Integrating economic and environmental impact analysis: the case of rice-based farming in northern Thailand

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Abstract

Crop production is associated with a range of potential environmental impacts, including field emissions of greenhouse gases, loss of nitrogen and phosphorous nutrients to water and toxicity effects on humans and natural ecosystems. Farmers can mitigate these environmental impacts by changing their farming systems; however these changes have implications for production and profitability. To address these trade-offs, a farm-level model was constructed to capture the elements of a rice-based production system in northern Thailand. Life Cycle Assessment (LCA) was used to generate environmental impacts, across a range of indicators, for all crops and associated production processes in the model. A baseline, profit maximising combination of crops and resource use was generated and compared with a greenhouse gas minimising scenario and an alternative inputs (fertilisers and insecticides) scenario. Greenhouse gas minimisation showed a reduction in global warming potential of 13%; other impact indicators also decreased. Associated profit foregone was 10% as measured by total gross margin. With the alternative farm inputs (ammonium sulphate, organic fertiliser and fipronil insecticide), results indicated that acidification, eutrophication, freshwater and terrestrial ecotoxicity impacts were reduced by 43, 37, 47 and 91% respectively with relatively small effects on profit.

Keywords: Rice; Bio-economic model; Optimisation; Life Cycle Assessment (LCA); Thailand

1. Introduction

Farmers make decisions on what to produce, the timing and level of variable inputs used in production and over the longer term, the level of land, labour, machinery and other capital resources. Although they have multiple objectives, including management of risk, it is clear that farmer responses to changing output and input prices are guided by profit seeking behaviour. For example, recent global elasticity estimates indicate that production supply response to own crop price changes is positive and significant – through both area and variable input change – for soybeans, maize (corn), wheat and rice: four of the world's major food crops (Mekbib *et al.*, 2016). If price changes fully capture all opportunity costs of production and if society is prepared to rely on new input and output technologies to meet a growing and changing demand for food, it could reasonably be concluded that the mainstream, commodity-based agricultural production on which the world relies is sustainable - and will continue to be so. However, it is clear, from theory and mounting evidence, that prices do not give a true indication of the full cost of agricultural production. Agriculture is subject to negative and positive environmental externalities: the prices of some of agriculture's major inputs - nitrogen and carbon in particular - are too low (or zero) when they leave the farm system in a form that has detrimental impacts beyond the farm. To take one major input, nitrogen fertiliser, as an example, Gruber and Galloway (2008) argue that "massive acceleration of the nitrogen cycle" is driving emissions of nitrous oxide and ammonia to the atmosphere and loss of nitrate to water; respectively contributing to global warming, acidification and eutrophication pollution problems. In contrast, biodiversity and other ecologically-based outputs and resources are undervalued and thus undersupplied or managed inappropriately. The profit-seeking behaviour of farmers will therefore tend not to be optimal from a wider societal viewpoint, particularly if a longer term view is taken. If the above framework of farmer response to costs and benefits is accepted; and if a better allocation of resources is desired, it is necessary to understand and measure the nature of agriculture's environmental effects. A further step would be to value these effects - and for these valuations to respond to changing scarcity. However, this is often not pragmatic, not least because valuation is difficult and tends to divide researchers from different disciplines. An alternative framework for analysis, employed in this paper, is to make greater use of the increasing amount of information available on the *physical* impact of agriculture on the natural environment through techniques such as Life Cycle Analysis (LCA, e.g. Blengini and Busto, 2009), the use of mechanistic models (e.g. Gibbons et al., 2005) and the development of environmental metrics and indicators (e.g., Moldan *et al*., 2012). When combined with bio-economic models that capture the elements of decision making described above (for example, as described in Janssen and Van Ittersum, 2007), this information can be used in three important ways. First, the cost of achieving some environmental outcome can be evaluated; a more subtle variant of this is to evaluate costs 'with' and 'without' *adaptation* – in the former, the system is allowed to change; in the latter the system retains some or all of the features of its original state. Second, new *interventions* designed to address sub-optimal environmental outcomes can be modelled. These can be introduced as different policy options – for example, to compare regulatory- or incentive-based approaches to achieving a desired outcome. Third, the effect of change on other aspects of the system can be assessed: land use, production, calorie and protein supply, susceptibility to risk, other environmental outcomes.

In this paper our objective is to apply the above framework to a rice production system typical of northern Thailand as an example. LCA was used to generate environmental indicators for all processes and inputs involved in the production of seven crops typically grown on farms in the region. A bio-economic optimisation model was constructed for the farm system, with all activity options and input requirements over the course of one production period calculated on a per hectare basis and linked to the per hectare LCA indicators. Baseline profit maximising production and environmental outcomes were generated and, following the above framework, compared with two alternative scenarios. The first represents farm-system *adaptation*, by farmers, to reduce detrimental environmental impact (reduced greenhouse gas emissions); the second represents external *intervention*, by enforcing an alternative, 'environmentally friendly' farm input (alternative fertilisers and insecticides) farm plan. In both cases, we estimate the impact on other environmental indicators, including an indicator of human health: the use of some agricultural pesticides has been linked to health problems among farmers in Thailand. The paper is organised as follows. Section 2 considers the wider environmental impacts of rice production; Section 3 describes the data and the two (LCA, bio-economic model) analysis tools. Results from the two scenarios are presented in Section 4 and in Section 5 we discuss the main findings and consider the extent to which the approach addresses current concerns about the sustainability of agriculture in Thailand. Section 6 concludes.

2. Environmental Impacts of Rice Production

Although declining, rice continues to be an important source of energy for humans: in 2009, in Asia alone, 28% of calories in consumer diets derived from rice (Reardon and Timmer, 2014). Rice is also a major source of anthropogenic methane. Global emissions from the microbial decomposition of organic matter in anaerobic conditions in flooded lowland paddy fields account for *circa* 20% of total emissions from all anthropogenic sources (Neue, 1997; IPCC, 2006). Nitrate losses from rice paddy in Thailand across a four-month cropping season have been estimated at between 3.6 kg nitrate-N per ha (Pathak et al., 2004) and 8.0 kg nitrate-N per ha (Asadi et al., 2002). A range of pesticides used in Thai agriculture play a role in causing illnesses of farmers as well as environmental contamination. Thai farmers have shown acute symptoms related to organophosphate pesticide exposure such as muscle spasm and weakness, respiratory difficulty, nausea and chest pain (Norkaew et al., 2010, Taneepanichskul et al., 2010). There also appears to be a potential risk of long term pesticide exposure: Siriwong et al. (2008) found residual levels of organochlorine pesticide in freshwater, aquatic organisms and sediment collected in an agricultural area of central Thailand. The risk of cancer in fishermen in this region correlated positively with exposure to organochlorine pesticides in water bodies (Siriwong et al., 2009).

LCA assessments of rice production have been made in a number of geographical locations, including Italy, China and Japan (e.g. Blengini and Busto, 2009, Wang et al., 2010 and Hayashi, 2011). Most studies have focused on greenhouse gas (GHG) emissions and global warming potential, but without considering other potential impacts or the farm system more generally. Yossapol and Nadsataporn (2008) cite a figure of 2,908 kg CO_2 equivalent per ha of GHGs emitted from rice production in the north-eastern region of Thailand; Pathak and Wassmann (2007) report a lower value of 2,252 kg CO_2 equivalent per ha for a 'continuous flooding' rice farm using urea as fertiliser and removing straw from fields to feed animals. Thanawong *et al.* (2014), assessing the 'eco-efficiency' of three rice production systems in the north-eastern region of Thailand, found that rain-fed systems generally showed lower environmental impacts per ha and per kg of paddy rice produced.

