environmental science & policy 13 (2010) 688–694

available at www.sciencedirect.com

journal homepage: www.elsevier.com/locate/envsci

# Greenhouse gas emissions from nitrogen fertilizer use in China

*Fredrich Kahrl*[a,b,*], *Yunju Li*[a], *Yufang Su*[c], *Timm Tennigkeit*[a,d], *Andreas Wilkes*[a,c], *Jianchu Xu*[a,c]

[a] *Centre for Mountain Ecosystem Studies, Kunming Institute of Botany, 132 Lanhei Road, Heilongtan, Kunming 650204, China*
[b] *Energy and Resources Group, University of California, 310 Barrows Hall, Berkeley, CA 94720, USA*
[c] *World Agroforestry Centre, East Asia Office, 132 Lanhei Road, Heilongtan, Kunming 650204, China*
[d] *Unique Forestry Consultants, Schnewlinstraße 10, 79098 Freiburg, Germany*

## ARTICLE INFO

*Keywords:*
Nitrogen fertilizer
Ammonia
Urea
Energy
Greenhouse gas emissions
China

## ABSTRACT

The use of synthetic nitrogen (N) fertilizers is an important driver of energy use and greenhouse gas (GHG) emissions in China. This paper develops a GHG emission factor for synthetic N fertilizer application in China. Using this emission factor, we estimate the scale of GHG emissions from synthetic nitrogen fertilizer use in Chinese agriculture and explore the potential for GHG emission reductions from efficiency improvements in N fertilizer production and use. The paper concludes with a discussion on costs and financing for a large-scale fertilizer efficiency improvement program in China, and how a GHG mitigation framework might contribute to program design.

## 1. Introduction

Synthetic nitrogen (N) fertilizers have played an important role in maintaining China's food security over the past three decades. In contrast to its low levels of synthetic N fertilizer production and use in the early 1970s, China is now the world's largest producer and consumer of N fertilizers. In the 1990s, the scientific community began to raise concerns over the potential overuse and environmental impacts of N fertilizer application in China, and since then a growing body of research has identified the need to improve N fertilizer use efficiencies.

While many of these concerns have centered around N fertilizers as a non-point source of water-borne pollution, application of N fertilizers in China is also a major driver of energy use and greenhouse gas (GHG) emissions. In addition to reducing water-borne pollution and other ecological impacts associated with anthropogenic reactive nitrogen, improving N fertilizer use efficiency could free up scarce energy resources, reduce GHG emissions, and contribute to poverty reduction goals by reducing input costs to farmers.

This paper estimates: a GHG emission factor for synthetic N fertilizer application; the scale of energy use and GHG emissions embodied in N fertilizer application; and GHG emission reductions from improvements in N fertilizer production and use efficiency in China. The paper concludes with thoughts on the costs and financing of a fertilizer efficiency program, and how a GHG mitigation framework might contribute to program design and funding.

Because our focus here is on chemical rather than organic N fertilizers, we use the term 'N fertilizer' to refer exclusively to synthetic N fertilizers in the text below. Additionally, our emphasis here is on N fertilizer use in agriculture, and the use

---

[*] *Corresponding author at*: Energy and Resources Group, University of California, 310 Barrows Hall, Berkeley, CA 94720, USA. Tel.: +1 510 642 1640.
E-mail address: fkahrl@berkeley.edu (F. Kahrl).

1462-9011/$ – see front matter © 2010 Elsevier Ltd. All rights reserved.
doi:10.1016/j.envsci.2010.07.006

of the phrase 'N fertilizer application' refers to application on cropland.

## 2. Use and overuse of nitrogen fertilizers in China

China is the world's largest producer and consumer of synthetic N fertilizers, accounting for an estimated 31% of world consumption in 2005 (FAOSTAT). Increasing the use of chemical fertilizers was a key part of the Chinese government's efforts to expand food production and ensure an adequate food supply, beginning in the early 1970s (Naughton, 2007). From 1970 to 2008, chemical fertilizer use increased from 2.4 to 60.1 Mt yr$^{-1}$ (total nutrients), a 25-fold increase (NBS, various years).[1] In the 1990s concerns began to emerge over the overuse and environmental impacts of N fertilizers in China. Since that time, a substantial literature has emerged on the ecological implications, and, to a lesser extent, behavioral drivers of N fertilizer use (Zhang et al., 1996; Yong and Zhang, 1999; Xing and Zhu, 2000; Zhu and Chen, 2002; Ju et al., 2004, 2009; Cui et al., 2006; He et al., 2007; Lin et al., 2007; Wang et al., 2007; Huang et al., 2008; Han and Zhao, 2009).

A key argument implicit across much of this literature is that, at an aggregate level, the marginal productivity of N fertilizer use in China is declining. Zhu and Chen (2002) argue that, although increased use of N fertilizers contributed to the substantial increase in China's agricultural output from the 1970s to the 1990s, the marginal contribution of N fertilizer to food production (narrowly defined)[2] has declined at an increasing rate since the 1950s. Ju et al. (2009) report that N fertilizer application increased by 271% from 1977 to 2005, while grain yields increased by only 98% and total grain output increased by only 71%.

