emissions report and verification report by a 3rd party submitted by April 30. Compliance by May 31. Not allowed to use the same verification agency for 3 consecutive years.
Compliance	Enforcement/Penalty (including administrative penalty)	Failed to submit emission report in due time: rectification and then ¥ 50k/firm. For non-compliance:3-5 times market carbon price for every missing allowance.	Failed to submit emission report in due time or refusal of validation: ¥ 20k-50k/firm. For non-compliance penalty: 3 times average market price of the month before surrender	Fraud in emission report: rectification and then ¥ 10k -30k penalty/firm. Impediment of verification work: rectification and then ¥ 10k -30k up to 50k penalty/firm. For non-compliance penalty: deduct 2 times the missed quota from next year's allowances and 3 times average market price. Also penalties for fraud of trading exchange and verification entities.	Fraud in emission report: rectification and halve next year's allowances. Fraud in trading and other activities: rectification and <150k penalty. For non-compliance penalty: deduct 2 times the missed quota from next year's allowances and 2 times market carbon price for every missing ton. Also penalties for fraud of trading exchange and verification entities.	Failed to submit emission report in due time: or fraud in emission report rectification and then ¥ 10-30k/firm. Impediment of verification work: rectification and then ¥ 30k -50k /firm. Non-compliance penalties range from 5,000-100,000 RMB/firm. Also penalties for fraud of trading exchange and verification entities.	Failed to submit emission verification report in due time or fraud: rectification and then ¥ 10-100k/firm. For non-compliance: reduction from following year's allowances and 3 times the market price for every missing allowance. Also penalties for fraud of trading exchange and verification entities.	Levels and details of penalties not specified.
Compliance	Reporting date	April, 15th	April, 20th	x	Last working day in Feb.	April, 15th	March, 31st	April, 30th
Compliance	Surrender date	June, 15th	June ,20th	x	Last working day in May.	June, 1st to June, 30th	June, 30th	May, 31st
Other	Linkage	Encourage regional linking.	x	Potential linking with Hubei announced in 2011. Open to potential linkage with EU or California. Encourage regional linking.	Potential linking with Guangdong announced in 2011. No information since then. Deemed unlikely. Encourage regional linking.	Encourage linking with other ETS.	x	Encourage regional linking.
Other	Allowed participants	Compliance entities.	Compliance entities.	Compliance entities.	All except DOEs and	Compliance entities.	Compliance entities.	Compliance entities.

		Investment institutions, individuals are not allowed	individuals and investment institutions, both domestic and international	individuals and investment institutions	banks	individuals and investment institutions	individuals and investment institutions	individuals and investment institutions
	Transaction modes	Spot, agreement transfer.	x	Listed bidding, check bidding, agreement transfer.	Fixed-price transfer & negotiation bargaining	Listed trade, agreement transfer.	Spot, electronic bidding, block transaction.	Web Spot, agreement transfer, auction.
	Remarks	Floor and ceiling prices were put forward but negotiations have stalled due to lobbying over fears of higher costs.	Was the least developed of the 7, notably due to technical issues (e.g. on building the registry) and city-level politics scandal (aftermath of the fall of Bo Xilai).	Biggest provincial economy, it will be the biggest market of the 7. Only scheme to auction some allowances for now. Four auctions have been carried out till April 3, 2014.	Private investors and foreign trading houses (belonging to other pilots) may be allowed to trade Hubei carbon permits	Only pilot of the 7 to cover aviation. Trades are de facto not restricted to spot exchanges since vintage 14-15 permits are already owned and have already changed hands.	First to kick off, it is also the smallest of the 7. Will share market design strategies and experiences with California. Double-counting of emissions from scope 2 sectors and electricity generators.	Peculiar treatment of double-counting issues: if both supplier and consumer are regulated they both need to hand over a permit for the same ton of CO_2.

Sources: Thomson Reuters Point Carbon, ICAP Interactive Map, local regulations when available (CEEX, CBEEX, CTEEX, CNEMISSION). Sometimes information comes from personal communications. Information presented herein must therefore not be taken for granted, all the more so that regulations are evolving.

Notes: TCE stands for Metric Tons of Coal Equivalent. 1 TCE is equivalent to about 2 tCO_2 emissions.

Annex 9: VER rules and development of CCER projects

Eligible methodologies

The VER Measures require all projects aiming to obtain CCERs to use methodologies approved by the NDRC. As of August 2014, there are 178 such approved methodologies, 173 of which stem directly from existing CDM methodologies with modifications according to China's circumstances, including the notoriously controversial HFC-23 and N_2O adipic acid destruction methodologies that are now banned from use in the EU ETS. The 5 new non-CDM methodologies target emissions reductions from forestry (forestation, bamboo forestation and improved forest management) and land use (sustainable grassland management) as well as SF_6 gas insulation metal seal combination electric appliance. Similar to the CDM process, a relevant project design document must be attached when submitting a new methodology for approval.

Project eligibility

Four kinds of projects are eligible to request for registration with NDRC: (*i*) new projects using methodologies registered with the NDRC; (*ii*) CDM projects already approved by the NDRC (acting as the designated national authority (DNA) in the CDM process) but not yet registered with the CDM EB; (*iii*) CDM projects which had previously generated emission reductions prior to registration with the EB, e.g., pre-CDM credits (should be without CER issuance); (*iv*) registered CDM projects yet not issued CERs.

