# RICE – TEEBAGRIFOOD

[ FINAL REPORT ]

# Valuation of rice agro-systems

Food and Agriculture Organization of the United Nations

IRRI

TRUCOST

Bioversity International

The Economics of Ecosystems & Biodiversity

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This final project report builds on two interim reports, which complement the TEEBAgFood valuation analysis on rice:

1. **Bogdanski, A., Attwood, S., DeClerck, F., DeClerck, R., Hadi, B., Horgan, F., Rutsaert, P., Turmel, M.-S., R. van Dis, Gemmill-Herren, B.** (2015). Shedding light on the invisible links within eco-agri-food-systems'. A narrative review of literature on how agricultural management can influence the positive and negative impacts of rice farming. FAO, unpublished project report for The Economics of Ecosystems and Biodiversity (TEEB) global initiative for Agriculture and Food

2. **Van Dis, R., Bogdanski, A., Gemmill-Herren, B., Attwood, S., DeClerck, F., DeClerck, R., Hadi, B., Horgan, F., Rutsaert, P., Turmel, M.-S., Garibaldi, L.** (2015). Counting the costs and benefits of rice farming. A trade-off analysis among different types of agricultural management. FAO, unpublished project report for The Economics of Ecosystems and Biodiversity (TEEB) global initiative for Agriculture and Food.

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# Executive Summary

**Background**

The UNEP TEEB Office has recently begun to undertake a study on 'TEEB for Agriculture and Food'. This study is designed to provide a comprehensive economic evaluation of the 'eco-agri-food systems' complex, and to demonstrate that the economic environment in which farmers operate is distorted by **significant externalities**, both negative and positive, and a lack of market, policy and societal awareness and appreciation of human **dependency on natural capital**.

The Food and Agriculture Organization of the United Nations (FAO) together with its partners, the International Rice Research Institute and Bioversity International as well as Trucost has applied the TEEB approach to the rice farming sector. Rice (*Oryza sativa* from Asia *or Oryza glaberrima* from Africa) production is essential to the food security and livelihoods of around 140 million rice farming households and provides a range of ecosystem services beyond food production (i.e. cereal grain) alone. At the same time, rice production has been linked to a range of different environmental impacts such as high GHG emissions, air and water pollution as well as an increase in water consumption. Policy makers need to make decisions on how to manage and mitigate these impacts while providing affordable, nutritious, equitably accessible and safe food for a growing global population with changing patterns of consumption.

**Study objectives**

As these challenges are not independent, but rather interlinked, reaching them is likely to require **trade-offs**. The question of interest is therefore of how to reduce trade-offs between these different goals. Where possible, one should identify **synergies** that allow for a maximization of benefits, while minimizing costs to society and the environment, (i.e. negative externalities), and the wellbeing of the farmer him or herself through the degradation of natural capital from rice production. It is therefore crucial to know which types of farm management practices or systems offer the best options to reach these synergies, and reduce trade-offs.

The specific objectives of this study were three-fold:

1. To identify visible and invisible costs and benefits of rice agro-ecosystems; i.e. externalities

2. To identify and assess those rice management practises and systems which reduce trade-offs and increase synergies

3. To make these trade-offs and synergies visible by assigning biophysical or monetary values to the different options

**The approach**

*1. Scope and framework setting*

In a first step, five case study countries were selected which cover rice farming globally and which represent a gradient from low intensified to high intensified production systems. Countries selected were: the Philippines and Cambodia in Asia, Senegal in Africa, Costa Rica in Latin America and California/The United States in North America. According to FAOstat (2013), Cambodia was, on average, the lowest yielding country with 3.3 tons/ha and the USA had the highest yielding production with 9.5 tons/ha.

In a second step, a rice production system typology was developed. On a first level, rice systems were distinguished by rice growing environments. The three main categories were Irrigated Lowlands, Rainfed Lowlands and Rainfed Uplands.

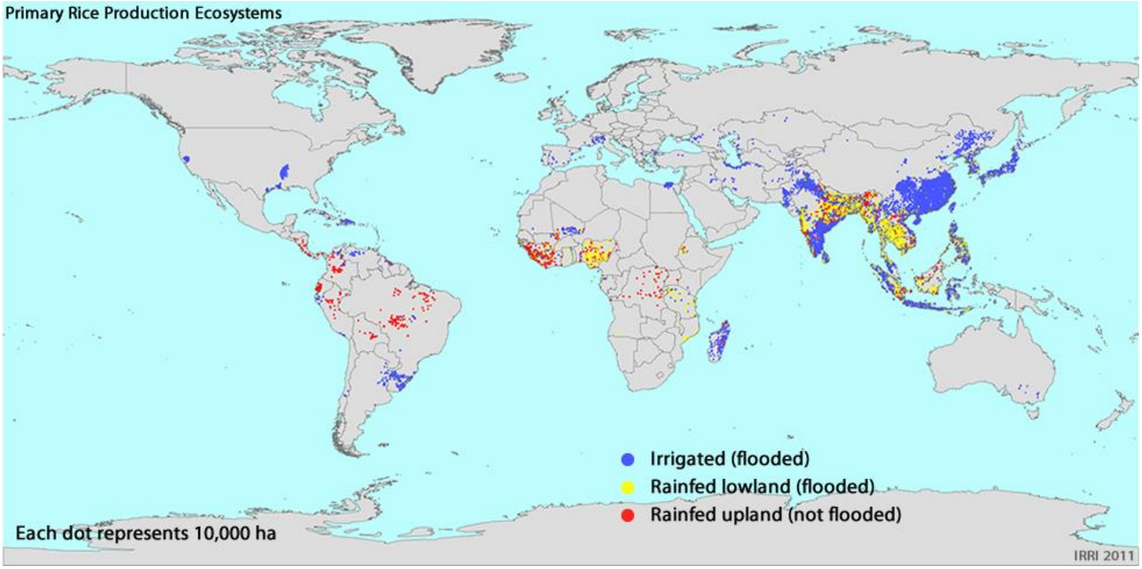


**Figure 0.1.** Map of different rice production systems globally, showing the considerable extent of irrigated rice (blue). Source: IRRI, 2009.

On a second level, the rice production systems were further categorized by rice management systems and practices. 28 different system and practice category comparisons were identified, starting with land preparation and finishing at harvest.

The study has set out to identify those **farm management practices** that offer the best options to reach synergies, and reduce trade-offs between different management objectives. Several scenarios, i.e. pairwise comparisons (table 1), were applied to show the effect of the various farm management practices on different environmental and/or agronomic variables:

1. The baseline scenario describes a conventional management approach, for instance herbicide use to combat weeds.

2. The alternative scenario describes a farm management practice that is expected to decrease an environmental impact or to increase an ecosystem service. For instance, instead of herbicide use, hand weeding or biological control could be practiced.

**Table 0.1.** Practice and system comparisons included in the study.

| Management practices (Scenarios) | | |
|---|---|---|
| 1. Preplanting | Land preparation | Dry tillage – puddling |
| | | Land levelling – no levelling |
| | | Minimum soil disturbance – conventional tillage |
| | | No tillage – conventional tillage |
| 2. Growth | Planting | Direct seeding – transplanting |
| | | Dry seeding – wet seeding |
| | Water management | Low irrigation frequency - high irrigation frequency |
| | | Improved water management - continuous flooding |
| | Soil fertility management | Reduced mineral fertilizer use - high mineral fertilizer application |
| | | No fertilizer use – mineral fertilizer application |
| | | Organic fertilizer application - mineral fertilizer application |
| | | Organic fertilizer application - no fertilizer application |

| | | |
|---|---|---|
| | | Mineral + organic fertilizer application – mineral fertilizer application only |
| | Weed management | No weed control - herbicide use |
| | | Biological weed control + hand weeding - herbicide use |
| | | Hand weeding – herbicide use |
| | | Reduced herbicide use – higher herbicide input |
| | Pest and disease management | No pesticide use - pesticide use |
| | | Reduced pesticide use – higher pesticide input |
| 3. Postproduction | Residue management | Winter flooding – no winter flooding |
| | | Straw incorporation – straw burning |
| | | Straw baling and removal – straw burning |
| | | Straw rolling – straw burning |
| **Management systems** | | |
| | | System of Rice Intesification – Conventional agriculture |
| | | Organic agriculture - Conventional agriculture |

In a third step, the project team identified pertinent policy and management issues related to the selected rice management systems and practices. These constituted the basis for the development of the analytical framework, which was built around a set of relevant costs and benefits related to rice production (see Table 0.2).