Although these descriptions capture what is likely an aggregate trend of declining marginal productivity of N fertilizer application in China, it is important to note that neither includes fertilizer use for cash crops, and thus both may be neglecting the influence of a significant structural shift from grain to cash crops that occurred in China beginning in the 1990s (Fig. 1). As grain crops typically use less fertilizer per area vis-à-vis cash crops (Ju et al., 2004; Zhang et al., 2007),[3] shifts in cropping patterns may explain some of the apparent declining marginal productivity of fertilizer inputs vis-à-vis staple crops. Zhang et al. (2007) report that cash crops accounted for 50% of fertilizer use in China in 2005.

Nevertheless, there is a growing body of field-based evidence to suggest that N fertilizer application for grain crops in China exceeds optimal levels, that the marginal product of N fertilizer for grain crops is low, and that N fertilizer use could be reduced without adversely affecting grain yield. Table 1 catalogues a number of fertilizer use estimates across different regions of China. Although optimal N fertilizer use levels are site specific, these estimates compare against a range of around 150–200 kgN ha$^{-1}$ considered optimal for grain crops in China (Zhu and Chen, 2002; Ju et al., 2004, 2009). While the majority of research on N inputs in China has focused on grain crops, N use efficiency for vegetables and fruit may be similarly low.

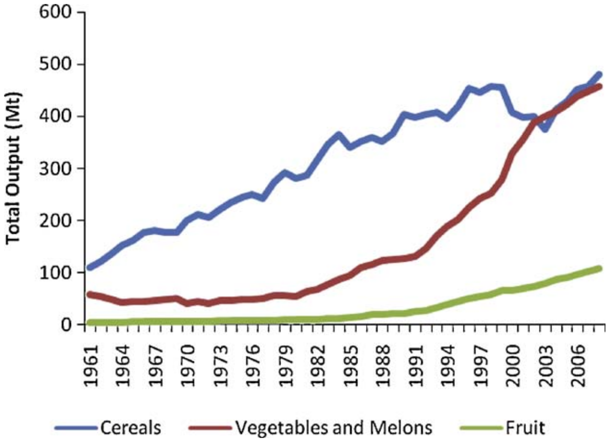

**Fig. 1 – Cereals, vegetables and melons, and fruit production in China, 1961–2008.Source: Data are from FAOSTAT.**

A growing number of studies confirm the potential for reducing N fertilizer application rates without reducing yield. In a 16-village experiment in Guangdong, Hunan, Hubei, and Jiangsu Provinces, Huang et al. (2008) report a 23% reduction in total fertilizer use as part of a training project to encourage farmers to use less fertilizer. Based on field trials in Jiangsu Province, Ju et al. (2009) estimate that total fertilizer use could be reduced by 30–60% without compromising yield. While these studies suggest the potential for significant improvements in N fertilizer use efficiency in China, how to achieve these improvements on a large scale remains an open question.

## 3. Energy use and GHG emissions from N fertilizer application in China

Rising N fertilizer use in China has contributed to a number of environmental problems (Zhang et al., 1996; Yong and Zhang, 1999; Domagalski et al., 2007; Guo et al., 2010), including an increase in GHG emissions. The use of N fertilizer induces process-based and combustion $CO_2$ emissions from the production of ammonia, combustion $CO_2$ emissions from the synthesis of N fertilizers from ammonia, and $N_2O$ emissions from the denitrification of N inputs.

Ammonia and fertilizer production in China are more energy and $CO_2$ intensive than the global average. While natural gas is the primary feedstock and source of process energy used for ammonia and N fertilizer production in most of the world, in China anthracite coal is the primary feedstock for ammonia synthesis and coal and electricity provide the bulk of process energy used in both ammonia and N fertilizer production. Additionally, for historical reasons a large number of China's ammonia-fertilizer producers are small and

---

[1] NBS data are from China Data Online.

[2] This definition only includes crops reported in the *China Statistical Yearbook* series: cereals, pulses, potato, and sweet potato.

[3] See also FAO FertiStat Fertilizer Use Statistics, online at: http://www.fao.org/ag/agl/fertistat/index_en.htm.

**Table 1 – Estimates of N fertilizer use in different cropping systems and regions of China.**

| Region | Crop(s) | Fertilizer use | Source |
|---|---|---|---|
| Jiangsu Province | Paddy rice | 300–350 kgN ha$^{-1}$ | Lin et al. (2007) |
| Beijing Municipality | Winter wheat | 309 kgN ha$^{-1}$ yr$^{-1}$ | Zhao (1997), c.f. Zhao et al. (2006) |
| | Summer maize | 256 kgN ha$^{-1}$ yr$^{-1}$ | |
| Henan Province | Multiple crops | 587 kgN ha$^{-1}$ yr$^{-1}$ | Gao et al. (1999), c.f. Zhao et al. (2006) |
| Shandong Province | Multiple crops | 652 kgN ha$^{-1}$ yr$^{-1}$ | |
| Shandong Province | Winter wheat | 369 kgN ha$^{-1}$ yr$^{-1}$ | Cui et al., 2006 |
| Yunnan Province | Summer maize | 360 kgN ha$^{-1}$ yr$^{-1}$ | Authors, 2008 unpublished surveys ($n = 458$) |

medium sized (Wong, 1986; Li, 2001; Zhang et al., 2009), and tend to be less energy efficient than larger facilities (Cao et al., 2008).

