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Agricultural development in China

- environmental impacts, sustainability issues and policy implications assessed through China-UK projects under SAIN (UK-China Sustainable Agriculture Innovation Network), 2008 – 2017

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Agricultural development in China: environmental impacts, sustainability issues and policy implications assessed through China-UK projects under SAIN (UK-China Sustainable Agriculture Innovation Network), 2008 – 2017

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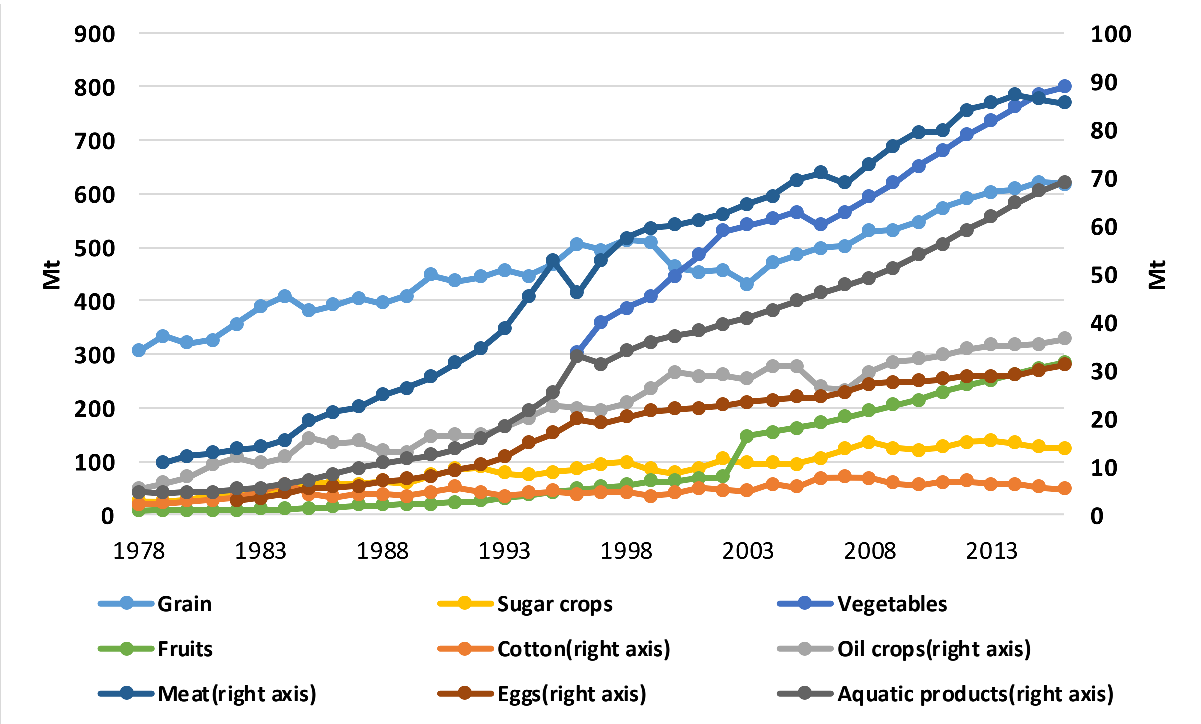
1. Introduction

Production of China's main food commodities has increased two to eight fold since the major policy reforms in 1978 (Figure 1). This has made China one of the most food-secure countries in the world. But this success has been achieved at the cost of substantial environmental damage with an economic cost estimated at least US $32–67 billion, equivalent to about 3–7% of China's agricultural GDP (Norse and Ju, 2015). These impacts, which operate at local, regional and global scales, have major implications for future sustainability and quality of life and require wide ranging policy responses rather than purely technical solutions. In this respect the recent situation in China has much in common with the environmental and sustainability issues that the UK and other EU countries faced some 30 or more years ago because of poorly managed agricultural intensification. The UK and EU addressed these issues by technological and policy changes that are, or could be, relevant to China's current situation. It was this potential for knowledge sharing that helped to shape the 2008 China/UK Memorandum of Understanding (MoU) on Cooperation in Agricultural Science Technology and the main mechanism for its implementation, namely the China-UK Sustainable Agriculture Innovation Network (SAIN), see Lu *et al* (2011). Consequently, the purpose of this article is to systematically review the findings and policy implications of the research that SAIN has supported since 2008 in the context of the above MOU.

Figure 1. Output of major agricultural products in China, 1978-2016 (Mt)



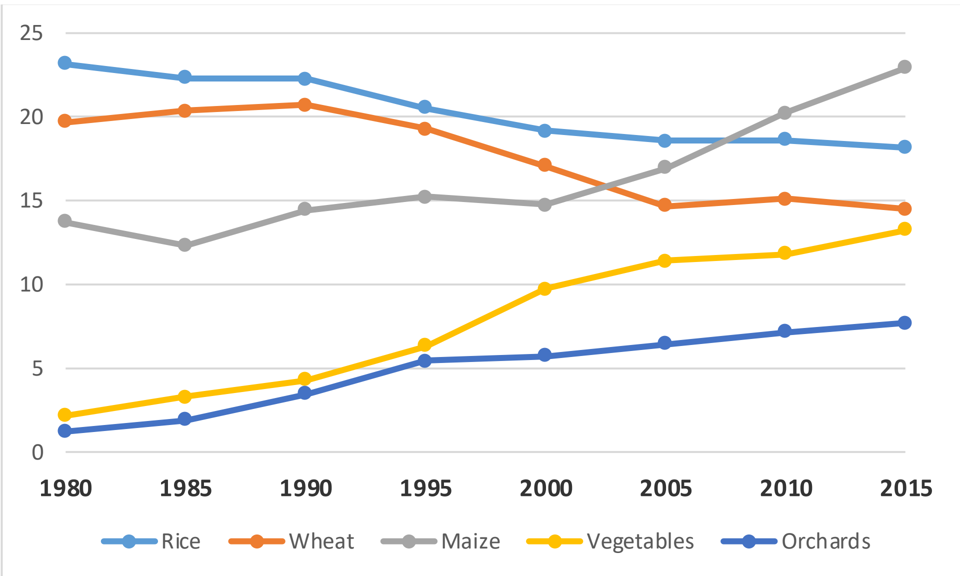
Many projects and publications over several decades have demonstrated a range of technical innovations and changes in management practice at farm level that can reduce the environmental impacts associated with increasing food production in China. However, adoption of these measures is almost always dependant on changes in policy or other actions by Government. In part this is because a large proportion of farms in China are extremely small, often less than 1 ha, with farmers deriving much of their income from off-farm activities and so not being influenced by normal economic factors that affect farming decisions. Also, because of the economic structures in China and subsidies that strongly influence agriculture, policy changes can have a particularly large influence on practice at farm scale. Infrastructure issues that are outside the farmers' control are also significant for improving the environmental impact of agriculture in China; these include access to certain crop varieties, access to specific fertilizer products, availability of irrigation, size of farms, contracting arrangements and transportation issues. In addition, farmer training and factors influencing delivery of information to farmers are highly important with evidence that traditional approaches often have very little effect on farmers' decision making. In many agricultural research projects these issues are overlooked, with emphasis being given to technical or academic issues. SAIN projects specifically seek to include the social, economic, practical or policy aspects, giving them a unique character and value. All projects under SAIN have led to one or more Policy Briefs, written in both English and Chinese, and communicated to senior staff in the policy arena through the SAIN Secretariats.

Although China is a huge country, only a small proportion of the land has soil and agro-climatic conditions suitable for crop and livestock production. A figure often quoted is that China has 21% of the world's population but only 7% of global land suitable for agriculture. Unsurprisingly, food security is an over-riding political priority for the Government of China (GOC) with a goal of maintaining high levels of national self-sufficiency in rice, wheat and pork. Furthermore, over the past 30 years there have been substantial population movements and changes in China's natural resource base and in the structure of agriculture and food production. Nonetheless, the GOC's food security and food self-sufficiency objectives have been met, and technological progress has been a major factor in this success.

The contribution of technological progress to growth in agriculture reached 54.5% in 2012 (OECD-FAO, 2013). Technologies included improved varieties (e.g. hybrid rice), crop protection, new farming practises such as polythene mulching film and polytunnels, and concentrated animal feeding operations (CAFOs). However, many of these "green revolution" technologies also require high fertilizer and pesticide inputs which have commonly been mismanaged and consequently there have been serious environmental impacts (Norse, 2005; Ju et al, 2009). Therefore, raising resource and input use efficiency is a challenge for science and technology development in China relevant to agriculture. Improved data and understanding on the spatial and temporal characteristics of shifts in food demand and the responses of agricultural technology (Agri-Tech) to these shifts are a key input to policy formulation and R&D priority setting. It is a challenge that China is well able to meet. China's capacity for Agri-Tech innovation is huge. For example, even in a narrow research field like remote sensing applications there are >20 well-funded specialist institutions or university departments with extensive research programmes. The Newton programme in China both feeds into this body of expertise and seeks to develop synergies whereby UK and Chinese expertise is shared.

The increased production of many products shown in Fig. 1 is partly a result of substantial increases in yields per unit area (e.g. for wheat and rice), but also from major changes in land use as shown in Fig. 2. Notable changes in cropping between 1980 and 2013 are an approximate 7-fold increase in the area used for vegetables, a 6-fold increase in the area used for fruit and an 85% increase in maize area at the expense of land devoted to rice and wheat. These changes in land use, and the massive increase in livestock production discussed later, all have major environmental impacts and profound implications for policy and national and local scale. SAIN's initial work focused on research gap analysis regarding these changes, and demonstrated the need for a more holistic life cycle approach to agri-environment analysis. For example, Policy Brief No 1 highlighted the large proportion of agricultural greenhouse gas emissions(GHGs) that came from N fertilizer production (17-20%) and livestock production and manure management (30-50%). Similarly Policy Brief No 2 demonstrated that the environmental impacts of China's agricultural intensification are greater and wider than previously thought. These findings helped shape the future direction of UK-China and particularly SAIN's research programme and the results described in the rest of this paper.