In these previous studies, the focus is on one, albeit dominant, crop. While this allows the effect of some interventions that affect production to be evaluated (for example, by changing the type or amount of fertilisers used and re-running the LCA) it does not capture farm system adaptions, nor the factors that a farmer has to consider when making decisions about such adaptations – most particularly, the limits imposed by the farm system itself and availability of credit. We therefore develop an approach that allows these system level effects to be evaluated.

3. Materials and Methods

Rice-based farming systems

Lowland rice production in northern Thailand requires a large amount of water and the production season normally starts with the beginning of the rainy season, in June-July. Rice production in this period is known as ‘in-season’ or ‘rain-fed’ rice. Time to maturity depends on the cultivar; however, it generally takes up to 5-6 months before rice is ready to be harvested. After harvesting, at the end of the rainy season (October-November), farmers usually choose crops with lower water requirements, mainly soybean and shallot; these take around three months to grow before they are harvested. There is then a more diverse third three-month season of non-rice crops, normally drawn from maize, soybean, garlic, peanut, mungbean and shallot, before rice is re-established at the beginning of the next rainy season. Water is stored and available for irrigation through a network of irrigation ponds.

LCA framework

A standard LCA framework consists of four main stages: goal and scope definition, inventory analysis, impact assessment and interpretation. Here, the aim of the LCA was to quantify per hectare environmental impacts associated with each of the seven crops within the farm system described above; results were then incorporated into the bio-economic model, again on a per hectare basis. With the exception of buildings (sheds and storehouses), the system scope for the LCA includes all the associated processes and inputs from land preparation to harvesting (‘cradle-to-the-farm-gate’) for each crop. Buildings were excluded - their lifetime on farms in Thailand can be very long and adequate data were not available. Figure 1 illustrates the system boundaries for the LCA.

An inventory analysis is essentially a collection of data on resource and input utilisation, energy consumption and environmental impacts that are directly related to each process within the boundaries of the farm system. Post-harvest processes (e.g. storing, drying, and husking) were excluded as being out of scope: these processes are usually located outside the farms and owned by different parties. All farm machinery associated with crop production and harvesting was included in the inventory, as were transportation of variable inputs (i.e. fertilisers and crop protection products, the latter subsequently termed 'pesticides') to the farm. Data were sourced from regional surveys and interviews conducted by government agencies and from relevant literature (Table 1). The amount of machinery used in terms of kg of machine required for a specific process was based on the weight, the operation time and the lifetime of the machine. Farm inputs were assumed to be transported 5 km, from local retailer to the farm. Other data, including production of fertilisers, crop protection products, farm machinery, fuel and transportation were taken from the 'Ecoinvent' database that accompanies the SimaPro 7.3 software.

Data relevant to direct field losses and emissions were derived from published field experiments for the northern region of Thailand, or, where region-specific data were not available, for the country as a whole. Where Thai-specific data were not available, GHG estimates were calculated using Intergovernmental Panel on Climate Change (IPCC, 2006) methodology. In the case of phosphate loss, contamination from pesticides and ammonia emissions, appropriate estimates were calculated using formulae in Nemecek and Schnetzer (2011) and regional survey data (i.e. quantity and type of fertilisers and pesticides used, Table 1). These were varied under the alternative input scenarios described below. The complete inventory data are shown in Tables 2 and 3.

Following Haas et al. (2000), inventory data were used to generate seven environmental impacts, as shown in Table 4. These encompass Abiotic Depletion (ADP), Global Warming (GWP100), Human Toxicity (HTP), Freshwater Eco-toxicity (FAETP), Terrestrial Eco-toxicity (TETP), Eutrophication (EP) and Acidification (AP) Potentials. GWP100 is global warming potential over 100 years, as calculated from the three main greenhouse gases, at their respective carbon dioxide equivalents. The methodology of the impact assessment was based on CML2001, established and developed by the Centre of Environmental Science, Leiden University (CML, Guinée, 2002) and embedded in the Simapro 7.3 software. To ensure that all impacts could be used in the bio-economic model, a functional unit of one hectare was employed.

The bio-economic model

The bio-economic model that we employ here is a linear programming optimisation model. This type of model has three core components: the financial net benefits of growing each crop (the gross margins); the land, labour and capital constraints that limit production; and the technical coefficients, such as litres per hectare required to irrigate a crop at an expected yield, that determine how much of the resource constraints are used for different combinations of crops; in the case here, over three seasons within a year. By optimal, we mean that the solution is the most profitable achievable, in the short run: fixed resources cannot change in the model. As we have accepted that prices do not represent true opportunity costs of production, we do not claim that the solution is socially optimal. However, from this maximum farm level profit solution, we can calculate the cost of change towards set environmental objectives. Where variable inputs were a linear function of crop area, 'gross margins' (value of output less variable costs of production), were calculated per hectare of each crop. Variable costs were inclusive of seed, fertiliser and pesticide costs, and where they varied directly with changes in crop area, fuel, hired labour and machinery costs. By maintaining the per hectare link, we were also able to directly link the LCA results to the bio-economic model. A summary of farm socio-economic data used in the construction variables and constraints in the bio-economic model can be found in Table 5; Table 6 gives the individual crop gross margins and their components. Although the objective function was specified as maximising the Total Gross Margin (TGM), with fixed resources, we can think of changes in TGM as a short run measure of changes in farm profit.

Constraints were set using data from Thai government agency reports coupled with other related literature as given in Table 1 and Table 5. The main limits on production are land, family labour time, water and financial capital during different periods of the year. Capital is the effective farm system limit on hired labour and machinery, as well as purchase of variable inputs for the next season's cropping. We assume a typical situation, where the farmer has long term liabilities in the form of a 15 year loan provided by the Bank of Agriculture and Agricultural Cooperatives. The initial capital position of the farmer was set at Thai Baht (THB) 28,500 and short term borrowing through the year was allowed, limited to a maximum of THB 50,000 per year, at an annual rate of 7%. Volume of irrigation ponds in Thailand varies considerably (Setboonsarng and Edwards, 1998); it was assumed that a 10,000 m^3 pond, with pumping equipment, was adjacent to the farm, with 20% of water lost through evaporation and seepage. Available water in each season was also constrained by rainfall. Transfer activities allowed crops in season 2 and 3 to draw on cash generated in season 1 (and season 2 for crops in season 3) and unused water, subject to the rainfall and pond constraints.

The most problematic data were the technical coefficients indicating the efficiency of use of labour and machinery, both for the farm family and for hired labour and machinery. Typical labour use values were available from OAE (2011b) and NSO (2010). For machinery, work-rates (hours required per hectare for each operation, from planting to harvest) were calculated from datasheets provided by Thai agricultural machinery suppliers using conversion rates given in Lander (2000). However, we recognise that there will be considerable variation in technical efficiency among farms. These work-rates were also used to calculate fuel use, both in the LCA and the bio-economic model.