In this paper we develop a GHG emission factor for applied nitrogen (in tCO$_2$e tN$^{-1}$) in Chinese agriculture using: China-specific estimates of energy use in ammonia synthesis; China-specific estimates of energy use in the synthesis of China's two main N fertilizers – urea and ammonium bicarbonate (ABC) – from ammonia; and more generic N$_2$O emission factors using China-specific coefficients where available. A detailed accounting of the data and assumptions used in calculating the total GHG emission factor is provided in the Supplementary Online Material. From estimates of total specific energy use (GJ tN$^{-1}$) and the GHG emission factor for applied nitrogen, we use data on N fertilizer application in Chinese agriculture to calculate total energy use and GHG emissions in 2005. The estimates here are not lifecycle GHG emissions, as consensus estimates of GHG emissions embodied in upstream (e.g., coal mining) and downstream (e.g., transport) activities are not available for China. For context, Gellings and Parmenter (2004) report that packing, transport, and application can account for around 10% of the energy required to produce, distribute, and apply N fertilizer.

Actual use of N fertilizers in China is difficult to assess with a high level of accuracy. FAO estimates that China's total N fertilizer consumption was 30.2 MtN in 2005 (FAOSTAT). The IFA estimates that N fertilizer consumption in China was 29.7 MtN in 2005 (IFA website). Official statistics from the China Statistical Yearbook report that total application of N-based and compound fertilizers was 22.3 MtN and 13.0 Mt total nutrients, respectively, in 2005 (NBS, 2006), but the composition of compound fertilizers is not published as part of these statistics. At an average of 40% elemental N in compound fertilizer nutrients,[4] total N application in China would have been 27.5 MtN in 2005. The difference in these values, though significant, is comparatively small, and we use the FAO's 30.2 MtN estimate in the remainder of this paper. Because our interest is in N fertilizer used for crop production, we draw from an estimate from Zhang et al. (2007) to scale down total N fertilizer consumption by the percentage used in agriculture (~90%), which gives a final N fertilizer use on cropland in 2005 of 27 MtN.

Using data and assumptions described in detail in Supplementary Material, we calculate embodied energy use per N applied for ammonia production (77 GJ tN$^{-1}$) and fertilizer synthesis (30 GJ tN$^{-1}$). Multiplying these values by total N fertilizer application (27 MtN), we estimate that N fertilizer application in China induced primary energy use of 2.9 EJ, or 4.4% of China's total primary energy use of 65.6 EJ in 2005 (NBS, 2009). Household energy use accounted for only 6.9 EJ (10%) of China's total energy use in 2005 (NBS, 2009), which implies that the energy embodied in fertilizer use is roughly 40% as large as total household energy use.

Multiplying total N fertilizer application in agriculture (27 MtN) by our estimated GHG emission factor for applied N (15–31 tCO$_2$e tN$^{-1}$), we estimate that the application of N fertilizers in China led to emissions of 400–840 MtCO$_2$e in 2005 (Table 2), equivalent to 8–16% of China's energy-related CO$_2$ emissions (5101 MtCO$_2$, IEA [2007]) in that year. Although total GHG emissions and energy-related CO$_2$ emissions are not strictly comparable, China has not conducted a GHG emissions inventory since 2000 (1994 GHG emissions) and the IEA estimate provides a useful reference point. The significant range in our estimated GHG emission factor for applied N is driven by uncertainty in N$_2$O emission factors, emphasizing the need for further research to better understand direct and indirect N$_2$O emissions.

## 4. Reducing GHG emissions from N fertilizer application in China

A number of unknowns make baseline demand for N fertilizers in China difficult to forecast. Population growth and continued changes in the composition of diets in China will induce higher demand for fertilizers, while changes in relative factor prices associated with rural socioeconomic restructuring (e.g., urbanization) will likely lead to changes in fertilizer use practices and potentially a decline in baseline fertilizer use. A detailed forecast of N fertilizer use in China is

**Table 2 – Estimated GHG emissions from N fertilizer use in 2005 (in MtCO$_2$e).**

| | N$_2$O range | |
|---|---|---|
| | Default N$_2$O | High N$_2$O |
| Embodied ammonia | 180 | 180 |
| Fertilizer manufacture | 70 | 70 |
| N$_2$O emissions | 150 | 590 |
| Total | 400 | 840 |

See Supplementary Material for a more detailed description of assumptions behind these estimates.