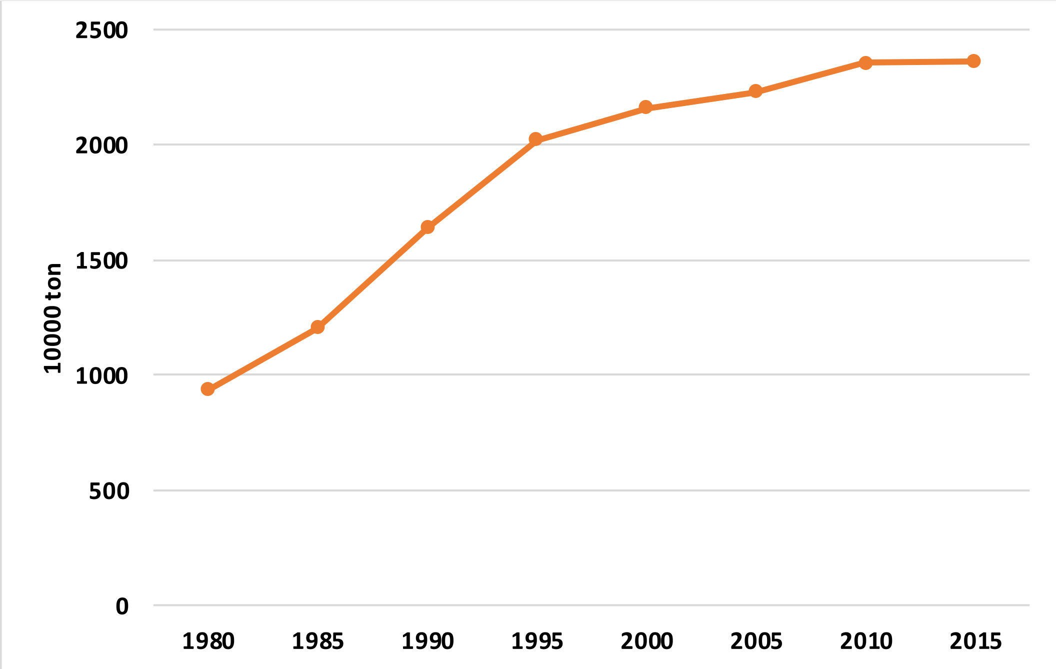
Figure 2. Change in crop sown area (Mha)



2. Nitrogen fertilizer applications to crops in China

Fig. 3 shows the increase in total use of N fertilizer in China since 1980. This increase has been driven by two main factors. First, the goal of increasing production of staple food crops driven by government policies and subsidy arrangements aimed at achieving a high degree food self-sufficiency (at least 95%) in key products including rice, wheat and pork. Second, the diversification of cropping over the last 20-30 years, especially the increased area devoted to vegetables and fruit, as shown in Fig. 2. With these high value horticultural crops it is frequently economic to use high rates of N (and other inputs), even though the N is generally used inefficiently and leads to large losses to water, atmosphere or both.

Figure 3. Nitrogen fertilizer use in China: changes since 1980



To meet national food security goals it was essential for fertilizer applications, including N, to increase compared to the very low usage pre-1980. However, during the 1980s, and especially after 1990, it became clear that N application rates were frequently well above those required for optimising crop production. In a seminal paper by scientists at China Agricultural University (CAU; Ju *et al*, 2009) the authors analysed data on N fertilizer application rates and yields of major grain crops (wheat, rice and maize) in two regions of great importance for food security in China – the North China Plain and the Yangtze River Basin. They concluded that "*more efficient use of N fertilizers can allow rates to be reduced by 30 to 60%. This would still maintain crop yields and N balance in rotations, while substantially reducing N losses to the environment*." Within the scope of SAIN, Norse *et al* (2011) summarised data from more than ten Provinces, again showing considerable over-use on about 30% of the arable land and commonly scope for decreases of 30-50% or more compared to current applications (Table 1).

In addition to the waste of resource and the environmental impacts, discussed below, over-application of N often leads to *decreased* crop yield as shown in Table 1. This is usually due to either lodging (crops over-supplied with nutrients becoming heavy and falling over, so becoming difficult to harvest) or increased susceptibility to diseases or pests.

Investigating the issue of N fertilizer over-use and misuse has been an important component of the SAIN programme but SAIN project have brought new perspectives to the issue and, in contrast to much previous research, have proposed practical and policy approaches to overcome the problem.

Table 1. Examples of N overuse by Province and crop (from Norse et al, 2011)

Province or region	Crop	Farmers' rate kg N/ha	Recommended rate [a] Kg N/ha^{-1}	% decrease compared to farmers' rate	Change in yield [b] compared to farmers' rate
Jiangsu	rice	300	200	50	+3 %
6 Provinces [c]	rice	195	133	47	+6%
North China Plain	wheat	325	128	150	+4%
North China Plain	maize	263	158	66	+5%
Shaanxi	wheat	249	125-225	~100	same
Shaanxi	maize	405	165-255	>60	>8%
Shandong	tomatoes	Up to 630	150-300	80-200	+10% n.s.

[a] Based on soil tests and field experiments.
[b] Positive value represents an increase in yield with the lower rate of N application
[c] Guangdong, Heilongjiang, Hubei, Hunan, Jiangsu, Zhejiang

2.1 Environmental, economic and health impacts of N fertilizer over-use

The main undesirable impacts of excessive N use, and the resulting loss of N to the environment, can be summarised as follows:

- Drinking water quality. Nitrate concentrations in drinking water often exceed limits deemed as safe by WHO.
- Surface water quality – nitrate (often with phosphate) causing algal blooms or growth of water weeds, causing a negative impact on fisheries. Some algal blooms produce toxins that are harmful to animals and humans.
- Increased emission of nitrous oxide (N_2O), a powerful greenhouse gas with approximately 300 times the global warming potential of carbon dioxide (CO_2).
- Increased emission of ammonia gas (NH_3). Much of this is deposited on soil, vegetation or water causing environmental or ecosystem damage through nutrient enrichment (eutrophication) and soil acidification. In the atmosphere, NH_3 combines with other components to form various small particles including $PM_{2.5}$ which contribute to smog and cause respiratory disease in humans. NH_3 also contributes to indirect emissions of N_2O

2.1.1 Nitrate residues in soil and movement of nitrate to natural waters

In a paper in a Special Issue of the journal *Agriculture Ecosystems & Environment*, sponsored by SAIN, Hofmeier *et al* (2015) quantify the extent of N fertilizer over-use in the rice-wheat double cropping system in the Taihu Region. At five experimental sites in farmers' fields, reducing N fertilizer applications by 23% for rice and 32% for wheat had little impact on total grain yields. These reductions, combined with changes in timing of applications, also led to increases in nitrogen use efficiency (NUE) as measured in several different ways. From these and from surveys of 43 farm households in the region the authors concluded that decreases of at around 20% were entirely possible for summer rice compared with the average rate of 270 kg N ha^{-1} typically used, and an even larger reduction for winter wheat compared to the current 220 kg N ha^{-1}.

In a SAIN project researchers from several UK centres (Rothamsted Research, UCL, and UEA) worked with colleagues at the North-West Agriculture & Forestry University, Yangling, Shaanxi Province, to study farmers' use of N fertilizer (Zhang *et al*, 2015). The study focussed on farmers in the Guanzhong Plain of Shaanxi Province, northwest China, a region less commonly associated with excessive N use. The project included household surveys in order to obtain information on economic and social factors influencing farmers' decision making, in addition to technical information from on-farm fertilizer experiments. A survey of 80-100 farmers growing a wheat-maize double cropping rotation showed that 77% were applying N at rates in excess of those recommended by the local advisory agencies (Zhang *et al*, 2015). In addition to showing the extent of N fertilizer over-use, this finding demonstrates the ineffectiveness of current advisory approaches. In trials in farmers' fields in the region it was found that N fertilizer applications to winter wheat and summer maize could be decreased by 70% and 20%, respectively, with no loss of yield and sometimes small increases (Zhang *et al*, 2015). Household economic assessments showed that even a 30% reduction in N use would increase household income by 2–9%; the poorest farmers benefit proportionately the most because fertilizer represents a larger percentage of their expenditure and they receive less income from off-farm sources (Norse 2012; Norse *et al* 2011), so policies and practices leading to more rational N use are clearly pro-poor. This project also led to development of a N balance sheet method for giving advice to farmers on N fertilizer management that was considered to be more understandable and empowering to farmers than the classical approach of recommendation sheets. Through collaboration with the local advisory staff, a simple in-field soil testing method for nitrate was developed as a possible addition to new advisory approaches.

There is a vast body of literature, from SAIN projects and elsewhere, demonstrating increasing concentrations of nitrate in both ground- and surface-waters in recent decades in China, and that this is, in large part, attributable to N from fertilizers with examples of nitrate accumulations in soil in the range of several hundred to >1000 kg N ha^{-1} (Gu *et al*, 2013; Li *et al*, 2007; Ju *et al*, 2009; Zhou *et al*, 2016; Zhu *et al*, 2006).