The full model allows for different combinations of crops and inputs, subject to constraints, assuming fixed technical coefficients for conversion of inputs into outputs. An initial run was used to establish the optimal farm plan and associated environmental impact (the baseline scenario); this baseline run was also subjected to a sensitivity analysis of variables and constraints that were key components of the optimal baseline solution. The Model was constructed using the 'Premium Solver Platform' running on Microsoft Excel™.

Additional criteria for the alternative scenarios

Two alternative scenarios were assessed: GHG minimisation and use of alternative farm inputs. The former represents a case where farmers are free to choose the best plan (from an economic perspective) to meet a specific environmental goal; the latter represents the situation where external agents, for example through a government extension programme, intervene and recommend (or dictate) that farmers make targeted changes to their farm systems. For the GHG minimising scenarios, we establish optimal emissions-minimising combinations of crops and inputs that achieve target levels of profit. Thus, the objective function of the bio-economic model is changed to minimisation of the environmental indicator for a given level of overall farm profitability. Relative to the baseline run profit, emissions are reduced in a way that meets each target profit level. Thus, under these alternative scenarios, minimal private cost is incurred in the form of profit forgone, while the environmental objective is achieved. The changes in farm plan for each profit target can be interpreted as the optimal adaptation path for a farmer with complete knowledge of his or her farm system, but with no knowledge of alternative production methods. The target level of profit was reduced by 10, 30, and 50%, respectively, from the baseline (profit maximising) plan and the effect on the other LCA-derived indicators recorded. An additional constraint, to grow rice to at least 2.0 ha, was imposed to ensure that a minimum amount of rice was available to the farmer for household consumption.

The alternative inputs scenario represents an external intervention that aims to reduce the negative environmental impacts associated with the farm system. From the LCA results, the application of urea as N-fertiliser was one of the major sources of direct ammonia emissions contributing to the acidification and eutrophication impacts. It is estimated that 10-25% of urea applied can be lost through volatilisation in general crop production; however, in rice paddy fields, the high pH of flood water can lead to up to 50% of broadcast urea being lost (Lægreid et al., 1999). In addition to ammonia emissions, the LCA analysis showed that manufacture of urea was the largest contributor to abiotic depletion. As an alternative, ammonium sulphate (AMS) fertiliser, at 21% nitrogen content, was introduced for rain-fed rice in the new scenario; the ratio of replacement is thus urea 1: AMS 2. The emission factor of ammonia to air per kg nitrogen for ammonium sulphate, as indicated in Nemecek and Schnetzer (2011), is 8% (urea is 15%). Solid dried poultry manure was also introduced as a fertiliser, with nutrient contents of 4.6% nitrogen, 3.3% phosphate and 2.5% of potassium oxide. Fertiliser quantities for each crop were adjusted to provide the same amount of available nitrogen as supplied under the baseline run. Assumptions regarding transportation and application method were the same as for manufactured fertilisers; ammonia losses associated with the use of organic fertiliser were taken from the Agrammon model (Agrammon Group, 2009); other emissions were generated from the Ecoinvent database. In addition to fertilisers, pesticides used for rice protection play significant roles in causing terrestrial and freshwater aquatic ecotoxicity. Cypermethrin is a pyrethroid insecticide used to control insect pests such as plant hoppers, worms, moths, aphids and weevils. However, due to its high toxicity to the environment, the use of cypermethrin has been restricted or prohibited in some countries such as India, Vietnam and the UK (Shardlow, 2006, MARD, 2012, and CIBRC, 2014). More recently, in 2011, the Minister of Agriculture of Thailand, in collaboration with the International Rice Research Institute, has launched a campaign to reduce use of cypermethrin insecticide in rice (Soitong and Escalada, 2011).

Therefore, fipronil (a phenylpyrazole compound) was substituted for cypermethrin; it has similar properties, but has been shown to be less toxic to the environment (DOAE, 2011).

4. Results

Results of the LCA for a functional unit of one hectare of crop production are shown in Figure 2. Crops vary considerably in impact across the indicators. Shallot production has a relatively high impact on abiotic depletion, acidification, eutrophication, human toxicity and freshwater aquatic ecotoxicity. As expected, rice is a key contributor to global warming; the terrestrial ecotoxicity is also high. Impact on human toxicity for rice is relatively low. Leguminous crops i.e. soybean, mungbean and peanut have lower impacts compared with other crops as they require less toxic pesticides and lower levels of fertiliser. Mungbean contributes the lowest impact in all categories. The results also show that higher gross margin crops such as rice, shallot and garlic (Table 6) tend to have a higher environmental impact per hectare; generally this is because they require more farm inputs (particularly fertiliser, hours of machinery and fuel) per hectare of production.

The optimal baseline results (Table 7) generate a profit maximising farm plan of 3.9 ha of rain-fed rice in the rainy season (S1) followed by 1.2 ha of shallot in the second season (S2) and 1.9 ha of shallot in season three (S3); land was only fully utilised in the rainy season for rice production. This reflects the typical situation in the region where rain-fed rice is the only crop grown when capital and water are relatively abundant. TGM was THB 279,522 per year. In other seasons, capital, rather than land was the binding constraint, with a large proportion of capital used for hiring farm labour. Shallot was grown in the second and third seasons, due to its high gross margin per ha and low water use. However, shallot requires relatively high expenditure on inputs and the capital constraint, although partially relaxed by available capital transfers from the sale of the first season's rice, becomes a key limitation in the following seasons. Rainwater and thus recharge of pond capacity is also a binding constraint in the second season, as rainfall becomes more limited. To grow shallot on all the available land in the second and third seasons would require additional credit of THB 366,199 at the beginning of the cropping year, and an extra 983 m^3 of irrigation water; relaxing these constraints (assuming no additional cost) would lead to full use of available land across the three seasons and a *circa* 90% increase in profit (to THB 539,457 per year).

Environmental impacts for the baseline plan are shown in Table 8. Manufacturing processes for rice fertilisers had the largest impact on resource depletion, as these processes consume a relatively large amount of abiotic resources. Direct field emissions from paddy fields were the main contributors to global warming, acidification and eutrophication impacts. Of all GHGs emitted from paddy fields, methane (CH_4) is the main contributor to GWP: the impact of rain-fed rice alone accounted for 2,043 kg CO_2 equivalent per ha of the farm's annual emissions. The high level of ammonia (NH_3) emitted from N-fertiliser applied in the field contributes substantially to the acidification and eutrophication indicators. The impacts associated with toxicity (human toxicity, terrestrial and freshwater aquatic ecotoxicity) were predominantly a function of pesticide use in the field. Triazophos (an organophosphorus compound), used to control leaf miners in shallot production, was the main contributor to human toxicity impact; cypermethrin applied in rice fields contributed most to ecosystems toxicity.

Greenhouse gas minimising scenario

The optimal farm plan at the target level of THB 251,570 (P-1, 10% lower than the baseline) produced 3.1 ha of rain-fed rice in the first season, 1.1 ha of shallot in the following season and a combination of 1.0 ha of mungbean and 1.7 ha of shallot in the final cropping season (Table 7). P-1 generates a 13% reduction in GWP (Table 8) compared to the baseline plan, largely due to the reduction in rice production in the first season. As GHG emissions are reduced, other environmental impact indicators improved although there were differences in extent: for example, at P-3, (30% lower profit), terrestrial eco-toxicity falls by nearly 50%. However, at P-5 (50% reduction in profit, Table 8), the trade-off between profit and reduction in GHGs is close to 1:1 and this 1:1 ratio also holds for the other environmental indicators. At P-1, human toxicity is the least 'coupled' impact to GWP reduction: i.e. reducing GHGs reduces human toxicity less than other indicators. For example, at 10% reduction in profit, rice, shallots and mungbean are grown; all of which are associated with the use of organophosphorus compounds (Table 3).