---

[4] For instance, in a 15-15-15 NPK fertilizer N would constitute 44% of total nutrients. By contrast, Lu et al. (2008) assume that N is 30% of total NPK nutrients.

beyond the scope of this paper, and we use a more heuristic approach here. Growth in N fertilizer consumption (total N nutrients in pure N and compound fertilizers) in China has slowed dramatically since the 1980s, to around 2% $yr^{-1}$ from 2000–2008 (NBS, 2008). Zhang et al. (2009) cite a Ministry of Agriculture forecast of 1.6% $yr^{-1}$ growth in N fertilizer demand in China between 2010 and 2030, which is somewhat higher than FAO's (2000) forecast of 1% $yr^{-1}$ growth in total fertilizer use in the East Asia region to 2015 and 0.8% $yr^{-1}$ to 2030. At a conservative 1.5% $yr^{-1}$ average growth over 2005–2020 China's demand for N fertilizer in agriculture would reach 34 MtN by 2020.

Based on He et al. (2007), Huang et al. (2008), Ju et al. (2009), and Wen (2010), we assume that achieving a 20–30% reduction in baseline N fertilizer use through improvements in use efficiency could be feasible over the next decade. A 20–30% reduction vis-à-vis a baseline of 34 MtN would lead to a decrease of 7–10 MtN in N fertilizer application in China by 2020 (50–80 kgN $ha^{-1}$ based on China's cultivated land in 2005 [NBS, 2006]). As long as N fertilizer demand growth is below about 1.5% $yr^{-1}$, a 20–30% reduction in N fertilizer use levels would mean a decline in absolute levels of N fertilizer use over 2005 levels by 2020, requiring a major readjustment process for China's N fertilizer industry given that it was already overcapacity in early 2010.

At our estimated 2005 GHG emission factor for applied N (15–31 $tCO_2e\ tN^{-1}$), a 7–10 MtN reduction in N fertilizer use by 2020 would lead to GHG emission reductions of 100–310 $MtCO_2e\ yr^{-1}$. These reductions would be equivalent to a 2–7% reduction in the IEA's (2007, 2009) Reference Case estimate of the growth in China's energy-related $CO_2$ emissions from 2005 to 2020 (4482 $MtCO_2$). As with end use efficiency more generally, reductions in demand do not lead to linear reductions in supply-side GHG emissions, and it is possible that surplus N fertilizer production resulting from offset demand in China would be exported abroad. A fuller treatment of this issue would require an analysis of global fertilizer markets and a more complete understanding of supply elasticities in China's ammonia and fertilizer industries. While an important consideration, such a treatment is beyond the scope of this paper.

Significant reductions in $CO_2$ intensity are possible in China's ammonia and fertilizer industries through equipment efficiency improvements, fuel switching, industry restructuring, and, more passively, reductions in the $CO_2$ intensity of electricity generation (Cao et al., 2008; Zhou et al., 2010). China's National Development and Reform Commission (NDRC) has set a target for energy use in large ammonia plants to fall from 1210 ktce $tNH_3^{-1}$ (35.5 GJ $tNH_3^{-1}$) in 2005 to 1000 ktce $tNH_3^{-1}$ (29.3 GJ $tNH_3^{-1}$) in 2020, an improvement of 17% (NDRC, 2004). However, given its large number of less efficient, small- and medium-sized ammonia plants, average specific energy use in ammonia plants in China was 59.4 GJ $tNH_3^{-1}$ in 2005 (see Supplementary Material), and larger improvements in efficiency are likely possible. For example, reaching the International Fertilizer Association's (IFA's) estimated 2008 global average of 36.6 GJ $tNH_3^{-1}$ (IFA, 2009) would require a 38% improvement in aggregate efficiency.

Energy use in fertilizer synthesis in China is also significantly higher than in OECD countries. Urea manufacturing in the U.S. and EU, for instance, require an estimated 2.5–2.8 GJ $t^{-1}$ (USDOE, 2000; Worrell et al., 2000) and 3.2–4.6 GJ $t^{-1}$ (Gerlagh and van Dril, 1999) of primary energy, respectively. For China we estimate that urea manufacturing requires, on average and across fuels, 12.2 GJ $t^{-1}$ (see Supplementary Material). Although structure of technologies in China's fertilizer industry is, to some extent, constrained by natural resources and history, comparisons with OECD countries suggest that major gains in efficiency are possible through either industry restructuring (e.g., forcing small plants out of business) or facility upgrades (e.g., retrofitting new technologies). Fertilizer is considered to be a ''high energy consuming'' industry in China, and the need to improve the energy efficiency of N fertilizer production is increasingly recognized.

Possibilities for fuel switching in ammonia and fertilizer manufacturing in China are less clear. A shift to natural gas as both a feedstock and energy source would reduce the energy and $CO_2$ intensity of N fertilizer production, but may not be compatible with China's natural resource endowment. China has relatively limited natural gas reserves (1% of the world's proven reserves in 2007) but has an abundant supply of coal (14% of total proven coal reserves) (BP, 2009). Whether scarce natural gas resources have a higher social and environmental value in ammonia and fertilizer production or in other uses is ultimately a question to be determined by policy. In its 2007 Natural Gas Use Policy, the NDRC listed ammonia production in its ''restricted use'' category (NDRC, 2007). Without fuel switching from coal to natural gas, with current technologies there are limits to efficiency improvements in ammonia and fertilizer manufacturing (Zhou et al., 2010).