In addition to general N over-use, the conversion of large areas of land previously used for grain crops to production of vegetables and fruit has led to increased N applications with consequent nitrate accumulation in soil and movement to natural waters. For example, in a project that grew from earlier SAIN work, >200 sites in Shaanxi Province, northwest China, where kiwi orchards had replaced the wheat-maize rotation were surveyed (Lu *et al*, 2016). It was found that annual inputs of N (total from fertilizer, manure, irrigation water (which is generally overlooked) and atmospheric deposition) increased on average from 425 kg N ha^{-1} under the cereal rotation to 1201 kg N ha^{-1} after conversion to kiwi fruit (Lu *et al*, 2016) – i.e. an increase of almost 3-fold. Another project in the same region based on UK-China collaboration, but just pre-dating the establishment of SAIN, focussed on nitrate residues in soil under "sunlight greenhouses" – structures covered with plastic for warmth in winter but open and subject to rainfall in summer and used for vegetable production (Zhou *et al*, 2010). Large areas of land throughout China are being converted from cereal cropping to this form of horticulture, so the results have widespread applicability. In this survey of some 200 commercial greenhouses N fertilizer applications rates were 600-750 kg N ha^{-1} yr^{-1} and >500 kg N ha^{-1} had accumulated as nitrate in the soil to a depth of 1m. Similar data was found in a recent survey of literature from 7000 samples covering the whole of China (Zhou *et al* 2016). The average quantities of nitrate-N in the soil profile (0-4m) were 453 ± 39 kg N ha^{-1} under wheat, 749 ± 75 kg N ha^{-1} under maize, 1191 ± 89 kg N ha^{-1} under open-field vegetables, 1269 ± 114 kg N ha^{-1} under "sunlight greenhouses" and 2155 ± 330 in orchards. Large stocks of nitrate such as these accumulated in the soil profile are subject to movement to groundwater, the time-course of this being influenced by annual variations in rainfall and likely greater movement under the influence of more extreme rainfall events caused by climate change: this could be regarded as a nitrate pollution "time-bomb" in addition to current gradual downward movement and entry in ground-waters. It is also an extraordinary waste of a valuable resource.

2.1.2 Nitrous oxide emissions and implications for climate change

An inevitable consequence of N applications to soil, whether as inorganic fertilizers or organic manures, is some evolution of N_2O to the atmosphere. However, where excess N is applied, N_2O evolution is greater than necessary and can be non-linear, i.e. a small increase in excess N causing a proportionately greater evolution of N_2O (Ju *et al* 2011). Because N_2O is such a powerful greenhouse, approximately 300 times greater than CO_2, this is a very strong reason for avoiding excess N inputs. As with nitrate leaching, a change from grain crops to horticulture can lead to a large increase in N_2O emission. In one study involving leading SAIN participants in the Yangtze River Basin, Jiangsu Province, annual N_2O emission was seven times greater from intensive vegetable production than from the previous rice cropping (Min *et al*, 2012).

For calculating national inventories of greenhouse gas emissions, the IPCC introduced the concept of an "emission factor" (EF). In the case of N_2O, the EF for N fertilizer use using Tier 1 methodology is 1% (0.3-3.0% uncertainty) of N applied to soil, and 0.3% (0.0-0.6 uncertainty) for flooded rice fields (IPCC 2006). This is based on a large and variable global dataset, which makes it difficult to obtain accurate estimates that reflect regional/national soils, climate and N management. If more country specific EFs are produced, the national inventories can use these disaggregated factors in a Tier 2 assessment of emissions. This is especially desirable for the Chinese national inventory because the country represents one third of total global N fertilizer consumption and thus a major contributor to global N_2O emissions. In a major SAIN study involving the Chinese Academy of Sciences Institute of Soil Science and four UK participants (Shepherd *et al* 2015), a detailed analysis was made of all published N_2O emission data for Chinese agriculture; a summary of the results is given in Table 2. Several of the UK participants are leaders of the Defra-funded work to produce the UK N_2O inventory, thus this SAIN project built on the existing UK effort and the experience of examining a vast body of data from China benefitted the UK participants.

Table 2. Summary of disaggregated N_2O emission factors for N fertilizer use in China, from Shepherd *et al* (2015). The authors provided more detailed disaggregated values taking account of N fertilizer form, soil types (texture, pH), and additional sub-divisions of crops.

	EF, % of fertilizer N applied emitted as N_2O-N
IPCC global value for Tier 1 estimations	1.0 (0.3-3.0)
Disaggregated Tier 2 values for China:	
Lowland horticulture	0.74-1.26
Non-flooded upland crops (inc. wheat, maize, soya, oilseed rape)	0.4-1.54
Flooded rice, temporarily drained	0.29-0.66
Flooded rice, undrained	0.15-0.37

In many situations EFs in Chinese agriculture are somewhat lower than global mean of 1% for non-flooded crops. This detailed analysis will now facilitate greater accuracy in estimating N_2O emissions from agricultural soils in China – a matter of global interest in view of the magnitude of China's N use and its large area of agricultural land. Studies using a novel method in which N_2O emissions were measured continuously showed that in the conditions of North China Plain (soils low in organic matter, sub-humid climate), N_2O was mainly derived from nitrification of ammonium in soil rather than through denitrification of nitrate which is thought to be a more significant source in many places globally (Ju *et al* 2011). This is an important finding for increased scientific understanding of a key process but also has the practical implication that greater use of nitrate-based N fertilizers, at least in some situations in China, could be one approach to decreasing total N_2O emissions. This was highlighted in SAIN Policy Brief No. 5.

2.1.3 Ammonia volatilisation

Volatilisation of ammonia gas (NH_3) to the atmosphere is a major pathway of N loss in Chinese agriculture because urea, the main form of N fertilizer used, is prone to such loss especially when applied to the soil surface as is the most common practice. Unpublished data from scientists at China Agricultural University (CAU, a major collaborator in several SAIN projects) shows that sub-surface application of urea can reduce NH_3 volatilisation from 27% of the applied N to 8%. This mode of application is achieved using a hand-held application machine that is suitable for use by small farmers: this innovation was highlighted in SAIN Policy Brief No. 5 and in documentation requested by the Climate Change section of the National Development and Reform Commission (NDRC) following discussion with members of a SAIN project.

It is well known that ammonia volatilisation is highly undesirable environmentally because much of the evolved NH_3 is deposited on soil, water or semi-natural vegetation causing nutrient enrichment and contributing to soil and water acidification. In addition, there is recent evidence indicating that NH_3 in the atmosphere contributes to the formation of fine particulate matter ($PM_{2.5}$ particles; Wu *et al* 2016) which are a major cause of poor air quality in many Chinese cities and associated with human respiratory disease. Thus any measures to decrease ammonia volatilisation, as well as being agriculturally beneficial, would directly contribute to improving human health.

2.2 Causes of N fertilizer over-use and inefficient use

The reasons contributing to the over-use and inefficient use of N fertilizer have been intensively studied, both in the SAIN programme and elsewhere, though SAIN projects have been particularly effective in identifying social and economic factors and the policy implications. There is clear evidence that the following factors are major contributors (Jia *et al*, 2013; Ju *et al*, 2016; Huang and Ding 2016):

- Very small farm size making it difficult for farmers to purchase N fertilizer in sufficiently small quantities.
- Small area allocated to each farm household, and the separated small plots they manage, make mechanisation difficult. Appropriate fertilizer application machinery would increase accuracy and uniformity of application and decrease risks of loss.
- Small farm size means that many farmers are engaged in off-farm work that is often far more important than farming as a source of household incomes. This works against professionalization of farm operations making it less likely that farmers will gain or implement knowledge on appropriate rates or timing of N fertilizer use or improved methods of application.
- Off-farm work is generally conducted by younger generations in families, leaving older people to undertake farm activities. In general, these people are less well educated and less likely to become aware of new management practices or technological developments.
- Over several decades government messages to farmers, and subsidy structures developed, have been directed towards increased fertilizer N use in order to increase national food security. This was entirely appropriate in the 1970s or early 1980s when N use was low, but the continuation of these policies now is outdated and contributes to over-use.

2.3 Policy responses identified through SAIN projects

2.3.1 Increased professionalism with N management

Any policies or actions that increase professionalism among farmers will almost certainly lead to more rational use of N, in terms of quantity, timing and method of application. There is general evidence globally that, as countries become more prosperous, N fertilizer use first increases and efficiency of use decreases but, at a later stage, N use efficiency increases and rates of application stabilise or even decrease (Zhang *et al* 2015). According to the analysis by these authors, using data up to 2011, most of China is still in the phase in which N use efficiency is decreasing. There is evidence in China of N being used more rationally as farm size increases. A survey of >800 farmers growing maize in Shandong Province, with input from SAIN, showed some decrease in total N applications with increasing farm size. It also showed that, as farm size increased, there was a trend for farmers to increase the number of N applications during the growing the season (Huang *et* *al* 2012); both trends are indicative of more rational use of N by farmers, reflecting increased professionalism. Another analysis based on national surveys, within a recently initiated UK-China Newton Fund project, also showed a strong trend for decreasing total fertilizer use with increasing farm size (Ju *et al* 2016). These authors also took this as evidence of greater professionalism by farmers, taking more notice of advice from extension services or other independent sources of information and paying greater attention to management practices.