The project starting time should not be prior to February 16th, 2005. To be granted CCER credits, the interim VER Measures recognize GHG reductions from the 6 approved GHGs under the UNFCCC or carbon removal enhancements achieved by an offset project. Type (*iii*) credits are a subject it is noteworthy to dwell on for a moment. Registered CDM projects that have already been issued CERs can request CCER issuance for pre-CDM registration emission reductions only, all the while being allowed to remain in the CDM registry for further CER issuance. Again, registered CDM projects that have not yet being issued CERs can apply for CCERs for pre-registration emission reductions but are allowed to choose whether to remain under the CDM to receive CERs for planned reductions or to switch to the Chinese pipeline and request CCER issuance for these planned reductions, like type (*iv*) credits.

However there is still much uncertainty left for the time being. First, it is unclear whether projects no longer eligible under the EU ETS (e.g. projects destroying HFC-23 and N_2O), albeit theoretically eligible for credits, and those previously receiving NDRC approval but rejected by the EB, would actually generate CCERs for the domestic market. The latter projects will certainly have to reapply with modified documents to the NDRC. No such projects have sought approval so far. It is also still questionable as to whether NDRC will in turn approve pre-registration credits from CER-issued CDM projects to enter its offset market.

Project registration

While foreign as well as national entities and individuals are allowed to buy CCERs, only business entities registered in the P.R.C are authorized to apply for project registration. Central-level, large state-owned enterprises (SOEs) supervised by the State-owned Assets Supervision and Administration Commission of the State Council (SASAC) are allowed to request project registration directly with the NDRC; other business entities must first get approval from relevant provincial DRCs – the counterpart of the DNA under the CDM. SOEs are therefore likely to be subject to shorter registration periods. Similar to the CDM process, project validation by a NDRC-accredited third party is also necessary, prior to request for registration. The CCER Project Validation and Verification Guidelines were published by the NDRC in November 2012, outlining requirements for entities seeking accreditation with the NDRC as well as the principles, procedures and requirements of validation and verification. As of August 2014, China Quality Certification Center, Guangzhou CEPREI Certification Body, China Environmental United Certification Center, Foreign Economic Cooperation Office of MEP, China Classification Society Quality Assurance Ltd and Beijing Sino-Carbon Ltd have been accredited as eligible validators and verifiers for CCER projects. These six accredited CCER auditors are all domestic Designated Operational Entities (DOEs) under the CDM. This list may extend but it seems unlikely foreign DOEs will be able to get accreditation. The guidance on public review, document review, possible site visit and other procedures resemble those in the CDM Validation and Verification Manual.

CCER issuance and transactions

To request CCER issuance, the verification report, completed by a qualified verifier, along with the monitoring report, needs to be submitted to the NDRC. Project validation and verification are allowed to be carried out by the same entity, except for projects with annual emission reductions exceeding 60ktCO_2e. As of August 2014, about 285 projects have entered or completed the "validation" process on the China Certified Emission Reduction Exchange Info-Platform. Most of these projects are wind, hydro and solar energy and 40% have registered with the CDM EB (type (iii)) with an estimated annual emission reduction of 56 MtCO_2e while 45% belong to type (i). It is worth noting that there are about 20 rural household biodigesters projects which all fall under project type (*iii*) claiming pre-CDM credits and 2 forestry projects- one in Guangdong intending to request CCERs through carbon-sequestration by afforestation. Hubei topped the location lists with 27 projects requesting validation, followed by Guangdong with 23 projects, while other pilots host only 1-4 projects. As of August 1st 2014, 49 projects have successfully registered at NDRC, composed of 11 type(i) projects, 2 type(ii) projects and 32 type (iii) projects including the Guangdong afforestation project.

CER transactions will be restricted to trading platforms recognized by the NDRC, and their trading systems should be connected to the national VER Registry to track real time transfer of CCERs. All the trading platforms in the 7 ETS pilots have been authorized to host CCER trading. Until January, two transactions have been completed for yet-to-be-issued CCERs, with two branches of China National Petroleum Corporation purchasing 10k CCERs each, from two wind power projects at a price of ¥16 and ¥20 per ton.

Annex 10: Discussions on levying carbon tax in China

As early as September 2009, research institutes attached to the MOF, MEP and NDRC have initiated relevant research studies on carbon tax in China. Released reports gave different suggestions in terms of tax rate, introduction period, taxpayers, use of tax revenues and other aspects. The NDRC wants to separate the carbon tax as an individual tax, attributable to the regulation of the National Energy Bureau while the MEP favors it included in the list of environmental tax. It is reported that carbon tax program was being discussed in the NPC. The report released by the MOF suggested levying a carbon tax in China following the reform of resource tax while the NDRC and the State Administration of Taxation do not regard such a reform as a premise to imposing carbon tax. Regarding tax rate, the MOF report suggested a lower initial tax rate to only 10 yuan/t starting between 2013 and 2015 and gradually increasing to attain 40 yuan/t in 2020 while that of MEP recommended 20 yuan/t as the starting point to reach 50 yuan/t in 2020. In terms of taxpayers, the NDRC think energy producers should be liable to a carbon tax, at least in the initial stage, to avoid the challenge of emission accounting and minimize management cost, while the MOF and MEP believe imposing energy consuming companies to be more reasonable. NDRC favors tax revenue to be recycled to subsidize emission reduction actions while the MOF believes it should be directly included in public finance budget. However, carbon tax has not been put on the agenda, indicating no consensus being reached among decision-makers and more research and debate are expected on this topic before any decisions can be made.