As an anchor point, drawing on Cao et al. (2008) we assume that 25% improvements in aggregate energy efficiency, with no changes in the composition of fuel use, would be feasible in both ammonia and fertilizer manufacturing by 2020. Based on the IEA's (2007) Alternative Policy Scenario, we assume that a 25% reduction in the $CO_2$ intensity of electricity generation over 2005 levels would be feasible by 2020. With these improvements, the combined emission factor for ammonia and fertilizer $CO_2$ emissions embodied in N fertilizer use could be reduced from 9.3 $tCO_2\ tN^{-1}$ in 2005 to 6.5 $tCO_2\ tN^{-1}$ (i.e., by 30%) in 2020 (Table 3), a level more on par with that in OECD countries (Lal, 2004).

In tandem, improvements in N fertilizer production (as detailed in Table 3) and use efficiency (i.e., a 20–30% reduction in N use) would lead to GHG emissions reductions of 180–380 $MtCO_2e$, vis-à-vis a baseline based on 1.5% $yr^{-1}$ N fertilizer demand growth, by 2020 (Table 4). This range of emission reductions is equivalent to 4–8% of the IEA's afore-mentioned forecast of energy-related $CO_2$ emissions growth in China between 2005 and 2020 (4 482 $MtCO_2$). At these levels, increases in the efficiency of N fertilizer production and use could be an important mitigation strategy in China.

## 5. Fertilizer efficiency program cost and financing

Much of the discussion on improving the efficiency of N fertilizer use in China has focused on the importance of removing fertilizer subsidies. While price reforms are impor-

**Table 3 – 2005 aggregate energy and emissions intensities, efficiency/intensity improvements, and implied energy and emissions intensities in 2000.**

| | 2005 | % efficiency or intensity improvement | Implied 2020 |
|---|---|---|---|
| Specific energy use in ammonia synthesis | 59.4 GJ $tNH_3^{-1}$ | 25% | 44.5 GJ $tNH_3^{-1}$ |
| Specific energy use in fertilizer manufacture | 29.5 GJ $tN^{-1}$ | 25% | 22.1 GJ $tN^{-1}$ |
| $CO_2$ intensity of net electricity consumption | 1.04 $kgCO_2$ $kWh^{-1}$ | 25% | 0.78 $kgCO_2$ $kWh^{-1}$ |
| Total emission factor for embodied ammonia and fertilizer energy use | 9.3 $tCO_2$ $tN^{-1}$ | 30% | 6.5 $tCO_2$ $tN^{-1}$ |

See Supplementary Material for a more detailed description of assumptions behind these estimates.

tant, we argue that a program to improve N fertilizer use efficiency in China on a large scale would require non-trivial investments in agricultural extension and physical infrastructure (e.g., irrigation infrastructure). The cost implications of these two strategies are different. If reducing fertilizer use is as simple as removing subsidies and raising awareness, the direct costs of a large-scale fertilizer efficiency program could be small. If, however, such a program requires significant investment, then program design, costs, and financing become more important considerations.

We begin with an assumption that, given their constraints (e.g., labor endowment, amount and timing of water availability) and risk preferences, farmers are currently using optimal levels of N fertilizer. In other words, farmers have experimented with different levels of N fertilizer use and current levels of use provide the desired yield effects at an acceptable cost. In this case, the cost of reducing fertilizer would be the marginal value of applied fertilizer, which is equal to its unit cost per area.

Using estimates for cultivated land (130 Mha [NBS, 2005]) and N fertilizer used in agriculture (27 MtN), average N fertilizer use in China was 210 kgN $ha^{-1}$ $yr^{-1}$ in 2005. Reducing N fertilizer use by 20% would require a roughly 40 kgN $ha^{-1}$ $yr^{-1}$ decrease in average N use levels. Urea costs in China in 2009 were around 2 yuan $kg^{-1}$, or equivalently around 4 yuan kg $N^{-1}$. Assuming, for the sake of illustration, that per N urea costs are representative of N fertilizer costs, at 4 yuan kg $N^{-1}$ the value of this reduction to farmers would be 160 yuan $ha^{-1}$ $yr^{-1}$, or US$ 40 $tCO_2e^{-1}$ using our default $N_2O$ GHG emission factor (15 $tCO_2e$ $tN^{-1}$). If the cost of a fertilizer efficiency program is the cost of replacing the value of N fertilizer use, total program costs would be around 20 billion yuan $yr^{-1}$ (160 yuan $ha^{-1}$ $yr^{-1}$ multiplied by 130 Mha), or around US$ 3 billion $yr^{-1}$.