There is a trend for farm size to increase in China due to "land consolidation" with some rural citizens taking over management of cropland as others move to urban areas. However, as noted by Ju *et al* (2016), there are strong social and economic factors slowing this trend, particularly the lack of social security coverage for households designated as rural. Thus changes in social security policy would be beneficial for promoting more rational agricultural practices with decreased adverse environmental impacts. Huang and Ding (2016) draw attention to the significance of local policy implementation in promoting land consolidation. Policies to permit and promote formal transfer of operational rights from one farmer (who wishes to move to non-farm work) to another farmer have been in place since about 2010 and arrangements are made through a land transfer service centre at township level. However, not all townships have such a centre and, in a study of farm sizes in north and northeast China, Huang and Ding (2016) found that farm sizes were significantly larger where such a centre existed. They also found that access to local mechanisation services, offering contract services, was highly influential in promoting land transfers and an increase in farm size.

Despite the trend towards increasing farm size, in the medium term it seems likely that smallholder farms will remain the main sector in China, albeit with a gradual increase in size. In 2006 the average area of smallholder farms, as opposed to collectives, was 0.43 ha; these farms represented over 200 million households and 98% of all agricultural land (Ju *et al*, 2016). This average area may be an underestimate due artefacts in the statistics: Huang and Ding (2016) estimated that average farm size nationally in 2013 was 0.78 ha compared with 0.61 ha estimated from official statistics. Therefore, concurrently with measured aimed at increasing farm size, it is necessary to develop policies that encourage improved management of N fertilizer by farmers with small areas of land. These policies, and any practical measures that follow from them, need to recognise that many of these farmers are likely to be part-time, possibly with the older generation left to undertake many farm tasks. One approach proposed as a result of an early SAIN project is to promote the role of contractors (sometimes termed *"professional service providers"*) for applying fertilizer and making decisions on the quantities and timings of application; see Zhang *et al* (2013) and SAIN Policy Brief No. 5 (http://www.sainonline.org/SAIN-Website(English)/download/PolicyBrief%20No%205%20Feb%202012.pdf)

The benefits of using contractors for fertilizer applications include:

- Technical information and training needs to be provided to a smaller number of individuals rather than to the >200 farm families.

- These individuals have the opportunity to become specialists, not having to combine their work with off-farm occupations; likely to lead to more appropriate timing of applications.
- They are likely to purchase specialist equipment for fertilizer application, giving more accurate and even application. Where appropriate, they are more likely than individual smallholders to purchase machinery to give sub-surface application of urea in situations where surface application leads to large losses of ammonia.

Individuals becoming contractors are likely to be farmers in villages with an interest in more technical or entrepreneurial activities. Contracting already exists in Chinese agriculture, for example, for seeding and harvesting using small machinery (Huang and Ding 2016), and in some regions for pesticide spraying. To facilitate an extension to fertilizer management would require a number of changes in policy and organisational arrangements. First, potential contractors would need to receive appropriate training and some form of accreditation; with suitable major modification, the FACTS scheme for accrediting fertilizer advisers in the UK may provide a framework (see: https://www.basis-reg.co.uk/Schemes/FACTS/About-FACTS). Through one SAIN project, several Chinese colleagues were made aware of the FACTS scheme and met those organising it during visits to the UK. To our knowledge, this is the first time that the FACTS scheme has been explicitly communicated to Chinese scientists. Second, there would need to be changes in policies concerning fertilizer pricing and subsidy arrangements to make the use of qualified and accredited fertilizer specialists financially attractive, for example by ensuring that the total cost to an individual farmers of fertilizer products plus cost of the service is no more than for the farmer purchasing fertilizer individually. This would probably have to be accompanied by a system of authorised fertilizer dealers having special arrangements for sales to accredited contractors.

Another route through which fertilizer decision making and application are put in the hands of specialists could be through Farmer Professional Associations. These have been established since 2006 when policy changes permitted their formation. At present they predominantly operate within the horticultural sector and tend to concentrate on bulk purchasing of materials such as those required for constructing plastic greenhouses, but their extension to joint purchasing of information on fertilizers would seem entirely possible and desirable.

2.3.2 Technical innovations

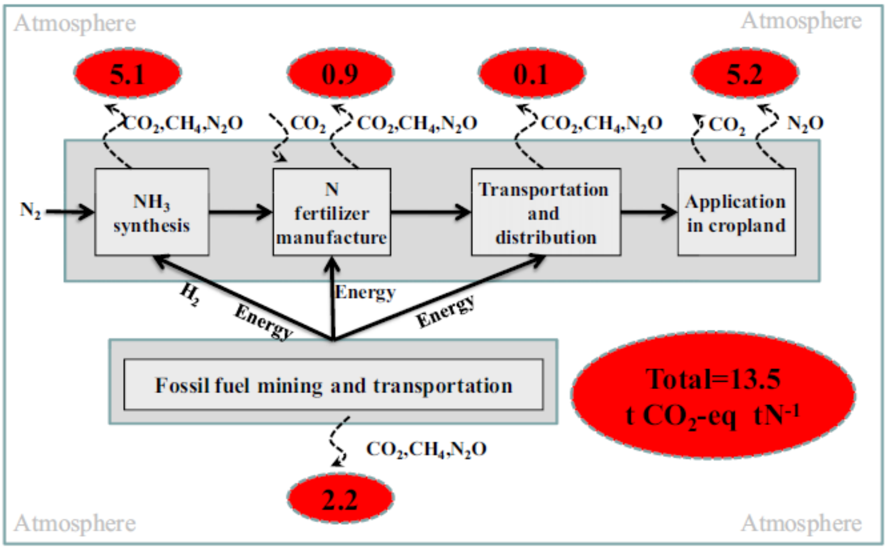
In addition to decreasing total quantities of N fertilizer applied to more rational rates, changes in timing are also very significant in reducing N losses, increasing N use efficiency, and permitting further reductions in the total quantity required to maintain yields. The usual change required is to apply less N at the time of seeding or planting and a larger proportion of the total later, when crops are growing vigorously with a rapid rate of N uptake. In many cases, increasing the number of later applications is also beneficial. However, this practice is more labour intensive compared with applying all or most N at the start of the season, so unattractive to part-time farmers. A technology that can assist is so-called “enhanced efficiency fertilizers” that slow or control the conversion of N into forms that are readily absorbed by crops, but also lost. These include products in which the fertilizer granules have a coating, leading to slow release, and those with inhibitors of urease or nitrification activity in soil included in the formulation. There has been much research on these, globally and in China, but so far their uptake in commercial farming practice has been limited due to the increased cost compared to standard fertilizers so they tend to be used in situations with high value products: typically for ornamental flowers, horticultural products or golf courses. However some inhibitors, especially urease inhibitors, are becoming cheaper; anecdotally some fertilizer manufacturers in China suggest that their inclusion in urea fertilizer manufacture may only increase cost by about 5%. Although there are data from experiments in many situations showing that inhibitors can be effective in decreasing N losses, and especially decreasing N_2O emissions (Gilsanz *et al*, 2016), they generally have little or no positive effect on crop yields, so are unattractive to farmers, even if the additional cost is small. There are several likely explanations for limited impacts of these fertilizer types on yields. In the case of N_2O, the quantities of N lost in this form are normally small, typically <5 kg N ha^{-1}; a decrease in emission of this magnitude is very significant environmentally because of the large greenhouse warming potential of the gas but is insignificant agronomically. Even where N losses by other mechanisms such as ammonia volatilisation or nitrate leaching are significantly decreased by using inhibitors it has generally proved difficult to clearly demonstrate a yield benefit. This is probably because experiments are often conducted with fairly high rates of N (Li T et al, 2018). It is therefore clear that farmers will not choose to use these products unless forced to do so through regulation or if the additional cost is met through a subsidy which could correctly be regarded as payment for environmental services. A review of this topic is in preparation, arising initially from an earlier SAIN project (Zhang *et al*, 2013 and SAIN Policy Brief No. 5) but now continuing within a Newton Fund project. It would seem logical to restructure subsidies in the fertilizer sector in China such that the current general subsidies to keep the cost of N fertilizer low be replaced by subsidies to meet the additional cost of including inhibitors.

In some situations it has been shown that use of an inhibitor can significantly decrease the rate of N fertilizer required to achieve maximum crop yield, a point overlooked in many studies; this was found in one experiment involving SAIN participants (Sun *et al,* 2015). In an experiment in Jiangsu Province with rice subject to mid-season drainage to reduce methane emission. Use of urea containing a nitrification inhibitor decreased the rate of fertilizer N required to achieve maximum grain yield by 60 kg N ha^{-1}. Although ammonia volatilisation was increased by the inhibitor, causing increased indirect N_2O emission, it was calculated that the overall impact on greenhouse warming potential was beneficial. But even in this situation, it is questionable whether saving on N fertilizer would compensate the farmer for the additional cost of the enhanced efficiency fertilizer.

2.3.3 N fertilizer manufacture

The manufacture of N fertilizer is very energy intensive and is usually the largest contributor to agriculture's overall carbon footprint (SAIN Policy Brief No 1). This aspect was considered in one SAIN project that assessed the greenhouse gas cost of different parts of cropping systems in China (Cheng *et al*, 2011; SAIN Policy Brief No. 8) and another that focused specifically on the manufacture and agricultural use of N fertilizer (Zhang *et al*, 2013; SAIN Policy Brief No. 5). Although the energy requirement for N fertilizer manufacture (and resulting CO_2 emissions) are widely recognised, the manufacturing aspect is often overlooked in studies of the GHG impact of agriculture and in seeking opportunities to decrease emissions. Fig. 4 (from the SAIN project reported by Zhang *et al*, 2013) shows the GHG emissions associated with each stage of the "N fertilizer chain". Manufacturing plus transport account for almost half the total emissions of 13.5 t CO_2-eq t^{-1} N. An additional 2.2 t CO_2-eq t^{-1} N is associated with the mining or other operations required to obtain the fossil fuel used in N fertilizer manufacture. This is significant in China as much of the manufacture relies on coal and there is significant emission of methane to atmosphere during coal mining, as discussed by Zhang *et al* (2013).