Alternative inputs scenario

Compared to the baseline, this scenario leads to a small reduction in profit (6%, Table 8). As expected, there is little change in crop mix as the changes introduced are for fertiliser and pesticides only. However, in terms of environmental impacts, abiotic depletion, acidification and eutrophication are improved by 20%, 43% and 37%, respectively, in comparison to the baseline (Table 8), as a result of the reduction in urea used. Use of fipronil reduces freshwater aquatic (47%) and terrestrial (91%) ecotoxicity impacts; and human toxicity impact (14% reduction). The GWP100 indicator is reduced by approximately 7%. The use of alternative farm inputs has quite a substantial effect on indicators for water quality: freshwater ecotoxicity, eutrophication and acidification fall to between 50 and 60% of the baseline values. The biggest reduction is for terrestrial ecotoxicity.

Baseline sensitivity

Four additional scenarios were identified from the key binding constraints and optimal crop choices generated by the baseline model. These were: changes in financial capital availability, rainfall, rice yield and shallot yield. Sensitivity was tested by varying the baseline default values by 20% up or down (hi- and lo-scenarios). As illustrated in Figure 3, the results show different patterns of percentage change in the total gross margin and environmental impacts responding to changes in the variable coefficients of interest. Farm profit responds strongly to variation of shallot yield as profit is reliant on the production of shallot in the second and third seasons. The increase of rice yield has a relatively large effect on the environmental indicators since more capital is transferred to the second and third season leading to increased production of shallot, a high environmental impact crop. In contrast, when the yield of shallot is reduced, model results show that garlic becomes more profitable with 1.6 ha grown in the third season instead of shallot. This reduces the impacts caused by shallot by approximately 10-18% (with the exception of TETP).

5. Discussion

While previous studies have focused on the environmental impacts from rice production, these have frequently failed to consider the combined farm-environmental system impacts across the farm system. Our integrated bio-economic and LCA approach addresses this criticism and is therefore more useful for both policy design and on-farm knowledge exchange practices. From our analysis, direct emissions from rice fields contributed to a number of environmental impact categories (acidification, eutrophication and global warming) while urea fertiliser production showed the highest impact on abiotic depletion. Terrestrial and freshwater ecotoxicity were dominated by pesticide use in rice production; however, the main source of human toxicity came from pesticide use in the production of shallots. Relative to the baseline run, minimising GHGs as an objective consistently reduced other environmental impacts, particularly terrestrial ecotoxicity. In contrast to other studies (for example, Gibbons *et al*, 2005) there is little evidence of an initial 'flat response' i.e. relatively large environmental gain at small financial cost. In part this is because the GHG minimising runs deliberately reflect the cost of achieving emissions' reduction with limited farmer adaptation i.e., the model allows adjustments to the existing farm system inputs and outputs but does not allow for new interventions. The main adaptation is the introduction of mungbean into season 3 (Table 7). As a legume, mungbean has a relatively low requirement for nitrogen (Table 2) and hence a lower global warming potential (Table 3) than other crops. It is however notable that the variance of mungbean output is relatively high (OAE, 2011a) and this risk – or indeed risk from growing any of the crops - is not captured by the model.

When new interventions are allowed, under the 'alternative input' run, global warming potential increases marginally (Table 8) but there are substantial reductions in acidification, eutrophication, freshwater ecotoxity; and particularly, terrestrial ecotoxicity. The trade-off effect on profit is small and less than 10%. The interventions are relatively straightforward and none have high capital requirements. The low cost extends to their 'trialability' (i.e. they are relatively easy for farmers to test and learn about before adoption, Pannell et al., 2006). In the case of organic fertilisers some caveats are needed: the application of such fertilisers on rice fields has been correlated to an increase in CH_4 emissions (Pathak and Wassmann, 2007; Wassmann and Pathak, 2007; Khosa et al., 2010). In the context of Thailand, however, a field experiment conducted by Sampanpanich (2012) showed that the addition of organic fertiliser on paddy fields reduced GHG emissions by 25-30%. Site specific variability of this kind adds weight to the argument that more site-specific data is needed to more realistically represent the individual farm situation. This also applies to the financial and physical data used to construct the farm level model: individual farms will vary considerably for factors such as yields and variable input use. We have not tested the impact of other interventions for example, policy mechanisms designed to encourage a more ecological approach to farming in Thailand. One Thai study that also focuses on rice and input use is Stuart *et al*., 2017. The authors report that adopting integrated management practices led to an increase in net income on farms and a decrease in the use of high environmental impact inputs such as fertiliser - suggesting that changes in input use can have both economic and environmental benefits.

To further encourage uptake of practice change, farmers could be given LCA information (perhaps in modified form e.g. ‘high’, ‘moderate’, ‘low’) as a proxy for environmental cost, thereby allowing environmental consequences to be considered in decision making. However, it is notable that after GHGs, the indicator that falls least is human toxicity. Given the evidence of toxic effects on farmers in Thai agriculture (e.g. Norkaew et al., 2010), this indicator may warrant greater weight: neither the GHG-minimising nor alternative input scenarios have much effect and other interventions to reduce human toxicity impacts would need to be tested, in particular with respect to pesticide exposure in the long term (Siriwong et al., 2008). Knowledge exchange activities that highlight both the environmental *and* personal health benefits of more efficient use of inputs would lead to a greater uptake of more sustainable agricultural practices.

The conflict between bio-economic modelling results and what farmers are doing on the ground raises specific issues. There is no direct reason why Thai farmers would factor LCA-based indicators into their decision making. However, there may be reasons for low uptake of organic manures: availability, ease of spreading, access to suitable labour and equipment or uncertainty about the nutrient content of the manure are all potential candidates. Again, for extension-based approaches, knowledge exchange between farmers and extension agents is needed; in some cases this will mean that model-based recommendations are adjusted once this additional knowledge is included. More widely, the issue of uncertain prices and yields, and availability of credit and water, is not dealt within in the model and thus the optimal plans considered here may not be optimal from a risk management perspective, in particular with respect to reducing risk. An obvious extension of the work would therefore be to develop indicators of risk for the broader farm system.

The LCA used here does not consider wider ecosystem services from agriculture, most notably biodiversity and the impact of the production system on soil resources. There are also some technical problems relating to the integration of LCA approaches into the bio-economic model. This is relatively straightforward under our short run assumptions; however, longer run adaptation will involve changes in machinery levels and thus the embodied environmental impacts, for example GHGs, will change. In this scenario, emissions would have to be linked to the input, rather than the crop as we have done here.

