Does higher-yielding agriculture mean more environmental harm?

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Summary

A criticism of the land-sparing approach to preserving biodiversity, by restricting farmland to a smaller, higher-yielding area, is that other impacts are higher in food produced this way. This study aims to investigate the evidence for this based on currently available data and models for greenhouse gas emissions, N, P and soil loss and water use. We asked 25 experts to identify and supply data to plot environmental impact per unit of product against yield for the beef, dairy, wheat and rice sectors. This produced data from modelling and field trials and the lifecycle assessment and field trial literature. The data were modelled statistically to adjust for differences between the studies. Given data limitations, it does not seem that higher yielding agricultural production has higher impacts, often quite the reverse. We ask those conducting field studies to collect data that can definitively answer this question.

Key words: Land sparing, externalities, yield-scaled

Introduction

Empirical data quantifying population-level responses to changing agricultural yield (production per unit area) consistently indicate that higher-yielding agricultural production is likely to be more beneficial for biodiversity, provided this allows farming to occupy less area and so leave more land under natural vegetation (Balmford *et al*., 2015, Phalan *et al*., 2011). One major criticism of this approach by those advocating less intensive, land-sharing approaches such as organic production, is that higher-yielding production is responsible for higher environmental impacts such as pollution (Tscharntke *et al*., 2012) – but this suggestion is largely untested. We examine this question by drawing on available data sets and models to compare yields and environmental impacts across different production methods within each of four agricultural sectors. The impacts considered are greenhouse gas (GHG) emissions, nitrogen (N) and phosphorus (P) pollution, water use and soil erosion. The sectors included are Latin American beef, UK wheat and dairy and Chinese paddy rice production. Some preliminary results are presented here.

This study aimed to use data from the published literature and peer-reviewed tools to produce a series of biplots for each sector of how environmental impacts per unit production vary with yield, across alternative management practices. A crucially important consideration is that the data in such plots must be corrected for any effects of site or measurement method, so that they solely reflect the effects of implementing alternative practices within a given location. These biplots would then allow us to determine how each of our environmental impacts tends to change with respect to yield (Fig. 1). It may also be possible to identify better- and poorer-performing agricultural practices.

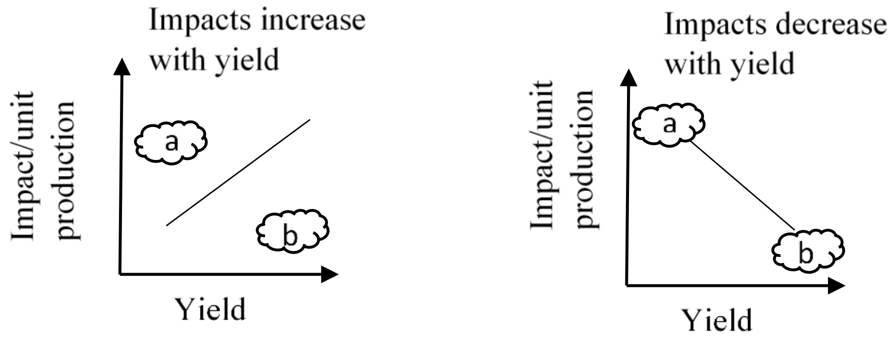

Figure 1 Potential impact/yield biplots. It may be possible to identify the management options which perform worse (a) and better (b).

Methods

The approach we have taken is to work with a team of experts who could provide expertise on each of our focal sectors and environmental impacts to enable us to identify available methods and datasets that can quickly be used to provide an initial indication of how impacts vary with yield. This has resulted in three main methods. 1) Use of an existing, single-site study where sufficient yield and impact data were available to produce a biplot of a suitable range of agricultural practices. This approach was used for P and soil loss in wheat production and nitrogen and phosphorus loss in paddy rice. 2) Use of a variety of studies to generate impact/yield plots while accounting for differences between the studies and locations through statistical modelling. This was the approach used for GHG emissions in paddy rice and beef and for water use in paddy rice. 3) Using process-based models to estimate how impacts change with yields for a range of agricultural practices. This approach was used for all dairy impacts and also as an alternative method for beef GHG emissions, for comparison with the statistical modelling method.

Most of the experts came together for a 4-day workshop where the methods were agreed and data sources identified. Prior to the workshop a conceptual model diagram for each sector was circulated to the experts for that sector to develop a shared understanding of how management affects environment impacts in the sector (Fig. 2). These were modified and recirculated and presented by each sector at the workshop. This was used to set system boundaries and prioritise literature searches. The arrows indicate the farm element that contributes to the environmental impact, with the width giving an indication of the relative importance of the pathway.