Figure 4. Life cycle assessment of greenhouse gas emissions from manufacturing and field use of N fertilizers in China and weighted emission factors (expressed as t CO_2-eq t^{-1} N) of the main processes based on current practices. From Zhang et al, 2013.



At the time of the SAIN study by Zhang *et al* (2013), almost two thirds of ammonia synthesis (the first stage of N fertilizer manufacture) in China was fuelled using coal, the majority in small scale facilities. Emission per tonne of ammonia produced in these is at least three times greater than in the most efficient large plants based on natural gas (Table 3). There is therefore considerable scope for reducing emissions by phasing out the small scale facilities using coal and, wherever possible, move to modern natural gas plants. This change, which is already occurring to some extent, requires a policy decision and it is recognised that conflicting interests exist. The small scale coal-based plants represent sources of employment so there is often resistance to their closure by local governments. At national scale, China has only small reserves of natural gas, with much having to be imported, but very large reserves of coal.

Table 3. Data on N fertilizer manufacture in China. From Zhang et al (2013)

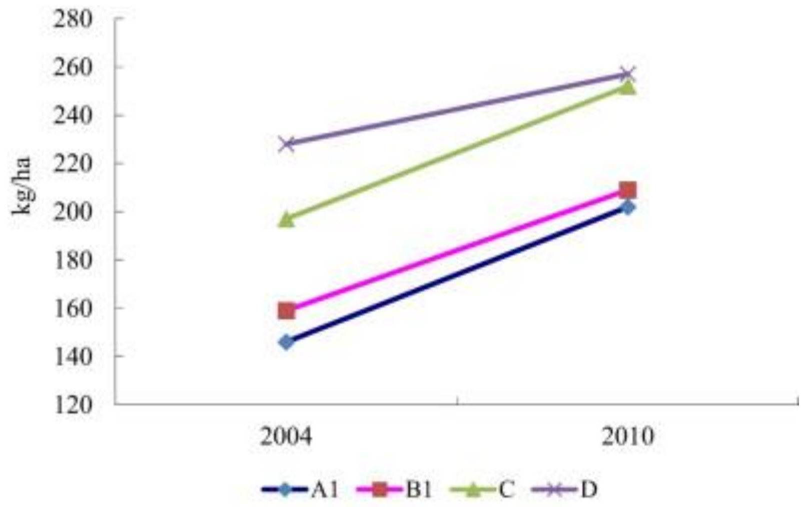
Type of manufacturing facility	Share of N production in China %	Greenhouse gas emission, tCO_2-eq t^{-1} NH_3-N produced
Small scale, coal	64	7.9
Large scale, natural gas	14	2.5

2.3.4 Farmer training and knowledge transfer approaches

There is a large body of evidence showing that the classic approach to extension ("expert tells farmer") is generally ineffective at altering farmers' behaviour. Even where decreases in N fertilizer use have been measured after training, it has been noted in SAIN projects and in other studies that intensive and continuing engagement between extension workers and farmers is commonly required if reduced applications rates are to be sustained and that achieving this for >200 million farmer families in China is challenging and costly (Huang *et al* 2012, 2015; Jia *et al*, 2013).

A highly significant finding in one SAIN-supported study concerns the initial and long-term impacts on farmer behaviour of different levels of training (Huang *et al*, 2015). The results were obtained through interviews with rice farmers receiving different intensities of training to determine what rates of N fertilizer they used and the extent to which this was altered by the training; initial interviews were conducted in the crop season immediately after the training sessions and repeated several years later. Some of the results are summarised in Fig. 4. With intensive training, including involvement of farmers in field experiments in addition to participation in training workshops, the effect on farmer behaviour in the subsequent crop season was substantial: on average it led to a decrease in N applications of 90 kg N ha^{-1} (Fig. 5). But if farmers were only given verbal training their decrease in N use was less than half of this. When interviewed six years later, N applications by all farmers had increased. The impact of intensive training was still evident, but the difference between N use by trained and untrained farmers had declined to 55 kg N ha^{-1}. For farmers that only received verbal training in a workshop, there was no continuing impact on their N use at this time compared with those receiving no training. It was concluded that, in order to have a lasting impact, intensive and repeated training would be required – and in the vast majority of situations in China this would not be feasible.

Figure 5. Impact of different levels of training on farmers' use of N fertilizer on rice crops in Jiangsu Province (from Huang *et al*, 2015)[a]



[a] Farmer types: A - trained and intensively guided by extension staff; B - trained, advice available from extension staff at farmers request; C – trained, no further intervention after training; D – non-trained

A study of extension in two counties in Shandong Province, partially funded through SAIN and led by the Centre for Chinese Agricultural Policy (CCAP) of CAS, revealed that extension officers were often assigned to other task, with only about 20% of their time spent on delivering public extension services and with little or no budget for this task (Jia *et al*, 2015). Globally, training approaches such as farmer field schools have proved more effective, in which "farmer trains farmer" with facilitation by a specialist and prior identification of innovative farmers. However, in a SAIN study in China also led by CCAP, even this approach proved less effective for communicating information on nutrient management than for pest management (Guo *et al* 2015). This raises important questions that need to be addressed regarding knowledge transfer in the context of the rapidly changing agricultural landscape in China.

Another approach, pioneered by key SAIN participants at CAU, is the "science and technology backyard" (STB) in which academics and students live in rural communities and strongly interact with farmers and introduce them to new or more efficient agricultural practices – see Yang (2016) and Zhang *et al* (2016). This has proved very effective in increasing crop production and improving efficiency and is also valuable as a means of training students to be effective extension workers. But, as with all approaches involving intensive activity by trained and skilled staff, questions remain about the extent to which it can be replicated across the country in a cost-effective way. However, given the success of existing STBs, it is an approach that deserves further study in order to learn and extract the salient features.

3. Increased production of meat and dairy products

Fig. 1 shows that between 1978 and 2016 meat production in China increased more than 8-fold, much of this being increased numbers of ruminant animals, especially cattle. This has been driven by the increased prosperity of the Chinese population and their changing dietary choices. The vast majority of the increased cattle numbers are in large scale operations with no associated land termed CAFOs (concentrated animal feeding operations) by the US EPA (In China, a CAFO is technically defined as any operation that has >50 pigs, 2000 broilers, 500 layers, or 5 dairy cattle. Chinese CAFOs are very small compared to CAFOs in developed countries, such as the United States).

3.1 Environmental consequences

There are numerous implications from this major change in agriculture but, before the establishment of SAIN, these had mostly not been evaluated in detail. Inevitably an increase in numbers of ruminant animals leads to increased emissions of methane from enteric fermentation; methane is a greenhouse gas with a global warming potential about 30 times that of CO_2. SAIN's first policy brief in 2010 highlighted the importance of these emissions. Two later studies by SAIN quantified recent methane emissions from enteric fermentation in China and another evaluated different approaches to decreasing emissions (SAIN Policy Brief No. 10). Using national level data for 2007, Norse (2012) calculated that methane from livestock accounted for 41-51% of total Chinese agricultural greenhouse gas emissions. Luo *et al* (2015) derived very similar values for the contribution of methane from enteric fermentation in milk production in Sichuan Province using data from household surveys in 2012 and IPCC emission factors for the various processes. The contribution was slightly smaller for large scale production units than for small scale household milk production. However, in this study, it was reported that the overall carbon footprints for production of animal products, expressed as kg CO_2-equivalent per kg product, were greater for chicken, pork and eggs than for milk. The largest contributions were from N_2O and methane emissions from manure treatment and storage and from emissions associated with the growing of animal feed; in large part, these feed associated emissions will be N_2O from the use of N fertilizer or manure applied to the forage crops. Because large quantities of soy bean are currently imported from South America, part of the crop-related N_2O emissions will have occurred elsewhere in the world.

An important impact of CAFOs is that large quantities of manure are accumulated with no ready access to land for utilisation. The environmental impacts of nutrients in manure, and their potential for replacing crop fertilizer requirements, had received relatively little research attention in China before the inception of SAIN projects. Because of the considerable effort devoted to these issues in the UK over several decades, sharing of this expertise has been a major success of SAIN.

In principle the concentration of manure within CAFOs can be beneficial because it avoids the practical challenges of collecting manure from large numbers of small farmers prior to some form of processing but, in practice, it often leads to major water pollution issues. This can arise either because of continued heavy applications to land near the source of the manure, often being applied to soil used for vegetable production where the nutrient content of manure is frequently ignored, or because of direct disposal into watercourses. The issue was highlighted by two papers in the SAIN Special Issue of *Agriculture Ecosystems & Environment* (Norse and Ju, 2015; Chadwick *et al*, 2015) and in several SAIN projects and Policy Briefs (Nos. 6, 12).