Our analysis suggests that new interventions of the type discussed in the introduction can be introduced into northern Thai agriculture at relatively low cost with substantial environmental benefits. The question remains as to what policy options might be used to encourage adoption of these interventions. Where public net benefits are relatively large and private net benefits are either marginally positive or marginally negative, Pannell (2009) argues that some form of positive incentive may be appropriate. In the context here, this might be a subsidy to encourage Thai farmers to make greater use of ammonium sulphate. Where private net benefits are greater, use of publicly-funded extension services would be a more appropriate policy response. However, the majority of the environmental impacts captured in the LCA are the consequence of negative externalities (global warming potential, eco-toxicity, eutrophication and acidification) for which the appropriate policy response is a disincentive – a signal to farmers that they should change management practice to reduce the detrimental environmental outcome. As a more pragmatic alternative, model-derived physical indicators – such as those presented in Table 8 – can be used as signals to farmers as a means of driving behaviour change. Similar arguments have been made by other authors (e.g. Dahl, 2012).

6. Conclusions

The integration of bio-economic and LCA techniques allows a wide range of system changes to be evaluated both at economic and environmental levels. In this study we model the trade-off between achieving agricultural management objectives (profitability) and a range of environmental impacts associated with rice-cropping systems in northern Thailand. A farm-level model was constructed using existing regional survey data. The baseline optimal plan was driven by system constraints - rice is always grown in season 1 - and followed by the high gross margin crop shallot.

Of the two impact reducing scenarios considered, modelling adaptation led to the introduction of mungbean which had a moderate reduction effect on profitability and environmental impact, although in part these reductions in impact were achieved by reducing rice production, with obvious food security implications. Employing alternative farm inputs led to larger effects: introducing ammonium sulphate and dried poultry manure to replace urea and fipronil insecticide instead of cypermethrin, showed that most of the environmental indicators, but particularly acidification, eutrophication and eco-toxicity potential impacts, were reduced at the cost of a *circa* 6% reduction in profitability. In terms of policy implications, if we consider environmental impacts such as GHGs as 'negative externalities' i.e. costs to society that are not accounted for in (farmer) decision making, the theoretical next step is to introduce private impact costs, through some market-based mechanism based on 'polluter pays' principles. However, these inevitably lead to unproductive debates as to the level of price to be charged and are likely to be impractical in countries such as Thailand where small-scale farmers are seeking to make a living on relatively marginal lands. While government intervention in the form of economic incentives or agricultural extension may be suitable, an alternative as argued here is to provide indicators of the environmental outcomes of different management practices and interventions; indeed, this could form part of government extension programmes. If coupled with information on costs saved – and consequent benefits to profitability, as shown by Stuart et al. (2017), these indicators would have a greater effect on farmer behaviour.

More generally, we acknowledge that the model presented here represents only some elements of the underlying farm systems in northern Thailand. For the processes considered, the LCA component of the analysis comprehensively captures environmental impacts according to recognised standards. Further work is needed to fulfil the potential of the associated farm level model, both to capture variability of input and output data across farms and to achieve greater understanding of the nature and range of the impact mitigating farm management practices available to farmers in northern Thailand. Reliable socio-economic data need to be collected to fill data gaps so that models reflect a more realistic situation for a specific farm. In addition, although there are numerous sets of well-established Life Cycle Impact databases available, a majority of data here were taken from European country scenarios. Databases for Thailand and other countries need to be developed; this could be achieved through international knowledge and data exchange programmes. There is also a need for better field measurements of GHGs and other environmental impacts activities, particularly if we wish to understand the site specific effects of encouraging farmers – by whatever means – to reduce the impact of their decisions on the environment.

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Figure 1 System boundaries for the rice-based farming system

Figure 2 Environmental impacts per crop hectare. Impacts are quantified relative to reference substance units (equivalence units, ‘eq’) for each impact category (Sb = Antimony, SO_2 = Sulphur Dioxide, PO_4 = Phosphate, CO_2 = Carbon Dioxide, 1,4-DB = 1,4-Dicholrobenzene)

Figure 3 Environmental indicators at different levels of profit (TGM) in the GHG minimising scenario. In each case, P = Potential; GWP = Global Warming; ADP = Abiotic Depletion; AP = Acidification; EP = Eutrophication, HTP = Human Toxicity, FAETP = Freshwater Eco-toxicity, TETP = Terrestrial Eco-toxicity

Table 1 Data sources and references used for the bio-economic model (BEM) and LCA

Data element	Data used for	Source
Crop practice	BEM	OAE (2007, 2011a, and 2011b)
Crop protection	BEM, LCA	DOAE (2011)
Labour	BEM	OAE (2011b), NSO (2010) and ILO (2010)
Fertilisers	BEM, LCA	Department of Internal Trade (2011) and MOAC (2010)
Seeds	BEM, LCA	Rice Department (2010), DOA (2009) and DOAE (2001, 2008)
Machinery and farm operations	BEM, LCA	NSO (2010), Chamsing et al. (2006), and Soni et al. (2013)
Water and Irrigation	BEM, LCA	Royal Irrigation Department (2010, 2011) and Setboonsarng and Edwards (1998)
Methane and Nitrous Oxide emissions (to air)	LCA	IPCC (2006) and FAOSTAT (2011)
Ammonia and Nitrogen Oxide emissions (to air); PO_4 loss (to water)	LCA	Nemecek and Schnetzer (2011)
NO_3^- leaching to ground water	LCA	Pathak et al. (2004) and Asadi and Clemente (2003)
Emissions from fuel combustion	LCA	Nemecek and Kägi (2007)
Pesticide contamination	LCA	Nemecek and Schnetzer (2011)
Indirect emissions	LCA	Ecoinvent version 2 in SimaPro 7.3

Table 2 Farm input inventory data for the baseline scenario (per ha of crop)

Input parameter	Unit	RF rice[e]	Maize	Soybean	Mungbean	Peanut	Shallot	Garlic
Farm Operations								
- Tillage by 2 wheel drive power tiller	hr	14.60	6.25	6.25	6.25	6.25	17.70	17.70
- Tillage, ploughing by tractor	hr	1.72	1.72	1.72	1.72	1.72	7.40	7.40
- Spraying by knapsack power sprayer	hr	5.36	4.46	5.36	3.57	4.46	7.14	7.14
- Irrigating by irrigation pump	hr	1.75	16.96	18.06	10.27	18.00	14.85	13.06
- Harvesting by combined harvester [a]	hr	1.25	0	0	0	0	0	0
Fuels (for farm operations)								
- Diesel	kg	40.0	17.6	17.6	17.6	17.6	64.8	64.8
- Petrol	kg	3.8	21.7	23.3	13.4	22.9	20.0	17.9
Seeds	kg	63	31	60	35	80	1875	1250
Fertilisers								
- N (as urea)	kg	69.2	59.5	4.1	3.3	21.3	45.1	37.8
- N (as DAP [b])	kg	7.8	19.6	14.9	11.7	13.7	28.9	24.2
- P (as DAP)	kg	20.0	50.0	38.0	30.0	35.0	74.0	62.0
- K (as KCl)	kg	26.0	25.0	19.0	15.0	18.0	99.0	86.0
Pesticides [c]								
- Insecticides	gAI	638.0	689.1	450.0	300.0	600.0	769.2	619.2
- Fungicides	gAI	100.0	2392.0	495.0	140.0	682.5	802.5	988.8
- Herbicides	gAI	434.7	276.0	1716.0	1716.0	1471.5	1580.6	1580.6
Transportation [d]								
- Fertilisers	tkm	0.576	0.672	0.306	0.241	0.372	1.090	0.929
- Pesticides	tkm	0.006	0.017	0.013	0.011	0.013	0.016	0.016
Packaging (polypropylene sacks)								
- Seeds	g	100.8	49.6	96.0	56.0	128.0	3000.0	2000.0
- Fertilisers	g	184.3	215.2	97.8	77.3	118.9	349.0	297.3
- Pesticides	g	1.9	5.3	4.3	3.4	4.2	5.0	5.1