**Table 0.2**. Benefits and costs related to rice cultivation. Those with an * could not be covered due to data limitations.

| Benefits | Costs |
|---|---|
| **Rice grain (Revenue)** | Water pollution |
| **Rice straw (Nutrient value)** | Air pollution |
| **Rice husk (Energy value)** | Land pollution |
| **Pest control** | Water consumption |
| **Nutrient cycling and soil fertility** | GHG emissions |
| **Carbon storage*** | Labor |
| **Ecological resilience (pests)** | Fertilizer |
| **Recreational and tourism opportunities** | Pesticides |
| **Flood prevention*** | Fuel* |
| **Water recharge*** | Capital costs (e.g. machinery)* |
| **Habitat provisioning** | Irrigation water* |
| **Dietary diversity** | Seeds* |

*2. Biophysical quantification and monetary valuation*

TEEB-AF has unique challenges in developing a means of analysis of the positive and negative externalities of agriculture; negative externalities align well with standard valuations of environmental pollination, but positive externalities – such as ecological resilience, or dietary diversity – are not well captured by standard monetary valuations methods. In this first phase, so that the gaps and needs can be better understood, a conventional process was followed to attribute monetary values to the costs and benefits above (many of which then could not be analysed or compared). Thus this section presents the conventional process, with gaps described at a later point.

Placing monetary values on the costs or benefits that arise due to different management practices takes place in three distinct steps. This process is guided at all times by an overarching research question, which outlines the aim of the monetary valuation, why the valuation is needed, and who the target audience is.

The first step, which measures the changes in physical conditions, has been performed in the academic literature used for this study. This includes the identification of the drivers for change, such as fertiliser or pesticide inputs. Additional to extracting this data in a standardized way across all five case study countries, a vote counting analysis was done to synthesize these results.

The second step requires the biophysical modelling of the impact, or impacts, that are caused by changing physical conditions. This includes identifying factors such as the endpoint of nutrient run-off, which may be adjacent freshwater ecosystems for example, and quantifying the change in the biophysical indicator that is to be valued, such as the change in the quality of human health, measured in disability adjusted life years (DALYs) (see below for more details).

The final step involves the economic modelling component of the valuation. This includes the identification of the final recipient of the impact, such as the local populations who experience the negative effects of eutrophication, and then selecting an appropriate valuation technique to monetize the change in biophysical conditions.

In this study, the biophysical modelling assigns the costs and benefits of the impacts to either human health, or ecosystems, arising from different management practices. Human health is measured in terms of disability adjusted life years, or DALYs. This metric quantifies the burden of disease on human populations, and can be thought of as one year of healthy life lost. The measure includes both the years of life lost due to illness (mortality), and the years of healthy life lost due to disability (morbidity). The valuation approach uses a willingness-to-pay (WTP) survey, which elicits values from society based on changes in factors like reduced income due to ill health, the pain and discomfort caused, as well as decreased life expectancy.

The costs or benefits of the impacts on ecosystems are quantified in terms of the change in ecosystem functioning, and then valued in terms of the change in the monetary value of the ecosystem services provided. Ecosystem functioning is measured as the change in net primary production (NPP) within ecosystems outside of the farm gate. Currently, impacts on the farm have not been considered. The monetary valuation approach involves conducting a meta-analysis of primary valuation studies of provisioning, regulating, and cultural ecosystem services. The approach allows the quantification and valuation of ecosystem services that are impacted due to changes in environmental quality. This can be due to the emission of air land and water pollutants, or to changes in water availability. Provisioning ecosystem services, such as rice and rice husk production, coming from within the farm gate have been valued using direct market pricing.

*3. Scenario analysis*

In the last step, we upscale management practices from field to country level. All results – costs and benefits – are given on a per hectare basis. Knowing the rice farming area in each country and the percentage of irrigated lowlands, rainfed lowlands and rainfed upland systems, one can calculate the production area in each rice growing environment. Multiplying this area by the difference in impact between two management practices, one can calculate the gains, losses or savings related to an environmental impact or ecosystem service when changing from one scenario to the other.

**Scenario analysis:**
**SRI versus conventional management**

The System of Rice Intensification (SRI) includes intermittent flooding as part of the production package. The system advises transplanting of young (eight to ten days old) single rice seedlings, with care and spacing, and applying intermittent irrigation and drainage to maintain soil aeration. In addition, the use of a mechanical rotary hoe or weeder to aerate the soil and control weeds is encouraged.

If Senegal was to change all its irrigated lowland systems from conventional management to SRI, the society would save about US$ 11 million in water consumption related health and environmental costs. At the same time, the rice producer community would gain a total of US$17 million through yield increases – a clear synergy.

If the Philippines were to change all their rainfed lowland systems from conventional management to SRI, the rice producer community would gain a total of US$750 million through yield increases. Data on water consumption was n

If Cambodia was to
rainfed lowland s
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SRI, the rice
community would g
US$801 million th
increases. No irrig
consumption costs
this farming syst
dependent on rainfa

While extrapolatin
from a few studies
entire country may
general trends, one
cautious about th
each study. Yield i
SRI are highly
mainly occur
weathered soils, whereas in ideal rice soils yields tend to be the same or less with SRI (Turmel et al. 2010).

**Results and discussion**

As this study has been designed to be a trade-off analysis, the results have been structured according to the effect of different management practices on two contrasting or synergistic ecosystem benefits or costs. The assumptions that underpin the analysis refer to rice production, on the one hand, and a range of different externalities, i.e. an environmental impact or ecosystem service, on the other, to show potential trade-offs or synergies between the two. Two examples are given below:

1. *Increasing rice yields versus reducing water consumption*

Worldwide, about 80 million hectares of irrigated lowland rice provide 75% of the world's rice production. This predominant type of rice system receives about 40% of the world's total irrigation water and 30% of the world's developed freshwater resources. The dependence on water of the rice farming sector is a huge challenge as freshwater resources are becoming increasingly depleted due to competing water uses from the residential and industrial sector and as rainfall is increasingly erratic due to climate change and variability. More efficient water use is therefore a must, yet it carries a number of trade-offs as this study has shown.

This study sought to assess and valuate trade-offs resulting from irrigation management, soil preparation and crop establishment on rice yields, on the one hand, and water consumption, on the other. The study analyzed the change in yield and water consumption under continuous flooding, alternate wetting and drying (AWD), during aerobic soils production and the system of rice intensification (SRI). The study further compared dry tillage to puddling, and direct seeding to the transplanting of seedlings. Figure 0.2 shows the effects of SRI and conventional management on irrigated (IL) and rainfed lowland (RL) system in Senegal, Cambodia and the Philippines based on data from Krupnik et al (2010), Krupnik et al (2012a), Krupnik et al (2012b), Miyazato et al (2010) Dumas-Johansen (2009), Koma (2002), Ly et al (2012), Ly et al (2013) and Satyanarayana et al (2007).