In the First National Census on Pollution Sources (MEP 2010) it was estimated that manures accounted for >90% of the chemical oxygen demand (COD), 38% of N and 56% of P discharges to surface water systems; they are also estimated to be responsible for 58% of total emissions of ammonia at watershed scale (Norse and Ju, 2015). Results from a survey of peri-urban regions around Beijing with large concentrations of animals were reported in the SAIN Special Issue of *Agriculture Ecosystems & Environment*. They showed that annual manure applications were very high: a mean of 46 t dry matter ha^{-1} yr^{-1} to all crops, but applications of up to 100 t dry matter ha^{-1} yr^{-1} to vegetables being common. This led to nitrate accumulations in soil to a depth of 2m in the range 500-1000 kg N ha^{-1}, and sometimes greater, and available P concentrations in soils of vegetable fields commonly at least 4 times the recommended value (Heimann *et al*, 2015). Fields such as these become point sources of N and P, contributing to eutrophication of surface waters as well as affecting drinking water quality. In addition, concentrations of copper and zinc (used as additives in animal feed) in soil were greatly increased when measured in 2009 compared to values observed in 1981, before animal production had become so dominant. High concentrations of several veterinary antibiotics were found in many soils that had received manures, pig and chicken manures being identified as the main sources (Heimann *et al*, 2015). As these can enter water used for human consumption they are a cause of great concern in the context of increasing antibiotic resistance being observed in several human pathogens, including recent reports in China (Wang *et al*, 2017).

In a separate SAIN project in the Beijing Municipality research focussed on soil P status in peri-urban soils in relation to manure and P fertilizer applications and the potential for water pollution (Jia *et al* (2015). Based on animal numbers, it was estimated that annual applications of P in manure increased almost 6-fold between 1980 and 2011. P applications as fertilizer increased about 3-fold over the same period, reflecting the fact that, until recently, residual P from manures was not taken into account when setting P fertilizer recommendations. These authors made recommendations regarding allocation of land for preferential application of manure to avoid further increasing soil P concentrations at locations where they are already high (i.e. greater application to cereal crops and less to vegetables) and avoiding sloping land where there is a large risk of direct movement of P to surface waters.

3.2 Policy responses

3.2.1 Limiting methane emissions from ruminant animals

Various additives to animal diets (probiotics, lipids, tea saponins) can, in some circumstances, decrease methane emissions from ruminant animals: their effectiveness was examined in the context of Chinese agriculture in a SAIN project reviewing a range of options for decreasing greenhouse gas emissions from livestock (SAIN Policy Brief No. 10) and by Wang et al (2012). If these additives prove to be effective, safe and acceptable it should be possible to regulate for their use, at least in CAFOs. Research in New Zealand is leading to development of a novel approach to mitigating methane emissions from housed cattle. Air from the animal house is circulated to bring it into contact with filters made from soil plus various additives that has previously been exposed to methane in order to build up the populations of methane oxidising bacteria. Although performance varies between products and conditions, in some situations >90% of methane can be removed (Syed et al, 2016). A similar approach is being tested in which a layer of filter material floats on the surface of liquid in dairy effluent ponds (Syed et al, 2017).

3.2.2 Improving management of nutrients and organic matter in manure

The quantities of nutrients contained in the animal effluent now produced in China are very large but, until very recently, were not well quantified. In a recent major SAIN study researchers carried out this quantification at national scale (Bai *et al*, 2016). They estimated that losses of nutrients through the manure management chain were very large: using data for 2010 it was estimated that 78% of excreted N and >50% of excreted P and K were lost to the environment. Through scenario analysis these authors suggest that improved manure management could lead to significant substitution of chemical fertilizer use by 2020 (27-100% reduction in fertilizer use) and 27-56% reduction in nutrient losses. To achieve this would require:

(1) Prohibition of manure discharges to surface waters.
(2) Improved collection and storage infrastructure.
(3) Improved manure application to cropland including methods to better take account of nutrients available from manure and application technologies to decrease ammonia volatilization.

These are all major transformations and require serious changes in practice that will certainly need government actions. Bai *et al* (2016) suggest changes to subsidies such that payments currently used to limit the costs of chemical fertilizers are devoted instead to the infrastructural and organisational changes needed to promote better utilisation of nutrients from recycled manures. In the case of improved recycling of manures from CAFOs, it should be possible to achieve change through planning regulations covering new operations and gradually tightening regulations on existing enterprises. Establishing infrastructure for collecting, storing and (where necessary) processing manure from large numbers of small livestock operations is more challenging. All of these approaches have been used in the UK during the past 20 years and this experience provides an opportunity for knowledge sharing with China. Finally it should be recognised that applying manure as a source of nutrients is more expensive and/or labour intensive for the farmer than applying chemical fertilizer, as pointed out by Bai *et al* (2016), and policies need to take this in to account.

In addition to the nutrient content of animal manures, they are also a valuable source of organic matter. At present their application is strongly focussed on soils used for growing vegetables or fruit: the organic matter content of these soils has increased to such an extent that they have become highly enriched in nutrients and a serious source of water pollution (Heimann *et al*, 2015; Jia *et al*, 2015). One of the drivers of this unbalanced utilisation is the fact that intensive animal enterprises are mainly sited close to cities, for easy transport of products to population centres. However, with the excellent road and rail communications now existing in China, it is no longer necessary for such enterprises to be adjacent to cities. If sited in rural areas, surrounded by cropland, the manure or its derivatives after anaerobic digestion for biogas production could more easily be utilised on cropland that has become low in organic matter content. Meat and dairy products could easily be transported to population centres using refrigerated trucks.

It is vital that any large scale utilization of animal manures, whether direct use of manure or slurry or use after composting or anaerobic digestion, takes account of public health issues. One issue is the risk of transmission of pathogenic organisms to humans, whether through the food chain or direct contact: the latter is an issue of concern for workers involved in manure handling. Another is the risk of antibiotic resistance that occurs in animal pathogens being transmitted to humans; this has recently been reported in China (Liu *et al* 2016; Sun *et al*, 2016) and has major implications for the use of antibiotics in livestock husbandry, not only in China (Wang *et al*, 2017).

4. Agriculture and low carbon development

Several SAIN projects have led to improved estimates of the different sources of greenhouse gas emissions from agriculture, the overall carbon footprint of cropping and livestock sectors, and of the potential contributions and costs of a wide range of mitigation strategies. These include Nayak *et al* (2015), Wang *et al* (2014), and Zhang *et al* (2013). A key finding from these studies, and other research, is that increasing the efficiency of use of N from fertilizers and manures, and decreasing N losses, is generally the factor that can make the greatest contribution to decreasing the overall carbon footprint of agricultural systems. In addition, as discussed above, in the livestock sector controlling methane emissions is a key factor. These conclusions reflect the reality that the non-CO_2 greenhouse gases, N_2O and CH_4, have very large greenhouse warming potentials: approximately 300 and 30 times that of CO_2. Appreciating the significance of these non-CO_2 greenhouse gas emissions from agriculture is extremely important for policy makers in the wider context of “low carbon development”. It is often not realised by policy makers that controlling these emissions is often more significant than a narrow concentration on CO_2 emissions or increasing soil C stocks to achive C sequestration. SAIN projects have been influential in communicating this to senior figures in the policy arena, including the NDRC and MOA.

4.1 Energy use and CO_2 emissions associated with irrigation

An aspect of agriculture that is often overlooked in this context, and where CO_2 emissions are the key aspect, is energy use associated with water use and irrigation. Rothausen and Conway (2011) have drawn attention to this at the global scale and Wang *et al* (2012) specifically for China in a SAIN project. Agriculture in China is responsible for 62% of freshwater use and, due to insufficient capacity from rivers, groundwater abstraction now accounts for about 70% of irrigation in northern China. Wang *et al* (2012) estimated that energy used for pumping groundwater and delivering it to farmers’ fields leads to annual emissions of 33.1 Mt CO_2 equivalent, accounting for 0.58% of total national emissions. This is increasing because water is having to be pumped from greater depths and there is also an increase in the use of pumping with irrigation using surface water, and hence further CO_2 emissions.

There are, of course, wider issues of water availability for agriculture and competition for water from household and industrial uses. From a sustainability viewpoint there is an urgent need to use water more efficiently in agriculture, including changing to crops that need. Surveys reported in the SAIN study of Wang *et al* (2012) showed that increasing the efficiency of pumps used for water abstraction is a promising way to decrease energy use and setting compulsory standards for pump efficiency would be an effective policy intervention. Good maintenance of pumps is important for ensuring efficiency and the use of renewable energy for pumping and water application should be explored; use of biogas from anaerobic digestion of animal manure in rural areas could be a win-win option.

Water conservation policies are in place to promote sprinkler and drip irrigation systems as these give greater efficiency of water use compared to flood or gravity irrigation. However, these newer more water-efficient systems require pressurisation to operate, thus increasing energy use. Thus, SAIN research has drawn attention to a previously unrecognised conflict between water conservation and energy use; further work is needed to address this issue.

5. Longer-term sustainability issues

5.1 Soil acidification

It is well known that continued application of N fertilizer in the ammonium form (e.g. as ammonium sulphate) or as urea leads to the gradual acidification of soil. This because the soil bacteria responsible for converting ammonium to nitrate produce H^+ ions as part of the process, thus causing acidification. The phenomenon has been studied at many locations worldwide including the long-term fertilizer experiments at Rothamsted Research. Prior to the establishment of SAIN the phenomenon was also investigated in China, mainly through collaboration between Rothamsted Research and CAU or CAAS. For example, Guo *et a*l (2010) used data from surveys covering almost 5000 sites to quantify soil pH changes between the 1980s to the 2000s in the major Chinese crop-production areas. There had been a marked increase in soil acidity (decrease in pH) during this period attributed mainly to the excessive use of N fertilizer rather than to acid deposition from the atmosphere, though this was also occurring. Although it is difficult to assess the impact of this trend on crop yields due to numerous confounding factors, there is evidence from elsewhere that soil acidification eventually has a negative impact on crop yields, e.g. in North America (Schroder *et al*, 2011) and UK (Goulding, 2016; Warren and Johnston, 1964).