[a] Combine harvester used for harvesting rice only

[b] Di-ammonium Sulphate

[c] Quantities of pesticides are in grams of active ingredient (gAI)

[d] Transportation is in tonne-kilometres (tkm); the distance from the farm to the local retailer was assumed to be 5 km

[e] Rain-fed rice

Table 3 Emissions inventory for the baseline scenario (per ha of crop)

Emission inventory	Unit	RF rice[b]	Maize	Soybean	Mungbean	Peanut	Shallot	Garlic
Emissions to air								
- Methane (CH_4)	kg	52.58	-	-	-	-	-	-
- Nitrous Oxide (N_2O)	kg	1.60	1.64	0.40	0.31	0.73	1.54	1.29
- Nitrogen oxides (NO_x)	kg	0.34	0.35	0.08	0.07	0.15	0.32	0.27
- Ammonia (NH_3)	kg	10.69	9.71	1.21	0.96	3.74	7.92	6.64
Emissions to water								
- Nitrate (NO_3^-)	kg	3.16	2.69	0.31	0.23	0.73	2.41	1.80
- Phosphate (PO_4^-)	g	254	267	262	258	260	277	272
Emissions to soil [a]								
- 2,4-D	g	403.2	-	-	-	-	-	-
- Acetamide-anilide compounds	g	-	-	1440.0	1440.0	1440.0	1440.0	1608.7
- Atrazine	g	-	2000.0	-	-	-	-	-
- Benzimidazole compounds	g	100.0	-	-	140.0	-	240.0	-
- Bipyridylium compounds	g	-	276.0	276.0	276.0	-	-	-
- (Thio) Carbamate compounds	g	-	637.5	645.0	-	-	150.0	250.0
- Dithiocarbamate compounds	g	-	392.0	-	-	-	-	720.0
- Nitrile compounds	g	-	-	-	-	562.5	562.5	-
- Organophosphorus compounds	g	619.2	-	300.0	300.0	600.0	759.8	459.8
- Phenoxy compounds	g	31.5	-	-	-	31.5	-	-
- Pyretroid compounds	g	18.7	18.7	-	-	-	-	-
- Insecticides (unspecified)	g	-	-	-	-	-	-	150.0

[a] Following Nemecek and Schnetzer (2011), it was assumed that all pesticides end up as emissions to soil.

[b] Rain-fed rice

Table 4 Recommended impact categories and corresponding indicators considered in an agricultural LCA (Haas et al., 2000)

Impact Category	**Environmental indicator**
Depletion of abiotic resources	
- Energy	Utilisation of fossil fuels
- Minerals	Utilisation of mineral fertilisers
Global Warming Potential (GWP)	Emissions of Greenhouse gases
Human- and Eco-Toxicity	Application of hazardous chemicals
Eutrophication	Leaching of nutrients
Acidification	NH_3, NO_x and SO_2 emission

Table 5 Summary of key variables used in the Bio-economic model [a]

Detail	Value	Unit
Holding land area	3.9	ha
Members of the household	3.8	persons
Family labour (age 16-64)	2.8	persons
Outstanding debt at the end of the year [b]	86,899	baht
Average rainfall in the rainy season [c]	1037	mm
Average rainfall in the dry season [c]	148	mm

[a] Based on Office of Agricultural Economics (2011b)

[b] Including short-term and long-term loan schemes from the Bank of Agriculture and Agricultural Cooperatives and/or other sources

[c] The average amount of rainfall was obtained from the Royal Irrigation Department measured from Chiang Mai station from 1981-2010.

Table 6 Regional average economic and physical production values for each crop in the rice-based farming system (in 2010 values)

Crop	**Variable costs (baht/ha)**	**Yield (kg/ha)**	**Price (baht/kg)**	**Output (baht/ha)**	**Gross margin (baht/ha)**
Rain-fed rice	15,912	3,018	10.6	31,962	16,038
Maize	16,052	4,085	6.1	24,924	8,963
Soybean	14,258	1,564	13.7	21,362	7,203
Mungbean	8,267	776	20.7	16,035	7,649
Peanut	22,239	1,620	17.9	29,030	6,735
Shallot	105,051	11,394	16.5	187,611	81,269
Garlic	102,650	6,055	29.3	177,312	75,298

Office of Agricultural Economics (2011a and 2011b)

Table 7 Farm-level model optimal results for baseline, minimising GHGs and alternative inputs scenarios

Resource Input	Baseline			Minimising GHGs [a]				Alternative inputs [b]		
	S1	S2	S3	S1	S2	S3		S1	S2	S3
Optimal Crop	RFr	SH	SH	RFr	SH	SH	MB	RFr	SH	SH
Level of Activity (ha)	3.9	1.2	1.9	3.1	1.1	1.7	1.0	3.9	1.1	1.8
Crop product (kg)	11,768	13,715	21,957	9,350	12,870	19,285	776	11,768	13,091	20,890
Family labour (man-days)	288.6	198.1	185.6	229.3	198.1	185.6		288.6	198.1	185.6
Hired labour (man-days)	0.0	67.9	240.3	0.0	51.5	224.4		0.0	55.8	219.6
Machinery (hours)										
- Power tiller	56.9	20.6	32.9	45.2	19.3	35.2		56.9	19.6	31.3
- Tractor	6.7	8.9	14.3	5.33	8.4	14.2		6.7	8.5	13.6
- Harvester	4.9	0	0	3.9	0	0		4.9	0	0
Fertilisers (kg)										
- N fertiliser (Urea)	270	54	86	214	50	80		-	-	-
- N fertiliser (AMS)	-	-	-	-	-	-		205	31	51
- P fertiliser	78	89	141	62	81	156		14	63	103
- K fertiliser	101	119	188	81	110	183		53	95	155
- Organic fertiliser	-	-	-	-	-	-		1,950	550	900
Pesticides (THB [c])	4,253	2,425	3,839	3,141	2,223	4,450		5,277	2,223	3,688
Total water use (m^3)	22,230	3,226	5,164	17,662	3,072	6,436		22,230	2,957	4,892
Borrowing Credit [d] (THB)	45,320	4,680	0	29,779	20,221	0		50,000	0	0
Total Gross Margin [e]		**279,522**			**251,570**				**261,955**	

AMS = Ammonium sulphate, RFr = rain-fed rice, SH = shallot, MB = mungbean and S = season (S1, S2, S3 = first, second and third season)

[a] Greenhouse gases minimising scenario at 10% reduction profit maximising (baseline) level

[b] The alternative, i.e. poultry manure, ammonium sulphate fertiliser, and fipronil insecticide, are combined as one run

[c] Equivalency of currency unit: 1 USD = Thai Baht (THB) 32.5

[d] The borrowing credit allowance was set to be THB 50,000 based on a short-loan conditions defined by the Bank of Agriculture and Agricultural Cooperatives

[e] TGM is total farm output less total farm variable costs.