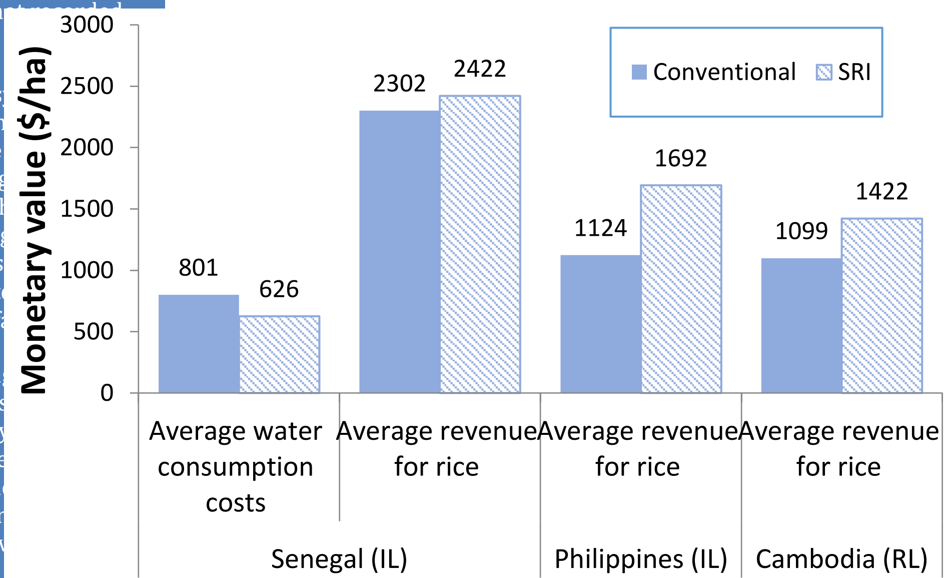


**Figure 0.2.** Comparison of the effects of conventional management and SRI on the revenue and environmental and health costs of water consumption per hectare in irrigated lowland systems (IL) and rainfed lowland systems (RL).

2. *Increasing rice yields versus reducing GHG emissions*

Global estimates attribute about 89 percent of rice global warming potential to $CH_4$ emissions which are due to flooding practices in irrigated and rainfed lowland systems (RL) (Linquist et al, 2006). To a much smaller degree, the production and application of N-fertilizers contributes to the rice global warming potential. And also emissions from rice straw burning impact global climate change. In addition to rice production being a major emitter of GHGs, rice systems also sequester carbon via soil organic carbon. Yet overall, rice production is a net producer of greenhouse gas emissions.

This study sought to assess and monetize the trade-offs resulting from irrigation water management, residue management, fertilizer application and the choice of rice varieties on rice yields, on the one hand, and GHG emissions, on the other. The value of rice production was estimated on the basis of the country specific revenue for rice grain received per ton of paddy rice. Primary data on GHG emissions as reported in the peer reviewed studies was used to model the GHG emission costs. The cost of GHG emissions were valued following the Trucost Greenhouse Gas methodology which provides a valuation coefficient for $CO_2$ equivalent emissions based on the social cost of carbon emissions.

Figure 0.3 shows the effects of SRI and conventional management on the revenue for rice grain of rice and GHG emission costs in RL systems in Cambodia.

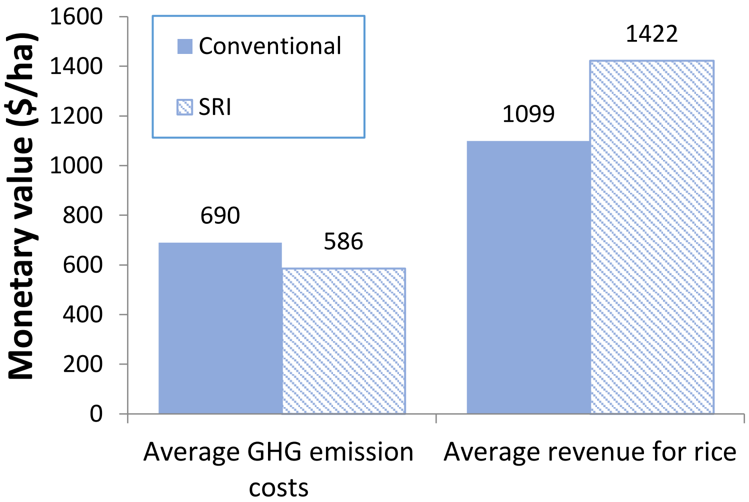


**Figure 0.3.** Comparison of the effects of conventional management and SRI on the revenue of rice production and social costs of carbon emissions per hectare in rainfed lowland systems.

**Conclusions**

**<u>Scenario analysis:</u>**
**SRI versus conventional management**

While the concept of SRI was originally developed under irrigated conditions, these systems have also been adapted to rainfed lowland (RL) paddies. The SRI in RL systems differ from the conventional management system in several parameters, but the focus of included research studies is on modified water and nutrient management. In these studies, SRI fields are moist during transplanting and drained several times during the growing season. Trade-offs are likely to occur between $CH_4$ emissions when the fields are flooded and $N_2O$ emissions when fields are drained.

Data from Dumas-Johansen (2009), Koma (2002), Ly et al (2012), Ly et al (2013) and Satyanarayana et al (2007) collected in RL systems in Cambodia led to a value of rice production of US$1099 per hectare when conventional management was practiced and US$1422 when SRI was implemented.

The monetary valuation for GHG emissions in Cambodia's RL paddies resulted in an average cost of US$690 per hectare of rice production for conventionally managed systems and US$586 for SRI – a reduction in costs of 15%.

If all rice farmers in RL systems in Cambodia would change to SRI, they would increase the revenue of rice by US$ 801 million. At the same time, society would have to spend US$ 258 million less in GHG emission costs.

The results show that the development of a solid typology that is further disaggregated into specific farming systems and practices is key to valuing externalities from the agriculture and food sector. Farming is very diverse, and so are the environmental impacts and ecosystem services that are linked to each type of production. Typologies therefore need to zoom in on management practices and systems as much as possible to reflect the reality of (rice) farming and the diversity of its values. It would be illusionary to think that there is ONE type of producton that leads to ONE specific set of positive and negative externalities.

The study results further confirm that a trade-off analysis is mandatory if the study is to inform policy. Focusing on environmental impacts or ecosystem services alone without considering the impacts on food production, for example, would fail to provide a sound basis for decision making. One therefore needs to value all potential benefits and costs at the same time, providing a holistic assessment of a farming system that is truly multifunctional.

This requires that experimental studies provide a comprehensive data set that goes beyond food production alone as is typically done in agronomic studies. Likewise, ecological and environmental studies need to record agronomic values, including yields, and widen their often restricted focus on natural resources and biodiversity alone. Furthermore, there is a need to enhance models that can mimic agro-ecological processes where specific data points are missing, and where field studies are not feasible.

Alternatively, farmers themselves are carrying out just such experiments, varying their practices to attain multiple benefits. Instead of relying on the scientific data alone, where experimental protocols generally require that most aspects are held constant while one or a few variable are manipulated, there may be large scope for applying a TEEB-type analysis to specific farms, and making greater use of on-farm, farmer-led research.

There is also a need to improve current valuation methodologies, as there is a clear lack of those that can value agroecosystem benefits as opposed to costs, as noted above. There is a need to link economic valuations to market costs, and avoided costs for the farmer. Methods are urgently needed to be able to assess and compare multi-dimensional values, as monetary analysis is not appropriate for all positive and negative externalities of agriculture. Furthermore, one needs to better adapt current models for valuation to the realities of developing countries.

Recognising that national assets extend well beyond GDP, or gross domestic product, there is an initiative underway to bring in methods to account for other forms of capital including natural capital, to national statistical accounts, through the UN initiative on Systems. Of Environmental-Economic Accounting. TEEB-AF, in addressing the current challenges to develop multi-dimensional valuation, also may provide and share important insights with the System of Environmental-Economic Accounting for Agriculture (SEEA-AGRI). While ecosystem valuations usually focus on the local level, ecosystem accounting methods aim to aggregate information to produce statistical results at the national level. Since both areas of expertise are still in its infancy, it is timely to join forces now in order to follow a coherent approach in the future.