In another pre-SAIN UK-China collaboration, funded through Defra and MoA travel grants, serious soil acidification was observed as a result of 15 years of inorganic fertilizer applications, including N, in an experiment at a sub-tropical site in Hunan Province (Zhao *et al*, 2010). In this case the situation was so serious that it led to almost total crop failure of wheat and greatly compromised yields of maize. Soil acidification is clearly a serious threat to the sustainability of cropping systems in China. Even with rational rates of N fertilizer the trend exists, but excessive use accelerates the problem. In many parts of the world this trend is counteracted by regular applications of lime; currently this is not widely practiced in China and it is sometimes stated that the cost of lime makes it impractical for farmers (Duan *et al*, 2011). In view of the widespread trend noted by Guo *et al* (2010) and the serious impact of acidification on crop yields in long-term experiments (Zhao *et al*, 2010), this is a topic that should be urgently addressed in China as it could well be a cause of “hidden yield loss". Research needs to be undertaken to quantify the impact of acidification on crop yields and its significance nationally; this could easily be achieved by initiating field experiments with lime with various crops in different regions. Only when this is done, and it is likely that it would be 3-5 years before clear results are available, can policy decisions be made on the appropriate course of action. For example, one possibility would be a subsidy to offset the cost of lime to farmers, but it would be premature to implement this until the scale of the problem is determined and the worst affected regions identified. In addition to yield loss, as soils become increasingly acid N and other nutrients will be used less efficiently by crops thus worsening pollution impacts caused by N or P not absorbed by crops.

In the UK the cost of lime was subsidised until the 1970s. Since subsidies ceased use of lime by UK farmers has decreased greatly: there is evidence that 40% of arable soils and 57% of grassland are now below optimum pH in the UK (Goulding, 2016) and it is very likely that crop yields are being affected. It is virtually certain that this is also the situation in China, though the evidence is circumstantial. Some data in China shows that applications of manure, combined with inorganic fertilizers, can counteract or slow soil acidification (Zhao *et al*, 2010; Duan *et al*, 2011). This is yet a further reason for utilising manures widely on cropland and not concentrating their use solely on high-value horticultural crops. Stimulated by previous SAIN research, a new study on this aspect has been initiated through one of the Newton Fund Virtual Joint Centres on Nitrogen Agronomy. However, maintaining soil pH in this way is not the full answer and does not necessarily give the benefit to crop yields achieved by lime, as shown by some of the data reported by Zhao *et al* (2010).

Research under SAIN has also demonstrated significant acidification of northern grasslands in China covering the Tibetan Plateau and Mongolian Plateau. Overall it was found that soil pH had decreased by an average of 0.63 units over a 20-year period between the 1980s and the 2000s (Yang *et al*, 2012). In this case the cause is likely to be acid atmospheric deposition as rates of N fertilizer in these regions of extensive grassland tend to be small. In these situations a serious consequence of soil acidification, in addition to possible loss of grass production, is the increased release of potentially toxic metals including manganese and aluminium as soil minerals degrade. A consequence of this is the entry of increased quantities of these metals into the food chain with potential impacts on human and animal health. This is an issue even where soils are not contaminated with metals but contain natural background quantities. However, it is known from other research in China that, where metal contamination does exist, liming is one means of decreasing entry of toxic metals into crops (Zhao *et al*, 2015).

5.2 Changes in soil organic carbon content

There is concern globally, as well as in China, about levels of organic matter (expressed as soil organic carbon, SOC, content) in agricultural soils. There are two main reasons for this concern. First, SOC concentration is a key factor determining the quality and functioning of soils, especially soil physical properties that strongly influence root growth and crop establishment. Second, the quantity of C (SOC stock) in soil is very large at the global scale and relevant to climate change: global SOC stock is currently estimated to equal about twice the quantity of C in atmospheric CO_2 (Batjes, 1996). If SOC stocks decline due to agricultural activities, additional CO_2 is released to the atmosphere, worsening climate change. By contrast, if additional C can be transferred from atmosphere to SOC, via plants, this can slow climate change – a process termed C sequestration. Several studies within SAIN projects have documented changes in SOC over time or the impacts of management practices on SOC: Yan *et al* (2011), Feng *et al* (2011). Other have used various approaches to estimate the potential for climate change mitigation through soil C sequestration, often in the context of a comparison of a range of climate change mitigation measures: Feng *et al* (2011), Cheng *et al* (2013), Wang *et al* (2014), Nayak *et al* (2015).

Yan *et al* (2011) compared SOC values in a set of >1300 soil samples taken from croplands in 2007-8 and compared them with data from a national survey made in 1979-82. The result was something of a surprise to many scientists: in the period of about 20 years SOC had increased by 8-31% in three of the major soil types examined, albeit from a low initial value. This was attributed to increased crop yields resulting from general improvements in agriculture in China (new crop varieties, fertilizer inputs, improved protection from pests, diseases and weeds, irrigation). Increased yields lead to larger returns of organic matter into soil from roots and stubble. In soils that started at a higher SOC level, there was little change between the two sampling times. By contrast, soils in one soil type concentrated in northeast China were found to have decreased in SOC content by an average of 22%; this was attributed to the relatively short period during which soils in this region had been in cropping, so SOC was still decreasing from the high natural level under the former forests or grasslands following clearing in the 1960s. In addition, there has been over-intensive cultivation in this region, including deep ploughing, which will have exacerbated loss of soil C. The authors drew some parallels with trends in SOC in England and Wales with similar processes operating. Using reduced tillage or zero tillage techniques for crop establishment, instead of mouldboard ploughing, would be expected to slow the loss of soil C. There has been much research on reduced tillage in China but the uptake in commercial practice has been low. This may be because of the cost involved in changing crop establishment machinery, particularly for small farmers.

There is considerable interest, globally and in China, in the possibility of mitigating climate change by sequestering extra C into soil organic matter through changes in land management. Several SAIN projects have addressed the issue and made estimations of the extent to which greenhouse gas emissions could be decreased through soil C sequestration as compared to other approaches: Cheng *et al* (2013), Wang, *et al* (2014), Nyak, *et al* (2015). Cheng *et al* (2013) draw attention to the important distinction between SOC increases that are theoretically possible (e.g. by comparing SOC stock in cropland with that in soil under natural vegetation; termed the "biophysical potential") and increases that are practically achievable within the constraints of the agricultural system – a distinction not always made. They concluded that there is some potential to sequester C in cropland soils through return of straw, reduced tillage and use of recommended fertilizer applications where these practices are not already being followed. They estimated that the practically achievable amount of soil C sequestration in China (around 0.8 Pg C) is only about 30% of the biophysical potential. Wang *et al* (2014) Nayak *et al* (2015) report a SAIN study using an approach previously developed by the UK partners, termed marginal abatement cost curves

(MACC curves), to compare the potential in China for mitigation through different changes in agricultural management and the cost (or in some cases cost saving) of each approach. They conclude that, whilst soil C sequestration does offer some potential, the greatest and quickest GHG emissions reductions in cropland can be achieved most readily through more rational management of N fertilizer and manure.

Adding biochar to soil has been widely discussed as an option for sequestering C, and possibly producing other benefits for soil quality and functioning. However, there are widely divergent opinions in the scientific community about (1) the scientific basis for claims regarding benefits of biochar, (2) its efficacy for climate change mitigation through differing interpretations of data, and (3) the practicalities and economics of biochar production and use. These differing views are reflected in different SAIN projects and biochar was included in the comparison of different mitigation options as reported by Nayak *et al* (2015) and Wang *et al* (2014). A study reported by Liu *et al* (2016) is supportive of the idea that biochar addition to soil can cause C sequestration and climate change mitigation. Clare *et al* (2014) considered the potential role of biochar in China from the perspectives of economics and social suitability. They concluded that commercially produced biochar is uneconomic as an independent farming input but, in one of their four case studies, farm-produced biochar showed some economic potential – though they did not specifically evaluate the scientific issues. They also suggest that biochar research in China should shift away from on-farm production and application of pure biochar, towards combined biochar-inorganic fertilizer products.

5.3 Value of crop wild relatives as resources for crop breeding and agricultural sustainability

This dimension of long-term sustainability has become of increasing importance over the past decade because of a range of agro-climatic factors and policy decisions. Two of them are of particular importance. First, the growing evidence that climate change is having both negative and positive impacts on crop yields, with shifts in rainfall patterns and the demand for irrigation having serious implications for food security (Wang *et al*, 2017). Second, the GOC's target, announced in 2015, of achieving zero growth in fertilizer and pesticide use by 2020. Both challenges require crop improvements using all the genetic resources and advanced plant breeding tools available including crop wild relatives. The latter were the focus of one of SAIN's earlier projects.

Maize farmers in NE China are already responding to rising average temperatures by using longer maturing varieties (Meng *et al*, 2014). However, conventional responses are unlikely to be sufficient and crop wild relatives (CWRs) are an important source of diversity for crop improvement. They are likely to become increasingly important for adapting crops so that they can yield adequately under a range of increasingly detrimental abiotic and biotic stresses, as well as under more frequent and extreme climatic fluctuations.

China is one of the world's key centres of diversity for major food crops so it is important to assess which CWRs have the greatest potential as gene donors and which are under threat from climate change or land clearance. The SAIN project developed a methodology for prioritizing China's CWRs and formulated recommendations for a conservation strategy to protect them (Kell *et al*, 2015).