Of the 20 possible combinations of agricultural sector and environmental impact, we have identified data for 13 possible biplots (Table 1). All 20 were regarded as a having relevance to sustainability, with the exception of water use in UK wheat, where extractive water use is not usually practiced. We were unable to find sufficient suitable data for the other six plots. This was generally because yields were not reported together with environmental impacts for a variety of agricultural practices at a given site within our identified study region.

The data we gathered for this exercise has come from a variety of sources. Of the environmental impacts, only GHG emissions data were found for all of the sectors. This was facilitated by the international standardisation of reporting due to the high level of interest in studying climate change. Note that the land-use change component of the GHG emissions is not yet included in the results.

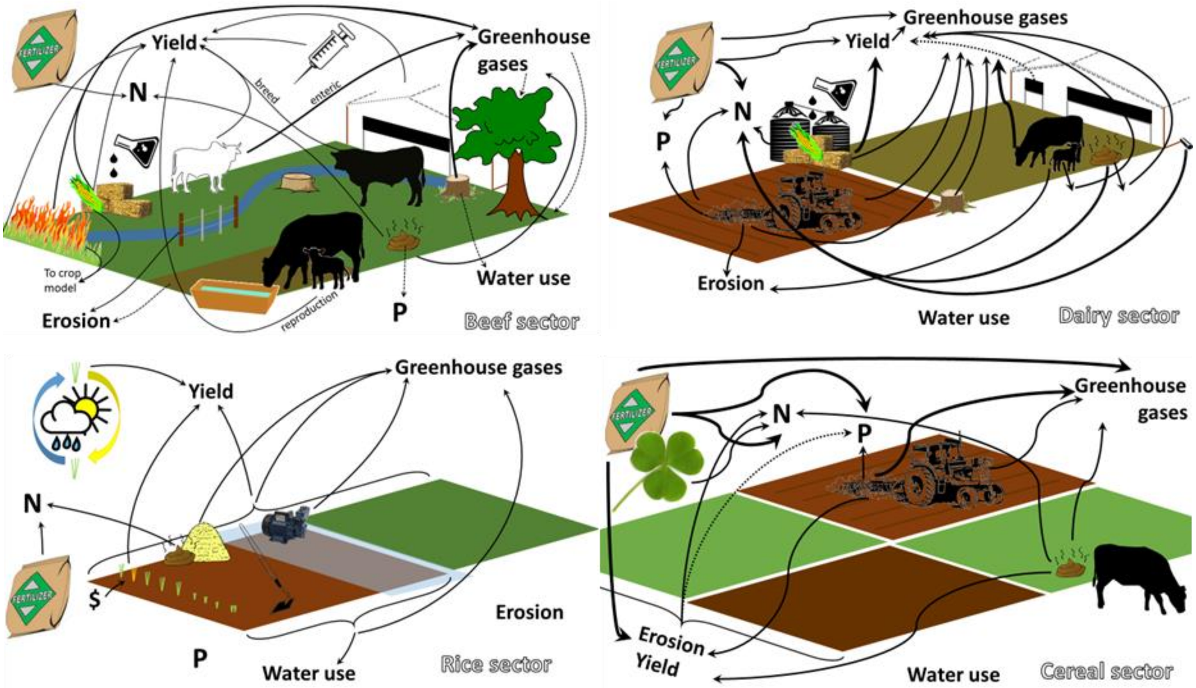

Fig. 2. Conceptual models of management factors which influence environmental impacts used to develop shared system understanding.

Table 1. *Agricultural yield and environmental impact data found for this study*

	Beef	Dairy	Wheat	Rice
GHG	Statistical analysis of 7 Brazilian LCA studies (N=29) and a Mexican farm survey (N=6) Modelled enteric emissions	Modelled enteric emissions, + IPCC manure emissions	Statistical analysis of Rothamsted field trials (N=96)	Statistical analysis of 20 Chinese field trials (N=182)
N pollution	× (lack of data)	Modelled emissions	Statistical analysis of 100 Rothamsted field trials	Single-site field study
P pollution	× (lack of data)	Modelled emissions	Single-site field study	Single-site field study
Water use	× (lack of data)	× (lack of data)	×(not relevant)	Statistical analysis of 142 Chinese field trials
Soil erosion	× (lack of data)	Modelled emissions	Single-site field study	× (lack of data)

For Latin American beef production we combined Mexican data from a National Autonomous University of Mexico farm survey (Ponce & Hernández-Medrano, 2016) with Brazilian life-cycle assessment data provided by Erasmus zu Ermgassen (to be published). We also ran the process-based RUMINANT model (Herrero *et al.*, 2013) for enteric emissions for a range of production systems including extensive grazing, improved pasture, and breeding, supplementary feed, feedlotting and silvopasture. We were unable to find data for the other impacts for beef production. This was partly because the reporting of the impacts rarely also reported yield. A search was performed for P pollution, water use and soil loss, but insufficient matching data were found.