Table 8 Economic - environmental trade-offs at different levels of profit as measured by TGM: GHG minimisation and alternative input scenario

	Unit	Baseline	P-1	Impact	P-3	Impact	P-5	Impact	Alternative[a]	Impact
TGM	THB	279,522	251,570		195,665		139,761		261,955	
% TGM reduction		0%	10%		30%		50%		6%	
ADP	kg-Sb eq	36.3	32.9	9%	23.3	36%	17.7	51%	28.9	20%
AP	kg SO_2 eq	139.5	121.7	13%	84.2	40%	67.6	52%	79	43%
EP	kg PO_4 eq	51.5	46.5	10%	32.8	36%	25.3	51%	32.3	37%
GWP	kg CO_2 eq	12,455	10,894	13%	7,512	40%	6,324	49%	11,643	7%
HTP	kg 1,4-DB eq	7,175	6,724	6%	4,844	32%	3,523	51%	6,137	14%
FAETP	kg 1,4-DB eq	32,435	27,616	15%	18,925	42%	14,752	55%	17,031	47%
TETP	kg 1,4-DB eq	7,230	5,803	20%	3,780	48%	3,653	49%	642	91%
Profit per kg GHG	THB $kgCO_2eq^{-1}$	22.4	23.1		26.05		22.1		22.5	

[a] Alternative inputs i.e. poultry manure, ammonium sulphate fertiliser, and fipronil insecticide were combined as one run. Percentage impact figures are reduction in impact from the baseline values.

Key: Total Gross Margin (TGM); Abiotic Depletion (ADP); Global Warming (GWP100); Human Toxicity (HTP); Freshwater Eco-toxicity (FAETP); Terrestrial Eco-toxicity (TETP); Eutrophication (EP); Acidification Potentials (AP); Global Warming Potential (GWP); Thai Baht (THB); (Sb = Antimony (Sb); Sulphur Dioxide (SO_2); Phosphate (PO_4); Carbon Dioxide (CO_2); 1,4-Dicholrobenzene (1,4-DB)