Furthermore, the project has produced an inventory of China's CWR which contains more than 24,000 species of actual or potential economic importance to China and the global community. The CWR inventory includes food, fodder, forage, industrial medicinal and ornamental crops, and has provided baseline data to help shape CWR conservation policies in China. For example, China is one of the world's main sources of wild grape (*Vitis*) germplasm (Jiang *et al*, 2015). Some of these grape CWRs are vulnerable to climate change and other pressures. About 22% are in nature reserves and are well protected, but many are not and need special measures to conserve them. Climate change may change this situation with both positive and negative impacts on the size of the areas favourable to grape CWR growth and this should be considered in conservation strategies.

6. Climate change impacts on agriculture in China

There has been considerable effort in China to understand and predict the likely impacts of climate change on agriculture, especially on crop yields and likely variability of yields between years, and some aspects have been investigated by SAIN projects. As would be expected in a country the size of China, impacts will vary between regions and agricultural systems. This is summarised succinctly in a statement in a paper describing SAIN work by Ju *et al* (2013b): "*Climate change can bring positive and negative effects on Chinese agriculture, but negative impacts tend to dominate*." However, in the short term, longer growing seasons in NE China have been beneficial for agricultural production. Ju *et al* (2013b) state that already the frequency and intensity of extreme weather events, especially drought, have increased. They also draw attention to more frequent and serious outbreaks of crop diseases and damage from insect pests and of soil degradation caused by climatic change and extreme events. The general increase in temperatures have already reduced cold damage in northeast China. It is projected that temperatures will increase by 3.9 – 6.0 °C by 2100 and average precipitation will increase by 9-11% but the spatial distribution of these trends will be crucial. And higher temperature will increase evaporation, so the possible benefits of increased precipitation may not be realised. Despite an average small increase in precipitation, drought is likely to become more prevalent in some key regions. Climate change will affect the optimum location of various crops and the scope for double or triple cropping will generally move northwards (Ju *et al*, 2013b). If no adaptation measures are taken this study shows that yields of irrigated wheat, maize and rice are projected to decrease by 2.2-6.7%, 0.4-11.0% and 4.3-12.4% respectively in the 2050s compared yields in 1961-1990. With adequate irrigation they consider that these decreases can be mitigated, but this will increase the conflict between agricultural and non-agricultural uses of water in the drier regions.

In another SAIN project (Ju *et al*, 2013a) the authors analyse in detail the factors contributing to uncertainty in making predictions of climate change impacts on crop growth and indicate the directions of research to improve simulations. Current model simulations that they reviewed showed that yields of all three major grain crops (maize, rice, wheat) decline under climate change, with only wheat showing an increase in some regions.

Other SAIN projects included a modelling approach to assess vulnerability of wheat to drought in a key region of North China (Li *et al*, 2015), and a study of the influence of increased atmospheric CO_2 concentration on the nitrogen dynamics of wheat, with implications for N fertilizer management, utilising a CAAS FACE (free air CO_2 enrichment) facility (Han *et al*, 2015). A major and high-profile assessment of climate change impacts on global food security (Wheeler and von Braun, 2013) was, in part, informed through UK-China interactions through SAIN.

7. Conclusions and wider policy issues regarding resource management and long-term agricultural sustainability

This final section brings together various conclusions from the preceding sections in the context of one of SAIN's primary objectives when is it was established in 2008, namely the promotion of collaborative research that supports the formulation of sustainable agriculture development strategies directed at achieving comprehensive gains in resource use efficiency. In China the aim was to promote the development of a circular economy and agriculture as part of a resource efficient and environmentally friendly society. In the UK the emphasis was on building a low carbon economy of which agriculture was to be an important part (Stern, 2007 and HM Treasury, 2007). Both concepts have the same fundamental requirements, namely the need for holistic approaches developed through inter-disciplinary collaboration that produce dynamic and de-centralised mechanisms for implementation. The UK-China collaboration under SAIN that fulfils these requirements is illustrated by the work on non-point source pollution and diffuse water pollution (NPS/DWP).

NSP/DWP first became an issue in the UK in the 1970s as a result of catchment studies on nitrate leaching into the groundwater. Its importance increased substantially in the early 1990s with the introduction of EU legislation to protect drinking water quality and reduce agricultural nitrate losses to ground and surface water. The need to act on this through legislation highlighted the lack of information on the geographical distribution of the nitrate problem, the management practices that affect N losses, and the practical ways of reducing these losses by technical and economic interventions. Hence the former UK Ministry of Agriculture funded a comprehensive multi-million pound R&D programme throughout the 1980s and 1990s covering both crop and livestock systems, which provided the evidence base for the mandatory and other control measures introduced in 1998. This R&D programme still continues but is focused more on improved monitoring of N losses.

The issue of NPS in China did not receive much attention until the late 1990s. As in the UK the first concerns were about increasing concentrations of nitrate in groundwater aquifers used for drinking water (Zhang *et al*., 1996) and, later, the increasing occurrence of algal blooms in lakes and rivers. It did not become a national issue until some 15 years later (Zhu *et al*, 2005), and as in the UK the early research highlighted the need for a greatly improved physical and social science data base for the formulation of appropriate policy responses. China's Five Year Plans since 2006 have noted the need for action on NSP but as yet there are no mandatory control measures, and the environmental consequences of NSP are getting worse.

Thus the UK has over 40 years of NPS related research and some 30 years of policy experience to share with China. So the central task of SAIN's Working Group 4 has been to formulate and complete a China-UK project on "Knowledge, policy and practice for sustainable nutrient management and water resources protection in UK and Chinese agro-ecosystems", which has also been a secondary objective of other working groups. Furthermore, the project was designed to provide advice on how the concept of the circular economy can be applied to agriculture by exploiting the opportunities for greater recycling, waste minimisation, and more efficient use of water and other critical resources identified by the research considered in sections 2-6 above.

The project's finding are summarised in SAIN Policy Briefs 12, 13 and 14, and in several publications and they complement and/or substantiate the results of the other collaborative research supported by SAIN, which can be brought together as a series of key messages.

- Although the overuse of N has started to decline in some areas and the GOC has set a target of zero growth in N fertilizer and pesticide use by 2020 there is still need for substantial improvements in the availability of basic information on nutrient sources and pathways. The latter include the nutrient content of synthetic fertilizers, which are often poorly labelled and of low quality; the nutrient content of manures; the N content of irrigation water which, in some regions, may exceed that required for a cereal crop and is usually ignored.

- Spatial planning needs to play a greater part in the location of intensive livestock operations (CAFOs) and high output vegetable enterprises, and in the formulation of NPS/DWP measures at the catchment and water basin level.

- The foregoing planning measures should be complemented by the more comprehensive development of building regulations on waste disposal and storage requirements to promote nutrient recycling and limit NPS/DWP.

- A complete reassessment is required of the current economic instruments and other support measures for technological innovation and farm uptake. This should include the perverse subsidies given to fertilizer manufacturers, and those given to farmers which are more in the nature of income supplements than incentives for technological improvements. And should bring together Chinese and UK experience in the use of payments to farmers for ecosystem services.

- Action plans need to be developed that take greater account of the needs and capabilities of different farm sizes. For example, certain regulatory measures may be feasible for large scale commercial farms but unrealistic for smaller ones. Similarly certain incentive measures and simple guidelines may be effective for averaged sized farms and may be a more effective way of promoting greater resource efficiency and reducing environmental impacts.

- Many of the most cost-effective climate change mitigation measures available to China involve improved N management in the crop and livestock sectors.

All these keys messages are consistent with the overall conclusion of SAIN's research programme to date. Indeed, SAIN projects have been influential in drawing attention to these issues and contributing to the positive changes that are beginning to emerge in China and to increased understanding of the issues in the UK and globally. The achievement of sustainable agriculture in a manner which is consistent with the wider objective of a low carbon, resource efficient circular economy requires a holistic approach to strategy development and implementation formulated through the type of interdisciplinary research and UK-China collaboration that SAIN has promoted. So there is no single or even a small subset of regulatory or policy measures that can bring it about. It will require a complex mixture of regulatory measures and incentives operating at multiple scales and involving multiple agents. Moreover, the full benefits of such a holistic approach will not be received in the short-term. For example, it is clear from the UK experience that the overuse of N can be reduced within 10 years but restoration of the freshwater ecosystems damaged by high N inputs can take 20-30 years.

None the less it is reassuring to see that some of the policy measures suggested by SAIN's research reviewed above are already emerging in official communications and policy changes. One major example is the "zero increase" policy announced in 2015 referring to there being no increases in the use of fertilizers or pesticides after 2020, though agricultural production must be maintained and increased. There is, however, no mechanism in place to monitor compliance. Another is the "Regulation on the Prevention and Control of Pollution from Large-scale Breeding of Livestock and Poultry" which came into force in 2014. Moreover, these specific actions are taking place in the context of consistent GOC support for agriculture such that for the 15th consecutive year, the "No. 1 central document" by the Central Committee of the Communist Party of China and the State Council has been devoted to agriculture, farmers and rural development. The 2018 document states that the country will "increase the output of high-quality products based on green and innovative production" and that "China will step up training for professional farmers, including professional agricultural managers, advisors and service providers".

In conclusion, it is apparent that SAIN is a unique and effective model of bilateral research collaboration. No other country has joined with China in adopting such a systematic approach to the development of research collaboration or is able to launch a comprehensive bilateral research programme equivalent to the one now supported by the Newton Fund. Many of the programmes and projects supported by the Fund build on work started by SAIN and its extensive network of researchers in the UK and China.

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