**Table 4 – Reductions in 2020 GHG emissions with a 20–30% reduction in N fertilizer use and improvements in fertilizer production efficiency (in $MtCO_2e$).**

| | Fertilizer use reduction | | Production efficiency | | Total | |
|---|---|---|---|---|---|---|
| Reduction % | 20 | 30 | 20 | 30 | 20 | 30 |
| $N_2O$ low | 100 | 150 | 80 | 70 | 180 | 220 |
| $N_2O$ high | 210 | 310 | 80 | 70 | 290 | 380 |

See the Supplementary Material for a more detailed description of assumptions behind these estimates.

An alternative way of approaching this problem would be to assume that, while fertilizer use may be optimal given constraints and risk preferences, operating conditions are far from optimal. Investments to improve operating conditions, for instance through investments in agricultural extension services or physical infrastructure, could ease constraints, lower risk, and improve fertilizer use efficiencies. Fertilizer efficiency program costs, then, would be the cost of these investments rather than the value of fertilizer reductions.

GHG mitigation provides a useful framework for thinking about investment levels. At a cost of 68 yuan $tCO_2^{-1}$ (US$ 10 $tCO_2^{-1}$), a fertilizer efficiency program that achieves 20% reductions in annual N fertilizer use from 2005–2020 would generate revenues (e.g., through an offset program) of about 40 yuan $ha^{-1}$ $yr^{-1}$, or total revenues of around 80 billion yuan (US$ 12 billion) over 15 yrs. Allocated over China's 2859 counties (NBS, 2008), 80 billion yuan (3–4 billion yuan $yr^{-1}$) would mean payments of about 30 million yuan (2 million -yuan $yr^{-1}$) per county. If this level of investment would be sufficient to provide incentives for improvements in fertilizer use efficiency, total program costs would be much lower than the value of N fertilizer reductions and would save farmers up to 20 billion yuan $yr^{-1}$ in fertilizer expenditures. Depending on the trade-offs between industry impacts and higher rural consumption, a fertilizer efficiency program could be net positive, with economy-wide benefits exceeding costs. Additionally, if US$ 10 $tCO_2^{-1}$ is cost-effective relative to other GHG emission reduction options in China, a domestic offset program could provide the necessary funding.

The above discussion highlights four important points:

(1) In designing a fertilizer efficiency program it is important to consider the marginal costs and benefits of current levels of N fertilizer use. We argue that simply removing subsidies and conducting a broad information campaign is unlikely to address root drivers of inefficiency in N fertilizer use.

(2) Instead, and much like energy efficiency (Blumstein et al., 1980), it is more likely that there are a number of barriers to higher N fertilizer efficiency levels in China, and that public sector investments will be required to overcome these barriers.

(3) Even without considering "external" costs (e.g., eutrophication, nitrate pollution, and climate change), the cost of a fertilizer efficiency program could be much smaller than the total net savings to farmers if well designed.

(4) If the mitigation costs of N fertilizer reductions are lower than mitigation costs elsewhere in the economy (e.g., in steel production), a domestic GHG offset scheme could provide the funds needed for implementing a fertilizer efficiency program.

## 6. Conclusions

There is a growing body of evidence that suggests that the efficiency of N fertilizer use in China is low, and that through efficiency improvements the use of N fertilizer could be significantly reduced without affecting yields. Because N fertilizer use in China is a major source of embodied $CO_2$ and $N_2O$ emissions, a large-scale program to improve N fertilizer efficiency could be an important part of a larger strategy to make the Chinese economy more resource and emissions efficient over the next decade.

This paper examines the potential for reducing GHG emissions through improvements in N fertilizer use efficiency. Using China-specific energy use estimates for ammonia and fertilizer synthesis and more generic approach for $N_2O$, we calculate an emission factor range of 15–31 $tCO_2e\ tN^{-1}$, with the significant range driven by uncertainty in $N_2O$ emissions.

Using the 15–31 $tCO_2e\ tN^{-1}$ emission factor, we estimate that N fertilizer application on cropland led to GHG emissions of 400–840 $MtCO_2e$, equivalent to 8–16% of China's 2005 energy-related $CO_2$ emissions. Using assumptions about N fertilizer demand growth to 2020, we estimate that a 20-30% reduction vis-à-vis 2020 baseline N fertilizer demand growth would lead to GHG emission reductions of 100–310 $MtCO_2e$ in 2020, equivalent to 2–7% of the IEA's forecasted growth in energy-related $CO_2$ emissions in China between 2005 and 2020.

Due to China's natural resource endowment and for historical reasons, ammonia and fertilizer production are much more energy and $CO_2$ intensive in China than in OECD countries. If 20–30% reductions in N fertilizer use are combined with 25% energy efficiency improvements in ammonia and fertilizer production and a 25% decline in the $CO_2$ intensity of electricity generation, total GHG emission reductions would reach 180–380 $MtCO_2e$ by 2020, or 4–8% of the IEA's forecasted growth in energy-related $CO_2$ emissions in China between 2005 and 2020. To a greater extent than in OECD countries, improving efficiency in N fertilizer production and use could be an important mitigation strategy in China.