# Acronyms

| | |
|---|---|
| Alternative wetting systems | AWD |
| Carbon Dioxide | $CO_2$ |
| Dry Direct Seeding | DDS |
| Food and Agriculture Organization of the United Nations | FAO |
| Global Warming Potential | GWP |
| Government of Senegal | GoS |
| Grande Aménagement | GA |
| Grand Offensive for Food and Abundance | GOANA |
| Green House Gases | GHG |
| Ground Cover Rice Production System | GCRPS |
| Integrated Pest Management | IPM |
| Integrated Production and Pest Management Programme | IPPM |
| Iron | Fe |
| Irrigated Rice Research Consortium | IRRC |
| Manganese | Mn |
| Methane | $CH_4$ |
| Moderate Resolution Imaging Spectroradiometer | MODIS |
| National Irrigation Administration | NIA |
| Nitrogen | N |
| Nitrous Oxide | $N_2O$ |
| Panicle initiation | PI |
| Périmètre Irrigué Privé | PIP |
| Périmètre Irrigué Villageois | PIV |
| Pesticide Impact Rating Index | PIRI |
| Phosphorous | P |
| Potassium | K |
| Recommended Management Practices | RMP |
| Rice Root Knot Nematode | RRKN |
| Saturated Soil Culture | SSC |
| Senegal River Valley | SRV |
| Site Specific Nutrient Management | SSNM |
| Soil Organic Carbon | SOC |
| Soil Organic Matter | SOM |
| Sub Saharan Africa | SSA |
| Sulphur | S |
| System of Rice Intensification | SRI |
| The Economics of Ecosystems and Biodiversity | TEEB |
| Urea Deep Placement | UDP |
| Water Use Efficiency | WUE |
| Wet Direct Seeding | WDS |
| Zinc | Zn |

# 1. Introduction

## 1.1 Rationale for the TEEB rice study and the bigger picture

The Economics of Ecosystems and Biodiversity (TEEB) is a global initiative focused on drawing attention to the economic benefits of biodiversity including the growing cost of biodiversity loss and ecosystem degradation. TEEB recognizes that essentially all productive sectors depend upon the benefits provided by biodiversity and ecosystem services (including cultural services) that are collectively referred to as natural capital. It is important to note that TEEB does not conflate valuation with monetization or commodification.

The fundamental aim of the TEEB approach is to help decision-makers recognize, demonstrate and capture the **values of ecosystem services and biodiversity**, and help us to rethink our relationship with the natural environment and alert us to the impacts of our choices and behaviours on distant places and people. In the TEEB approach, scenario analysis is carried out to assess the provisioning of ecosystem services with a policy change versus Business As Usual (BAU). Ecosystem services are the benefits that nature provides us. Thus if we assess changes in ecosystem services (first in bio-physical terms and then using valuation approaches) we can recognize and demonstrate the trade-offs that policy-makers face in electing to support the proposed policy as compared with the BAU counter-factual.

The UNEP TEEB Office has recently begun to undertake a study on 'TEEB for Agriculture and Food' (TEEBAgFood in short). This study is designed to provide a comprehensive economic evaluation of the 'eco-agri-food systems' complex, and demonstrate that the economic environment in which farmers operate is distorted by **significant externalities**, both negative and positive and a lack of awareness of **dependency on natural capital**. A "double-whammy" of economic invisibility of impacts from both ecosystems and agricultural and food systems is a root cause of increased fragility and lower resilience to shocks in both ecological and human systems.

The Food and Agriculture Organization of the United Nations (FAO) together with its partners, the International Rice Research Institute and Bioversity International as well as Trucost has applied the TEEB framework to the rice farming sector, following an agroecological approach where possible.

## 1.2 Global and regional rice production and consumption

Rice is central to the food security of half the world. Furthermore, rice is a source of livelihood of around 140 million rice farming households. More than 90 percent of world rice production and consumption is in Asia, where recent strategy paper addressing policy makers identified a set of significant challenges in ensuring an adequate and stable supply of rice which is affordable to poor consumers (FAO, 2014):

- The need to produce more rice to meet the rising demand driven by population growth
- Deceleration in the growth of rice yields
- Environmental degradation associated with intensive rice production
- A decline in rice biodiversity and loss of rice heritage
- The role of rice production in global climate change
- Increasing competition of land, labor and water from industrial and urban sectors
- Changes in dietary composition
- Changes in demographic composition of labor in rural areas.
- Achieving price stability for rice in the context of shocks due to increased interconnectedness of rice with other sectors and instability in trade policies of the major exporting countries.

Many of these challenges are related to a decrease in ecosystem services, increase in negative externalities and the degradation of natural capital in rice production. As an answer, the FAO has proposed various strategic objectives to tackle these issues. The organization calls to increase the productivity and nutritional value of rice and rice-associated biodiversity, to improve mitigation and adaptation to climate change and reduce risk, and to minimize the environmental footprint of rice production while enhancing the ecosystem functions of rice landscapes, including the protection and promotion of rice heritage and cultural and landscape management.

As these objectives are not independent, but rather interlinked, reaching them is likely to require **trade-offs**. The question of interest is therefore of how to reduce trade-offs between these different goals. Where possible, one should identify **synergies** that allow for a maximization of benefits, while minimizing costs to society and the environment, (i.e. negative externalities), and the wellbeing of the farmer him or herself through the degradation of natural capital from rice production. It is therefore crucial to know which types of farm management practices or systems offer the best options to reach these synergies, and reduce trade-offs.

**Management decisions** on which practices to use need to be weighted carefully in order to do reduce environmental impact, on the one hand, and negative effects on yields, and potentially food security, on the other (see table X for different **management objectives**).

**Table 1.2.1.** Management objectives with regards to rice farming

1. Increase rice yields
2. Reduce water pollution
3. Reduce water use
4. Reduce air pollution
5. Use rice residues as source for energy production and/or animal feed
6. Reduce greenhouse gas emissions
7. Provide habitat for aquatic species to increase food provision and dietary diversity, enhance wild biodiversity, maintain ecosystem functioning and create space for recreational activities
8. Maintain the regulation of nutrient cycling and soil fertility
9. Reduce pest and disease outbreaks

Trade-offs between these different claims are not always necessary. The new paradigm of "sustainable production intensification" as described within the framework of FAO's save and grow concept (http://www.fao.org/ag/save-and-grow/) offers win-win situations among different ecosystem services without trade-offs. The recent meta-analysis on the multiple goods and services of Asian Rice production systems (Garbach et al. 2014), for instance, analyzed the synergies and trade-offs between ecosystem services and yield in six agroecological systems of rice production in relation to conventional, intensified agriculture. The study showed that both, yields and other ecosystem services of rice production systems, can be maintained or increased simultaneously when the right farm management system is chosen. Yet, while these win-win situations are possible, favorable policy environments and substantial investments in extension for rice farmers to apply these agroecological management systems are often necessary to support their growth and adoption (Garbach et al. 2014). The right questions to ask are therefore those that aim to inform decision makers of the benefits of agroecological or, in general, more sustainable rice management practices and make the case for better policy support. Showing the additional value of ecosystem services in rice production on top of the value for the rice crop itself is a step in this direction.

*Case Study countries*
The TEEB rice study set out to describe a variety of trade-offs and synergies that occur in rice agro-ecosystems in five case study countries all around the globe: the Philippines and Cambodia in Asia, Senegal in Africa, Costa Rica in Latin America and California/The United States in North America.

The vast majority of rice production is in Asia, with China and India accounting for 28% and 21% of global rice production, respectively. In terms of the focal countries for this study the Philippines is the 8th largest producer in the world (18 million tonnes in 2012) and Cambodia is the 12th (9 million tonnes in 2012). By comparison, of the other focal countries and regions, Senegal is ranked 44th (630,000 tonnes in 2012), Costa Rica 61st (213,000 tonnes in 2012). California, meanwhile, produced over 2 million tonnes of rice in 2011 (CalRice 2013).

Following the management interventions of the Green Revolution in the 1960s, per area yields of rice have increased greatly. However, the increase in yield has not been experienced equally in different world regions, with yields in Asia and Europe now greatly exceeding those in Africa. The disparities in yield have important implications for food security, income from rice production and opportunities for, and potential implications of rice production intensification.