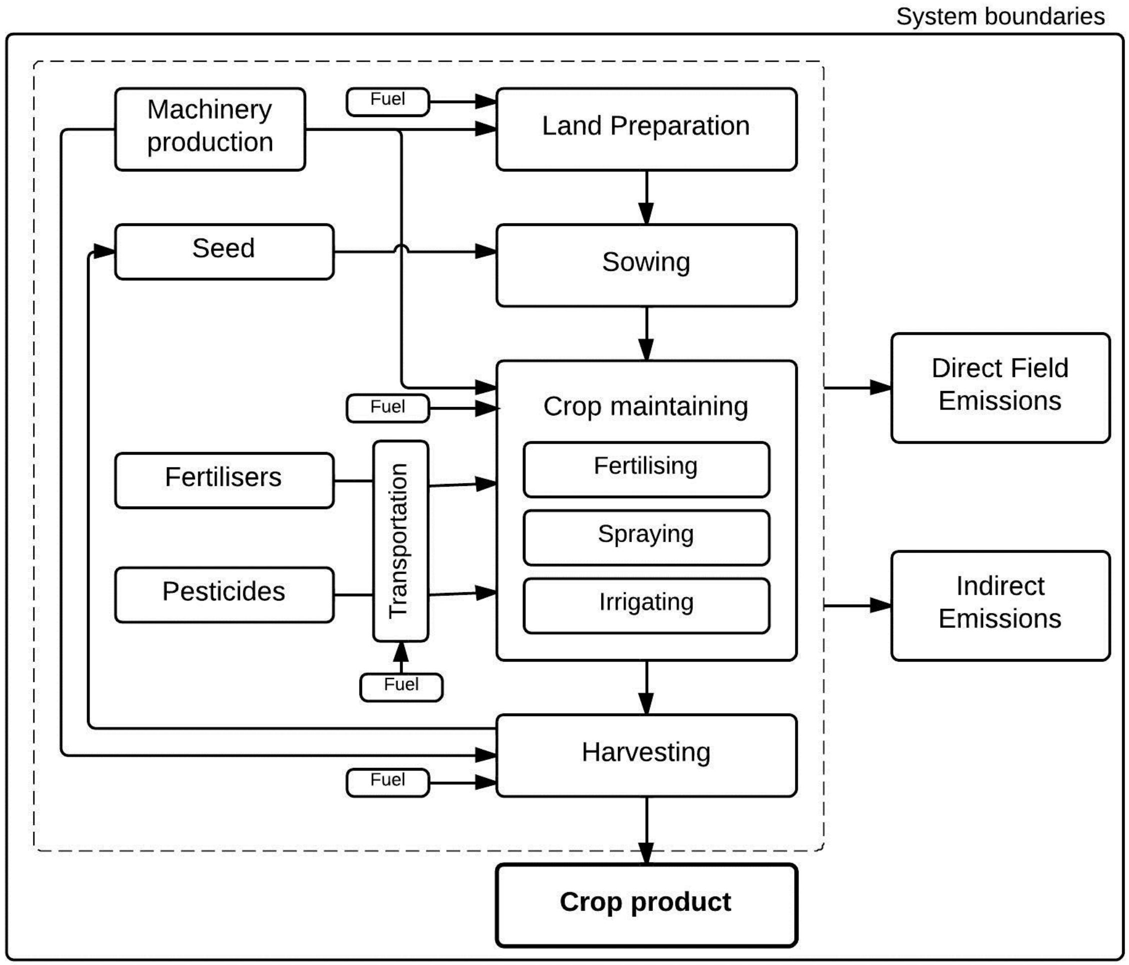

Figure 1 System boundaries for the rice-based farming system

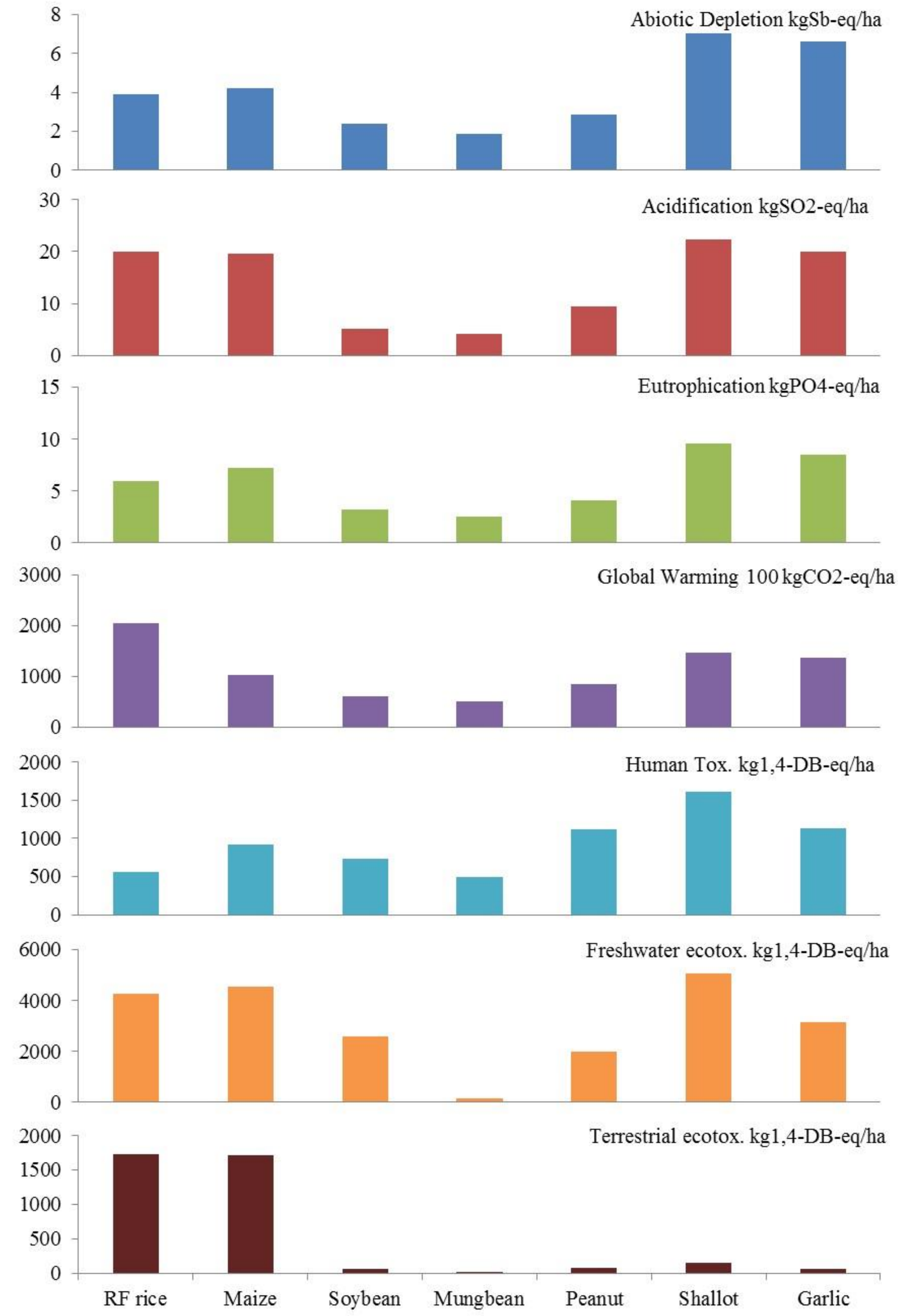

Figure 2 Environmental impacts per crop hectare. Impacts are quantified relative to reference substance units (equivalence units, 'eq') for each impact category (Sb = Antimony, SO_2 = Sulphur Dioxide, PO_4 = Phosphate, CO_2 = Carbon Dioxide, 1,4-DB = 1,4-Dicholrobenzene). RF rice = Rain-fed rice

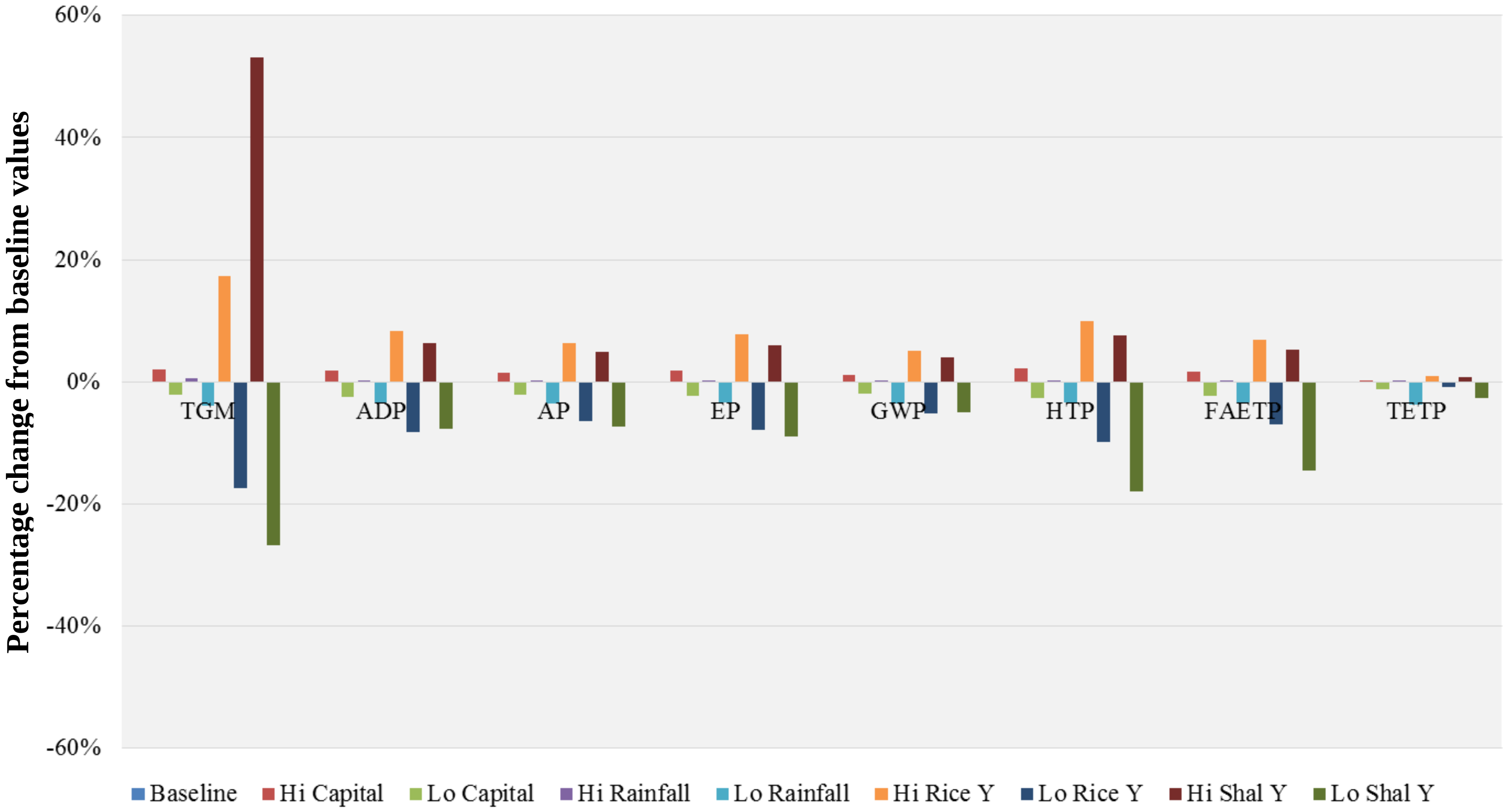

Figure 3 Percentage changes in profit as measured by TGM and environmental impacts responding to changes in the variable coefficients of interest. Key: Total Gross Margin (TGM); Abiotic Depletion (ADP); Global Warming (GWP100); Human Toxicity (HTP); Freshwater Eco-toxicity (FAETP); Terrestrial Eco-toxicity (TETP); Eutrophication (EP); Acidification Potentials (AP); Global Warming Potential (GWP); High Capital (Hi Capital); Low Capital (Lo Capital); High Rainfall (Hi Rainfall); Low Rainfall (Lo Rainfall); High Rice Yield (Hi Rice Y); Low Rice Yield (Lo Rice Y); High Shallot Yield (Hi Shal Y); Low Shallot Yield (Lo Shal Y).

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