A fertilizer efficiency program must address the root drivers of inefficient N fertilizer use in China. There are likely a number of barriers to higher use efficiencies for N fertilizers, and removing these barriers will require government policy intervention. Even if substantial investments from the public sector are required, the net economy-wide benefits of a fertilizer efficiency program (stimulus from savings to farmers minus direct and indirect program costs) may still be positive, even without including environmental externalities. Including one such externality – the cost of climate change mitigation – could provide an additional avenue for raising investment funds for a fertilizer efficiency program. If the costs of reducing N fertilizer use are lower than mitigation costs elsewhere in the economy, a domestic offset program could be an important source of program funding.

## Acknowledgements

This research was conducted with support from the Re-Impact project ENV/2007/114431, funded by the European Union Aid Cooperation Office Programmemes on Environment in Developing Countries and Tropical Forests and other Forests in Developing Countries. The authors wish to thank Professor Wang Yanjia for informal comments on parts of this paper.

## Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at doi:10.1016/j.envsci.2010.07.006.

## REFERENCES

Blumstein, C., Krieg, B., Schipper, L., York, C., 1980. Overcoming social and institutional barriers to energy conservation. Energy 5, 355–371.

British Petroleum (BP), 2009. Statistical Review of World Energy 2009. BP, London.

Cao, L., Zhang, W.F., Gao, L., Wang, L., Ma, W.Q., Gao, X.Z., Zhang, F.S., 2008. 中国合成氨生产能源消耗状况及其节能潜力.(Energy consumption in ammonia production in China and energy-saving potential). 化肥工业 (Fertilizer Industry).35 (2), 20–24.

Cui, Z., Chen, X., Li, J., Jiu, X., Shi, L., Zhang, F., 2006. Effect of N fertilization on grain yield of winter wheat and apparent N losses. Pedosphere 16 (6), 806–812.

Domagalski, J., Lin, C., Luo, Y., Kang, J., Wang, S., Brown, L.R., Munn, M.D., 2007. Eutrophication study at the Panjiakou-Daheiting Reservoir system, northern Hebei Province, People's Republic of China: chlorophyll-a model and sources of phosphorus and nitrogen. Agricultural Water Management 94, 43–53.

Gellings, C.W., Parmenter, K.E., 2004. Energy efficiency in fertilizer production and use. In: Gellings, C.W., Blok, K. (Eds.), Efficient Use and Conservation of Energy. Encyclopedia of life support systems (EOLSS) Publishers, Oxford, UK.

Gerlagh, T., van Dril, A.W.N., 1999. The Fertilizer Industry and its Energy Use: Prospects for the Dutch Energy Intensive Industry. Energy Research Center of the Netherlands Report ECN-C-99-045.

Guo, J.H., Liu, X.J., Zhang, Y., Shen, J.L., Han, W.X., Zhang, W.F., Christie, P., Goulding, K.W.T., Vitousek, P.M., Zhang, F.S., 2010. Significant acidification in Major Chinese Croplands. Science 327, 1008–1010.

Gao, W., Huang, J., Wu, D., Li, X., 1999. Investigation on nitrate pollution in ground water at intensive agricultural region in Huanghe-Huaihe-Haihe Plain. Eco-Agriculture Research 7, 41–43 (Chinese).

Han, H., Zhao, L., 2009. Farmers' character and behavior of fertilizer application—Evidence from a Survey of Xinxiang County, Henan Province, China. Agricultural Sciences in China 8 (10), 1238–1245.

He, H., Zhang, L., Li, Q., 2007. How to Reduce Non-point Pollution from Crop Production? The Case of Fertilization in China. Paper Presented at the 6th International Conference on the Chinese Economy, 18–19 October.

Huang, J., Hu, R., Cao, J., Rozelle, S., 2008. Training programs and in-the-field guidance to reduce China's overuse of fertilizer without hurting profitability. Journal of Soil and Water Conservation 63 (5), 165A–167A.

International Energy Agency (IEA), 2007. World Energy Outlook 2007. OECD/IEA, Paris.

International Energy Agency (IEA), 2009. World Energy Outlook 2009. OECD/IEA, Paris.

International Fertilizer Industry Association (IFA), 2009. Energy Efficiency and $CO_2$ Emissions in Ammonia Production: 2008–2009 Summary Report. IFA, Paris.

Ju, X., Liu, X., Zhang, F., Roelcke, M., 2004. Nitrogen fertilization. Soil nitrate accumulation, and policy recommendations in several agricultural regions of China. Ambio 33 (6), 300–305.

Ju, X., Xing, G., Chen, X., Zhang, S., Zhang, L., Liu, X., Zhen, C., Yin, B., Christie, P., Zhu, Z., Zhang, F., 2009. Reducing environmental risk by improving N management in intensive Chinese agricultural systems. Proceedings of the National Academy of Sciences 106 (9), 3041–3046.

Lal, R., 2004. Carbon emission from farm operations. Environment International 30 (7), 981–990.

Li, Z., 2001, Outlook for fertilizer production by medium and small nitrogen and phosphate plants in China. Presentation at the IFA Production and International Trade Conference, 13–14 September.