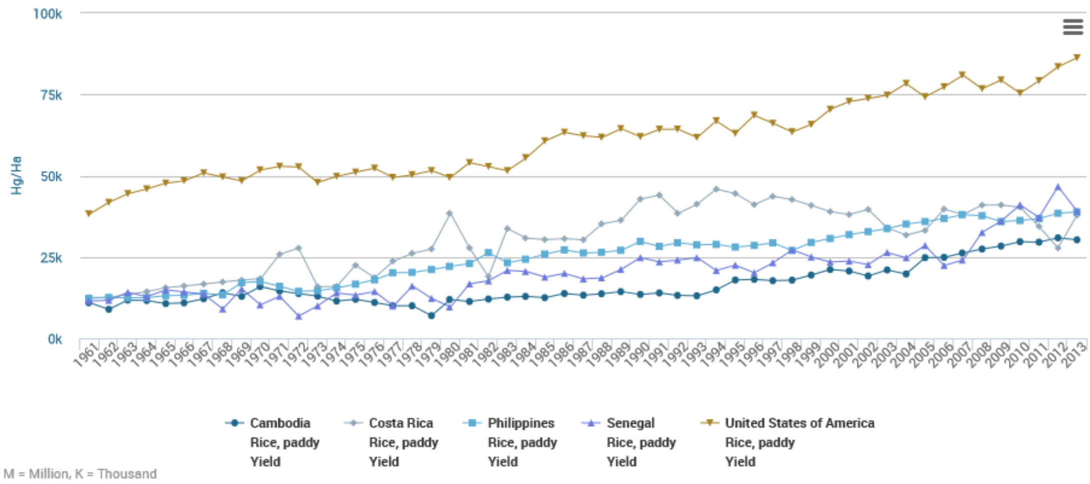


**Figure 1.2.1.** Rice yields in the five case study countries from 1961-2013.

The five case study countries covered in this analysis attempt to present a mix out of those countries that produce rice intensively based on high external inputs, and those countries that depend on a large part of smallholder, less extensive production methods.

In **Cambodia**, rice is by far the most important crop in the country accounting for over 80% of all agricultural production, by area (calculated from FAOSTAT data, 2015). Yet, average yields are still relatively small, despite the large increase over the past 10 years from 1.3 to 3.3 t/ha (calculated from FAOSTAT data, 2015), mainly due to increased use of fertilizers and inputs. Average yields in **Costa Rica** are 3.8 tons per hectare (calculated from FAOSTAT data, 2015). **The Philippines** has increasingly intensified over the past twenty years having considerable success in attaining higher yields. Average paddy yield is currently about 3.9t/ha and most farmers plant two crops per year (calculated from FAOSTAT data, 2015). **Senegal** has produced average yields of about 3.9 tons per hectare in 2013 (calculated from FAOSTAT data, 2015). **California,** one of the important rice producer regions in the United States, where average yields are 8.6 t/ha (calculated from FAOSTAT data, 2015) has average yields of 9.5 t/ha (USDA data from 2015). For a more detailed description of rice production typologies in the case study countries, please refer to the narrative review produced during this project.

As for consumption, in many countries rice is the staple diet, representing more than 40% of per capita calorie intake in tropical Asia, more than 65% of per capita protein intake in Bangladesh (Fairhurst and Dobermann, 2002), and about 95% of total grain consumption in Bangladesh (Sarma, 1999).

The traded volume of rice is quite small compared to most other agricultural commodities, amounting to around seven percent of total world consumption (FAO 2014). Nevertheless, trade has a key role in the food security of importing countries, and is a considerable source of revenue for many countries.

## 1.3 Valuation of environmental impacts and ecosystem services in rice

Agro-ecosystems provide a wide variety of different benefits, but can also cause costs to the agro-ecosystem itself, the environment and to society. Hence, rice agro-ecosystems may have positive or negative impacts, some of which are ecosystems services or disservices.

At the same time, rice agro-ecosystems rely upon or depend on a range of ecosystem services from the agro-ecosystem itself, and from other land uses and landscape elements proximate to rice (i.e. natural capital). The ecosystem services that rice agro-ecosystems depend on might be "pure" or "polluted" through man-made or natural interventions, e.g. clean water for irrigation versus water contaminated with heavy metals from the industry or the mining sector. Rice agro-ecosystems further depend on human capital such as labor and other man-made inputs such as knowledge or infrastructure. They further depend on the agro-ecological characteristics of the rice farming area, the socio-cultural context and the ultimate purpose of the rice farming operations. For instance, rice might be grown for self-sufficiency or as a commodity crop, or both.

In turn, as a result of these dependencies, rice agro-ecosystems impact, both positively and negatively, the system itself, the surrounding ecosystems (man-made or natural) and society. This shows that the relationship between impact and dependencies is not linear, but rather of circular nature.

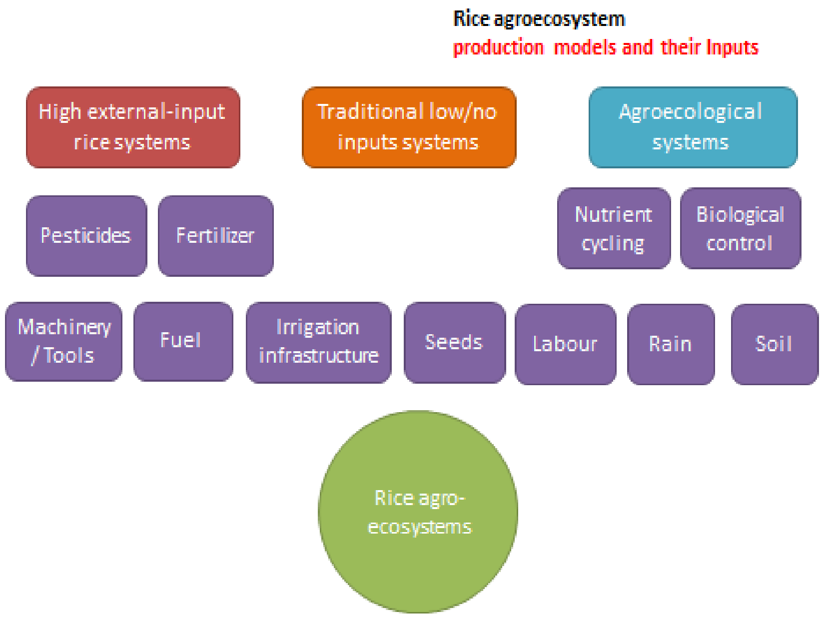


**Figure 1.3.1.** Simplified rice agroecosystem production models and their inputs.

Rice agro-ecosystems provide different visible or clearly perceived and less obvious benefits (positive impacts) and costs (negative environmental and socio-economic impacts) to the agro-ecosystem, the environment and to society; for the purpose of this project, they are also called outputs. Linking back to the previous section, agro-ecosystems generate ecosystems services that are depended on in other systems, or they generate impacts which can positively or negatively influence these inputs, natural or human capital.

Traditionally, agro-ecosystems have been managed for the provisioning of one service only, namely food production, yet depending on the type of management, they may also generate other provisioning services such as raw materials, and supporting, regulating and cultural ecosystem services leading to a variety of positive outputs, or positive externalities. For instance, depending on which type of farm management is used, some rice agro-ecosystems have been shown to improve ecosystem services such as dietary diversity, sequester carbon, mitigate greenhouse gas emissions, increase resilience, provide habitat for native flora and fauna and serve as recreational sites for hunting and fishing. The benefits that rice agroecosystems provide result from a combination of ecosystem services and specific types of agricultural management resulting in the provisioning service "rice grain", and different regulating, habitat and cultural services.