A University of Nottingham process-based model (based on Garnsworthy, 2004; Wilkinson & Garnsworthy, 2017) was used to quantify enteric emissions and manure output for UK organic and conventional dairy production. IPCC methods were then used to quantify manure emissions. These results were then combined with emissions modelling from Rothamsted Research to produce estimates for N and P pollution and soil loss. We were unable to find water-use data for management practices, which also reported yields.

Wheat production data came from Rothamsted Research long-term field trials (Bell *et al*., 2015; Harris *et al*., 1984; Cannell *et al*., 1986; Catt *et al*., 2000; Goss *et al*., 1993; Eltun *et al*., 2002) and included statistical modelling to combine studies for GHG emissions, and single study data for N and P pollution and soil erosion.

A large number of studies of Chinese rice production have been reported in the literature, which enabled the statistical combination of studies for GHG emissions and water use. There were smaller numbers of compatible studies for N and P, so single studies were used. To be able to include a large proportion of N emissions for paddy rice, the most important fractions of the N balance need to be included. These are NH_3 volatilisation, and NH_4 and N_2O runoff, but we did not find these reported together with yield across a range of production intensities. Total P in runoff was used for P loss.

Results and Discussion

The modelling is not finalised but some preliminary results are summarised here. These results are limited to the general direction the trend appears to take at this preliminary stage (Fig. 3).

For most of the plots, the variation in production systems that we were able to examine suggests that higher-yielding production is associated with lower impacts per unit of production. This includes all of the impacts modelled for dairy production, GHG emissions for beef and wheat production, N and P emissions for wheat and rice. For wheat soil loss, the results came from a study which varied tillage practices, which made a much larger difference to soil loss than it did to the yields. The rice GHG data suggest complex correlations that are still under investigation. In no case did we find that higher-yielding production systems were consistently associated with higher impacts per unit of production.

We are able to identify some preliminary management practices that perform better and worse for each of the sectors for which we have data. For GHG emissions in the beef sector it seems that improved pasture together with cell grazing performs best, and extensive grazing that includes neither of these performs least well. From the modelling and such empirical data as we have, silvopasture may perform best of all, but we were unable to include silvopasture empirical data in the study due to lack of compatibility. We would really encourage silvopasture researchers to collect data in a similar way to lifecycle assessment studies so that it can be compared on an equal basis. This would mean collecting manure N_2O emissions data even though such emissions are thought to be comparable to background levels. For example, the systems for which we had data were intensive silvopastoral systems with high density leucaena fodder plantings, which are of interest because of their high yields and low enteric and manure emissions and high carbon stocks above and especially below ground. But the data came from a variety of sources, which were not comparable. There are diverse types of silvopasture with widely varying emissions and yield performance, so standardisation and description of type used in studies is necessary to be able to combine datasets.