Lin, D., Fan, X., Hu, F., Zhao, H., Luo, J., 2007. Ammonia volatilization and nitrogen utilization efficiency in response to urea application in rice fields of the Taihu Lake Region, China. Pedosphere 17 (5), 639–645.

Naughton, B., 2007. The Chinese Economy: Transitions and Growth. The MIT Press, Cambridge.

National Bureau of Statistics, various years. China Statistical Yearbook. China Statistics Press, Beijing.

National Development and Reform Commission (NDRC), 2004. 节能中长期专项规划[[nl]]Medium- and Longer-term Energy Efficiency Plan. NDRC Policy Document.

National Development and Reform Commission (NDRC), 2007. 天然气利用政策[[nl]]Natural Gas Use Policy. NDRC Energy Document 2155 发改能源[[nl]](2007) 2155号.

United States Department of Energy (USDOE), 2000. Energy and Environmental Profile of the U.S. Chemical Industry. DOE, Washington, DC.

Wang, G., Zhang, Q.C., Witt, C., Buresh, R.J., 2007. Opportunities for yield increases and environmental benefits through site-specific nutrient management in rice systems of Zhejiang Province, China. Agricultural Systems 94, 801–806.

Wen, T., 2010. The True Cost of Fertilizer. (氮肥的真实成本).Greenpeace, Beijing.

Wong, C., 1986. Intermediate technology for development: small-scale chemical fertilizer plants in China. World Development 14 (10/11), 1329–1346.

Worrell, E., Phylipsen, D., Einstein, D., Martin, N., 2000. Energy use and energy intensity of the U.S. Chemical Industry. Lawrence Berkeley National Lab Report LBNL-44314.

Xing, G., Zhu, Z., 2000. An assessment of N loss from agricultural fields to the environment in China. Nutrient Cycling in Agroecosystems 57, 67–73.

Yong, L., Zhang, J., 1999. Agricultural diffuse pollution from fertilisers and pesticides in China. Water Science and Technology 39 (3), 25–32.

Zhang, W.L., Tian, Z.X., Zhang, N., Li, X.Q., 1996. Nitrate pollution of groundwater in northern China. Agriculture, Ecosystems and Environment 59, 223–231.

Zhang, W., Zhang, F., Ma, L., 2007. The Fertilizer Situation and Outlook in China. Presentation at the Sino-German International Research Training Group, Stuttgart, Germany, 13 November.

Zhang, F., Zhang, W., Ma, W., 2009. The Chemical Fertilizer Industry in China: A Review and its Outlook. International Fertilizer Association, Paris, France.

Zhao, J., 1997. The investigation and analysis of N application and yield in Beijing suburb. Beijing Agricultural Science 15, 36–38 (Chinese).

Zhao, R., Chen, X., Zhang, F., Zhang, H., Schroder, J., Römheld, V., 2006. Fertilization and Nitrogen Balance in a Wheat–Maize Rotation System in North China. Agronomy Journal 98, 938–945.

Zhou, W., Zhu, B., Li, Q., Ma, T., Hu, S., Griffy-Brown, C., 2010. $CO_2$ emissions and mitigation potential in China's ammonia industry. Energy Policy 38, 3701–3709.

Zhu, Z.L., Chen, D.L., 2002. Nitrogen fertilizer use in China–Contributions to food production impacts on the environment and best management strategies. Nutrient Cycling in Agroecosystems 63, 117–127.

**Fredrich Kahrl** is a Ph.D. candidate in the Energy and Resources Group at the University of California, Berkeley and a Senior Associate at the Center for Mountain Ecosystem Studies (CMES), a joint center of the Chinese Academy of Sciences and the World Agroforestry Centre (ICRAF). His research focuses on the economic, engineering, and environmental science dimensions of energy and agricultural systems.

**Yunju Li** is a Ph.D. candidate at the School of Resources and Environment, Nanjing Agricultural Univeriesty and a researcher with the Kunming Institute of Botany, Chinese Academy of Sciences. His research focuses on agricultural policy, and in particular the influence of land use and climate change on water resources.

**Yufang Su** is a program manager at the World Agroforestry Centre in China, where her research focuses on resource management, land use, climate adaptation, and rural energy. She holds an M.A. in sustainable development from Chiang Mai University.

**Dr. Timm Tennigkeit** is head of the Carbon Finance Section at UNIQUE Forestry Consultants, where he specializes in carbon projects in agriculture, forestry, and rangelands. He is also a Senior Associate at CMES.

**Dr. Andreas Wilkes** is deputy country representative for the World Agroforestry Centre in China. His research focuses on climate change mitigation in agriculture and other land uses in Asia, and institutional issues in small holder adaptation to climate change.

**Dr. Jianchu Xu** is head of the World Agroforestry Centre's East Asia Program, and is a scientist at the Kunming Institute of Botany, Chinese Academy of Science. He is former head of Water and Hazards at the International Centre for Integrated Mountain Development (ICIMOD) and former director of the Center for Biodiversity and Indigenous Knowledge (CBIK). His research interests cover natural resource management, land use, and watershed governance, with a focus on the Tibetan Plateau.