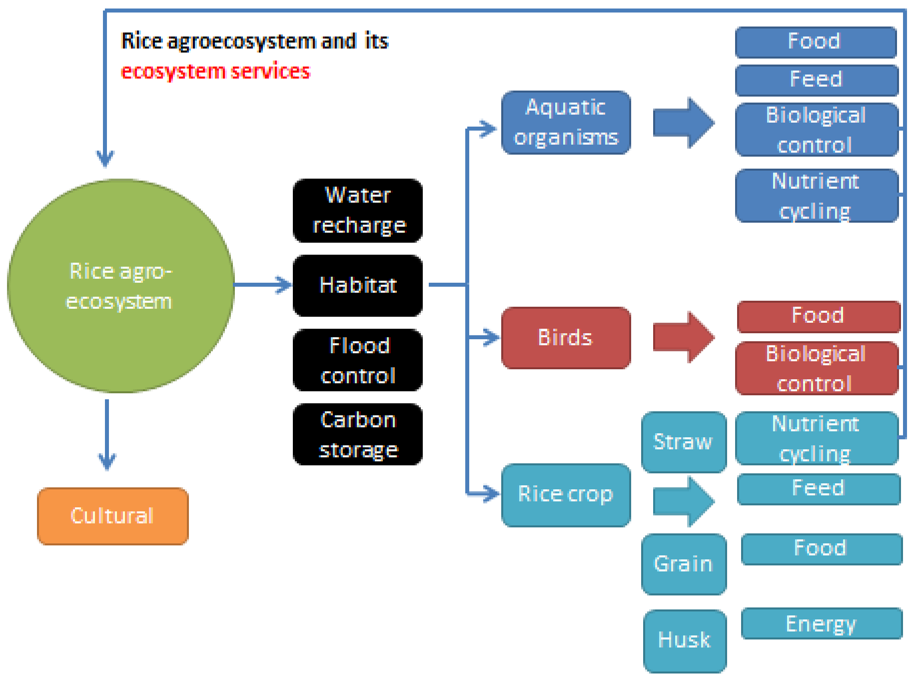


**Figure 1.3.2.** Simplified model of ecosystem services of rice production systems. The figure displays potential services found in rice agro-ecosystems. Not all of them could be quantified and monetized in this study due to lack of data.

Costs of rice systems all relate to environmental impacts and the degradation of natural capital, on the one hand, and the respective externalities for human health and the farm and its ecological infrastructure, on the other. Water contamination, air pollution, climate change, soil degradation and pest outbreaks and resistance are the main negative impacts that rice production can cause. If or to what degree these impacts materialize depends on the type of agricultural management applied.

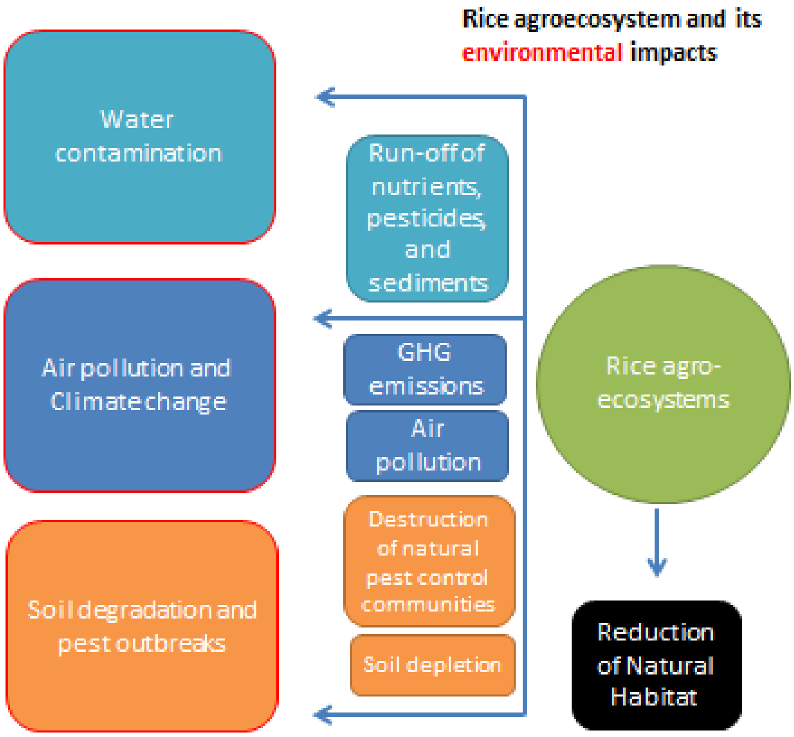


**Figure 1.3.3.** Environmental impacts of rice farming.

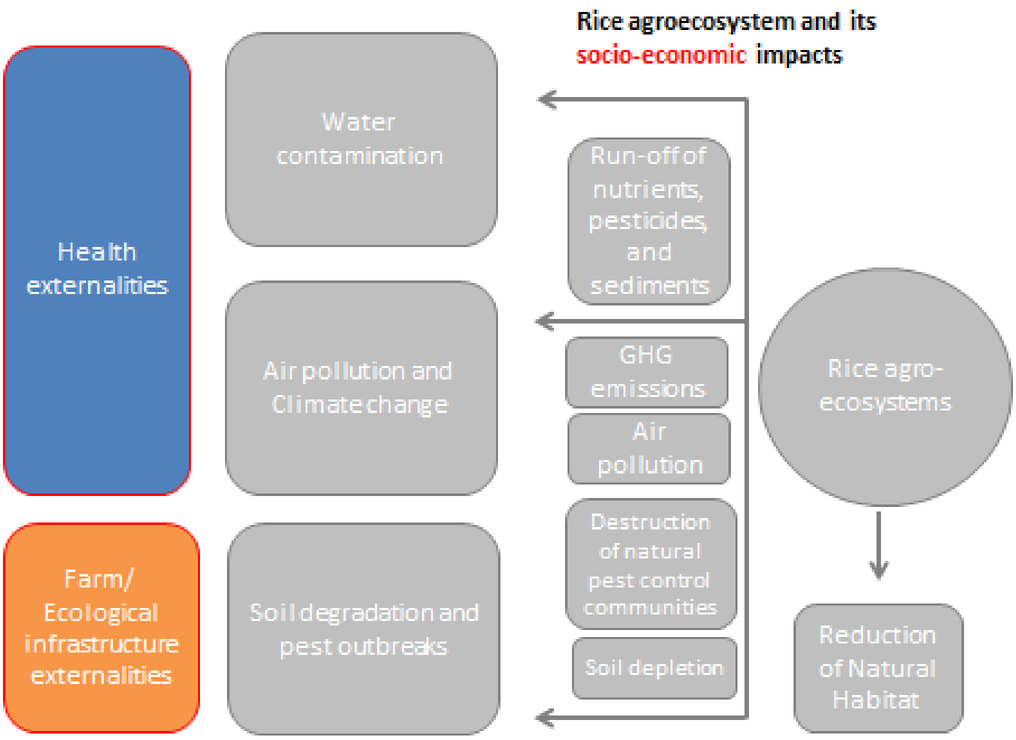


**Figure 1.3.4.** Socio-economic impacts of rice farming

## 1.4 Public versus private values

A change in the farm management, and consequently an increase or decrease of ecosystem services or environmental impacts, will be perceived differently by different stakeholders. Benefits or costs for one party (e.g. the landowner or producer) might not necessarily be the same for the other (the society). The international community is concerned about greenhouse gas emissions from intensive rice production systems; the local rice farming community has to deal with water contamination from excessive use of chemical fertilizers and pesticides; the consumers of rice might be interested in improving their dietary diversity, and the producer have to face input prices which increases production costs and impacts upon net income (more examples in table X).

**Table 1.4.1.** Examples of how the different stakeholder groups may be affected by a change in management practices and related ecosystem services.

| Affected party | How |
|---|---|
| **Producer** | Increased or decreased income from sale of rice or other products |
| | Increased or decreased expenditures for agricultural inputs such as pesticides and mineral fertilizer |
| | Enhancement or degradation of privately owned-natural capital (e.g. of soil quality and water availability) |
| | Increased or decreased work load |
| | Increased or decreased risks of climate change impact (e.g. pest outbreaks, drought) |
| **Society** | Positive or negative impact on groundwater recharge within the community area |
| | Positive or negative impact on health through safe versus contaminated food and water |
| | Increased or decreased risks of flooding of village |
| | Increased or decreased income from tourism. |
| | Increased or decreased expenditures for rice and other goods from rice systems |
| | Positive or negative impact on health through safe versus contaminated food and water |
| | Positive or negative impact on health through varied, nutritional diverse food versus a monotonous diet |
| | Increased or decreased risks of climate change impact |

## 1.5 Study objectives and target group

The specific objectives of this study were three-fold:

1. To identify visible and <u>invisible</u> costs and benefits of rice agro-ecosystems; i.e. externalities
    - Which ecosystem services are linked to rice production?