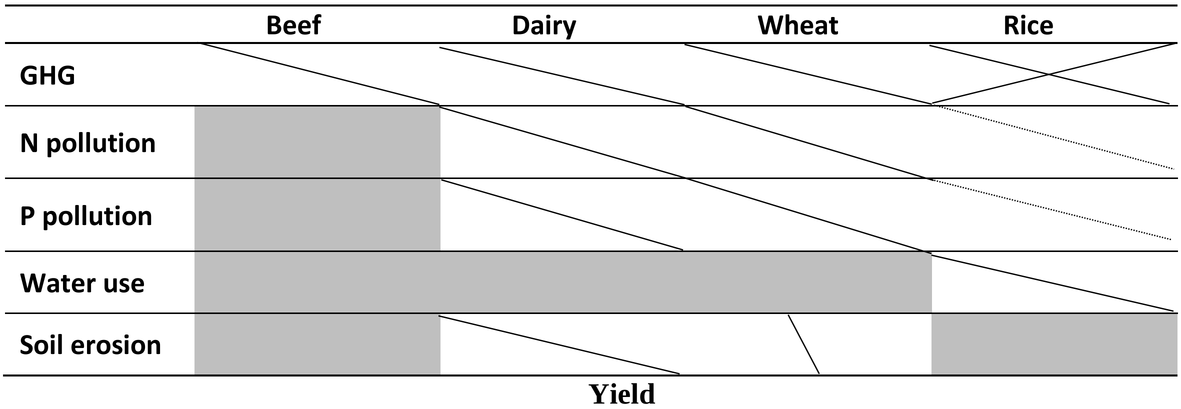

Fig. 3. The general direction of the trend of the yield/impact plots, based on preliminary analyses. Most of the plots show a negative relationship and none show a clear positive relationship.

For dairy GHG emissions, conventional production including inorganic fertilisers resulted in higher yields and lower emissions per unit of milk produced, especially with solid manure storage, which reduced emissions. Organic milk production performed least well, especially when cows are fed by exclusively extensive grazing. The results are the same for N and P pollution and sediment loss, largely because the milk yields are unchanged. There is less difference in N pollution rates between organic and conventional production.

The study of UK wheat GHG emissions varied N application rate, up to 300kg/ha/yr. Highest N applications produced highest yields and lowest GHG emissions, and conversely, the lowest rates had lower yields and much higher emission per kg of grain. Similarly with N pollution, the highest N application rates resulted in highest yields and lowest emissions per kg of grain, and applying no N resulted in the worst performance. Presumably the N rates did not exceed the level where yields respond, but if application rates exceed these levels then emissions per unit product would increase. The study that we used for P loss varied between organic, conventional and mixed livestock production with and without incorporation of farmyard manure. The systems that performed best were conventional arable and integrated forage, both without the addition of farmyard manure. The organic systems with farmyard manure performed worst for both P loss and soil erosion. Conventional forage without farmyard manure performed best.

For rice GHG emissions, N application rate increased yield and made little difference to GHG emissions. Continuous flooding was associated with higher emissions, but this was confounded by tilling after flooding. Multiple mid-season drainage seemed to increase yield but have little effect on emissions. Incorporating straw was associated with higher emissions, and other organic matter increased yield with no discernible effect on emissions. N loss appeared to decline with yield when considering the largest N component, NH_3 volatilisation, which may account for a little over half of the N loss for paddy rice. This will probably dominate the effect of the N_2O and NH_4 in runoff, where the relationship seems to go slightly in the other direction, because the volatilised quantities are larger, but this would need to be measured together in a single study to improve confidence. Also, it may be better to use eutrophication potential rather than total N and total P as an indication of potential environmental harm. Based on volatilisation alone, the study compared N application methods and found that basal application of controlled-release nitrogen fertilisers combined with urea top-dressing at the tillering stage performed best and a single basal application of sulphur-coated urea performed worst. The study of P emissions varied P fertilisation methods. Most P was lost in major rain events, when applying no P or half superphosphate and half pig manure performed best, and the highest rate of superphosphate performed worst. When there were no major rain events there was a relatively low rate of P loss and in this case the P loss/yield relationship went in the other direction, but the effect of rain events dominated.

Many studies of GHG emissions for agriculture present their results per unit area rather than per unit of product (eg Li *et al*., 2006). This tends to favour production systems that use land inefficiently. Such results need to be reanalysed to discover the yield-scaled GHG performance, which typically has the opposite trend. A study by Pittelkow *et al*. (2014) compares the two and found that optimal N application can address land use and GHG emissions simultaneously. Li *et al*'s modelling includes some of the same management variables as we do, and found that shallow flooding made a greater difference to GHG emissions than midseason drainage, which made more difference than moving straw incorporation to the off-season.

None of our preliminary results include the GHG emissions from land-use change. These are generally larger than the emissions due to agricultural production (Ranganathan *et al*., 2016; Smith *et al*., 2016), and would of course tend to be greater (per unit of production) for lower-yielding production, and so accentuate our findings to date.

Conclusion

The tentative conclusions from our preliminary results are that higher-yielding production does not tend to perform worse for measured environmental impacts when assessed per unit of production. Our findings are inevitably constrained by the limited range of environmental impacts which have been measured (alongside yields) across management practices. Data from other studies could be used more readily if yields were reported together with impacts, if data were consistently collected for whole years rather than just the growing season, and (in the case of N pollution) if all of the most significant fractions of the problem were measured. More generally, agricultural impacts need to be reported per unit of production, rather than per unit of land, in order to take into account land-use efficiency.

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