- Which types of environmental impacts does rice production have?

2. To identify and assess those rice management practises and systems which reduce trade-offs and increase synergies

    - How do costs and benefits change with different management approaches?

3. To make these trade-offs and synergies visible

    - Assign biophysical or monetary values to the different options

The immediate audience of this study is the TEEB secretariat of UNEP who commissioned feeder studies that inform the TEEB AgFood programme, and the building of a universal framework to capture positive and negative externalities of the food and agriculture sector.

At the same time, these *preliminary* results are aimed at informing decision makers in the food and agricultural sector, keeping in mind that valuation of this sector is still work in progress. One main target group are national and international policy makers as we are aiming to raise the awareness to the importance of this subject matter for better decision-making and to incentivize mechanisms for more sustainable rice production sector. This is particularly important as externalities of rice production concern the common good or third party actors which are different from the private actors causing the impact in the first place.

Another main target group are rice farmers and agri-businesses. This is particularly evident, as this study not only focuses on externalities, but also at costs related to the degradation of natural capital, i.e. the degradation of the privately-owned land and resources. Self-inflicted harm might arise owing to a lack of information on the dependency on ecosystem services. Farmers and agri-businesses are therefore likely be interested in the associated, mostly invisible, risks and opportunities related to rice production.

# 2. Methodology

## 2.1 Development of typology

The analysis makes a distinction between the three most common **rice growing environments**: irrigated lowland, rainfed lowland, and rainfed upland systems. The different typologies are based on altitude (upland vs. lowland) and water source (irrigated or rainfed).

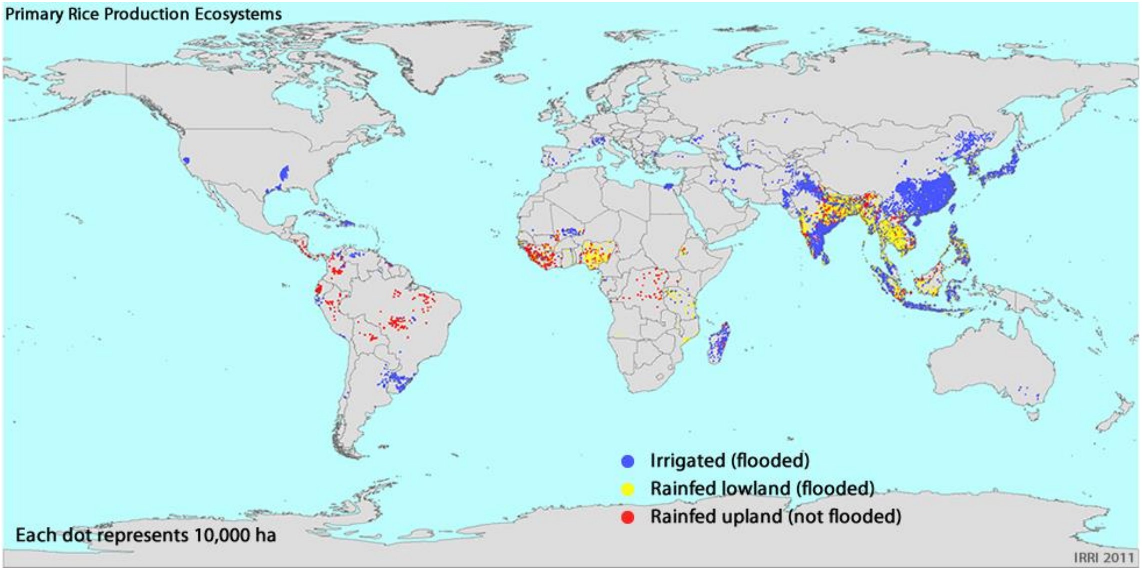


**Figure 2.1.1.** Map of different rice production systems globally, showing the considerable extent of irrigated rice (blue). Source: IRRI, 2009.

While these environments – which are sometimes also called **production systems -** are grouped according to similar environmental conditions, they still are characterized by a high degree of socio-economic and environmental heterogeneity, and the management practices applied within each farm or experimental site are very context specific.

This study has therefore focused not only on the three different main rice growing productions system, but also on the different **rice management practices and systems** related to plant production. Within this project, agricultural management practices or farming practices can be broadly defined as farming activities during the pre-planting, growth and post-production of rice. This includes the application of technologies and processes used in rice agriculture.

In this project, we refer to a "set of practices" or "management systems" when talking about a combination of different practices which usually follow a particular methodology or certain principles. These are sometimes arbitrarily called "systems" which should not be confounded with "farming systems" (see definition below). For instance, conservation agriculture is based on three principles which emphasize minimum soil disturbance, crop rotation and cover crops. Another example is integrated pest management which promotes an integrated control of insect pests and the enhancement of natural enemies. Organic agriculture is yet another combination of different practices, which are usually employed under certain principles and regulations by a certification body. While these management systems follow the same principles, they may look very different in their actual implementation. It is also important to understand that while these sets of practices do have fixed definitions, there is often more than one definition for each. By grouping and comparing agriculture within these "tags" or "brandings", one risks to group apples and oranges together.

"Farming systems" are not to be confounded with management systems - sometimes arbitrarily called "systems" or "management systems". Farming systems can be small subsistence units or large corporations. They are structurally complex and form various interrelationships between their numerous components (Dixon et al., 2001): different types of land, water sources and access to common property resources such as grazing lands, fish ponds and forests as well as other natural, human, social and financial capital. All these components, including the household, its resources, and the resource flows and interactions at the farm level are referred to as a farming system (Dixon et al., 2001).It is important to note that a farming system does not stop at the physical boundaries of the farm itself. The enabling environment is a determinant factor of the functioning of a farm system. This includes policies, institutions, markets and access to information (Dixon et al., 2001). Income from off-farm activities is also considered as part of the farming system, as it is often fundamental to maintain the farmers' livelihood and the farm itself. While farming systems would, in theory, be the best common denominator to compare the value of rice production, as they take all livelihood dimensions into account, they are hardly ever the focus of primary research. Hence no data is currently available to value these systems comprehensively. The analysis is therefore based on a comparison of different farm management practices and systems with a pre-planting, growing and harvesting phase. While each rice growing environment contains the same practice categories, there are important differences between the three systems in terms of practices, environmental impacts and ecosystem services, which are reflected in this analysis. The three main rice growing environments are:

*A. Irrigated lowlands*
Irrigated lowland rice constitutes around 75% of rice production yield globally and covers between 55 and 60 % of the global rice production area (see Fig. X). Lowland irrigated rice can be highly productive, with the potential to produce two or three crops per year. Application of water to the crop can depend on a number of factors, including water sources and availability, water distribution infrastructure and climate. In terms of temporal application, water can be used to augment other supplies in the rainy season, or can be applied only in the dry season. A major feature of irrigated rice, is that the rice production land is flooded permanently, or for most of the year.

B. *Rainfed lowland*
Rainfed lowland rice constitutes around 20 % of global rice production yields and around 30 % of the global rice production area. Systems lack irrigation and associated water control, and therefore are more prone to both flooding and drought (Jongdee et al 2006), and with productivity that is determined by the timing, frequency and amount of rainfall in the system (Saleh and Bhuiyan, 1995). Yields of rainfed lowland rice are generally less than that of irrigated lowland systems, averaging 4.9 t $ha^{-1}$ for irrigated, compared to 1.9 t $ha^{-1}$ for rainfed (Pandey, in Ladha et al. 1998). As well as water availability, yields are also determined by topography and soil fertility, and can be highly variable across small spatial scales (Wade et al. 1999). Another constraint on yield is that rainfall seasonality may restrict farmers to growing only one crop per year (IRRI, 2014).

*C. Rainfed upland*

Permanent upland rice production is mainly practiced by low-income farmers and is characterized by farming without bunds on typically sloping terrain, and produces less than 5 % of global yields on around 10 % of the global rice production area. In general, this is the lowest yielding rice system, with drought stress being a major constraint on production (Bernier et al. 2008), as an unbunded field system is entirely dependent on rainfall (Javier 1997). Low yields of upland rice are driving the development and distribution of drought resistant, high yielding varieties in order to improve upland production (Atlin et al. 2006). Upland rice is the most diverse in terms of varieties, reflecting the wide range of environmental conditions and soil types under which it is grown, and can be part of a shifting cultivation or permanent cultivation system (Javier 1997).

**Table 2.1.1.** Percentage of rice growing environments in the five case study countries/regions – Philippines, Cambodia, Senegal, Costa Rica and California

| | Irrigated lowland | Rainfed lowland | Rainfed upland | Other |
|---|---|---|---|---|
| **Philippines** | 70% | 30% | | - |
| **Cambodia** | 14% | 80% | 2% | 4% |
| **Senegal** | 70% | 30% | | - |
| **Costa Rica** | 34% | 66% | | - |
| **California** | 100% | - | - | - |

## 2.2 Setting of study boundaries

The study boundary details the content of the analysis- what is the scope, and what is the scale of analysis. The geographical scale of the study was set during the project development phase to the five case study countries as detailed above. Within each country, the study depended on primary research literature. Study sites were therefore those where research had been conducted – farm research sites or experimental sites of universities or research centers. These sites were usually restricted to the plant production area itself. Practices described fall under the pre-planting, growing and harvesting phase. Agro-ecosystem dependencies and impacts were usually confined to the study site itself. Impacts on the environment and human health such as eutrophication had to be modelled for the monetary valuation.

The study team identified and prioritized those dependencies and impacts that are directly related to the most common global **agronomic and environmental challenges** related to the intensification of rice production. Deceleration in the growth of rice yields, soil depletion, growing water use, increasing water and air pollution as well as climate change are some of the biggest areas of concern as already outlined in the introduction (see section 1 b). Tackling them at the same time is likely to require **trade-offs**, and **management decisions** need to be weighted carefully in order to do reduce environmental impact, on the one hand, and negative effects on yields, and potentially food security, on the other. The study team identified 9 management objectives related to rice farming to be addressed within the project (see table X for different **management objectives**).

**Table 2.2.1.** Management objectives with regards to rice farming

1. **Increase rice yields**
2. **Reduce water pollution**
3. **Reduce water use**
4. **Reduce air pollution**
5. **Use rice residues as source for energy production and/or animal feed**
6. **Reduce greenhouse gas emissions**
7. **Provide habitat for aquatic species to increase food provision and dietary diversity, enhance wild biodiversity, maintain ecosystem functioning and create space for recreational activities**
8. **Maintain the regulation of nutrient cycling and soil fertility**
9. **Reduce pest and disease outbreaks**

The study has set out to identify those **farm management practices** that offer the best options to reach synergies, and reduce trade-offs among different management objectives. Several pairwise comparisons ("Scenarios", see table X) were applied to show the effect of the various farm management practices on different environmental and/or agronomic variables:

1. The baseline scenario describes a conventional management approach, for instance herbicide use to combat weeds.
2. The alternative scenario describes a farm management practice that is expected to decrease an environmental impact or to increase an ecosystem service. For instance, instead of herbicide use, hand weeding is practiced.

**Table 2.2.2.** Practice and system comparisons included in the study.

| | | |
|---|---|---|
| **Management practices** | | |
| 1. Preplanting | Selection of rice varieties | Variety 1 – variety 2 |
| | Land preparation | Dry tillage – puddling |
| | | Land levelling – no levelling |
| | | Minimum soil disturbance – conventional tillage |
| | | No tillage – conventional tillage |
| 2. Growth | Planting | Direct seeding – transplanting |
| | | Dry seeding – wet seeding |
| | Water management | Low irrigation frequency - high irrigation frequency |
| | | Improved irrigation water management - continuous flooding |
| | Soil fertility management | Reduced mineral fertilizer use - high mineral fertilizer application |
| | | No fertilizer use – mineral fertilizer application |
| | | Organic fertilizer application - mineral fertilizer application |
| | | Organic fertilizer application - no fertilizer application |
| | | Mineral + organic fertilizer application – mineral fertilizer application only |
| | Weed management | No weed control - herbicide use |
| | | Biological weed control + hand weeding - herbicide use |
| | | Hand weeding – herbicide use |
| | | Reduced herbicide use – higher herbicide input |
| | Pest and disease management | No pesticide use - pesticide use |
| | | Reduced pesticide use – higher pesticide input |
| 3. Postproduction | Residue management | Winter flooding – no winter flooding |
| | | Straw incorporation – straw burning |
| | | Straw baling and removal – straw burning |
| | | Straw rolling – straw burning |
| **Management systems** | | |
| | | SRI – Conventional agriculture |
| | | Organic agriculture - Conventional agriculture |

## 2.3 Development of analytical framework

The study consists of two parts, a biophysical quantification and an economic valuation. The biophysical framework for analysis details what should be included in the analysis. The framework was developed in four steps: i. Writing of narrative report, ii. Identification of relevant response variables and indicators, iii. Data extraction and iv. Vote-counting analysis.

**The narrative report** details the rice production typologies in each country and outlines the most important environmental and agronomic challenges in each study area. The writing of the report further facilitated the systematization of different inputs and outputs related to different farm management practices, and hence the elements that need to be included in the valuation framework. These were captured in the form of 43 common **indicators** which provide specific information on the state or condition of the main **response variables** that describe an ecosystem service or environmental impact:

1. Rice yield, a provisioning ecosystem service
2. Nutrient cycling and soil fertility, which can be chemical or biological, the latter being a regulating ecosystem service
3. Pest control, which can be of chemical, cultural, mechanical, or biological nature, the latter being a regulating ecosystem service
4. Habitat, a habitat or supporting ecosystem service
5. Water quality, affected by pesticide and herbicide run-off, an environmental impact that results from specific types of agricultural management
6. Freshwater saving, resulting from a reduction in water consumption which in some cases can become an environmental impact
7. Mitigation potential of GHG emissions, an environmental impact that results from specific types of agricultural management

Additionally, the study included further relevant response variables which could not be directly extracted from the study literature, but were modelled or estimated based on related available data assumptions:

8. Provision of rice straw as a nutrient source
9. Provision of rice husks as an energy source
10. Air pollution, an environmental impact resulting from rice residue burning and the use of fertilizers
11. Water quality, affected by fertilizer run-off potentially leading to an environmental impact (eutrophication)

The project team further identified a set of relevant benefits which could only be qualitatively described, as no primary research data or estimates were available:

12. Dietary diversity, defined as the number of different foods or food groups consumed over a given reference period, can be enhanced by diverse rice-agroecosystems
13. Genetic variability, a supporting ecosystem service
14. Water purification, a regulating ecosystem service
15. Groundwater recharge, a provisioning ecosystem services provided by rice paddies and a hydrologic process where water moves downward from surface water to groundwater
16. Moderation of extreme events such as flood prevention or mitigation, a regulating ecosystem service provided by rice paddies when they hold water during heavy rainfalls
17. Cultural heritage, cultural services provided by rice paddies and their biodiversity

Market costs were assigned to agricultural inputs and labour when sufficient quantitative data and information on costs was available:

18. Pesticides
19. Mineral fertilizers
20. Labour

In a next step, we **extracted data** from peer reviewed literature from all five case study countries and synthesized them in a **vote-counting analysis** (see Van Dis et al 2015). The final outcome was a statistical review of primary research, i.e. peer reviewed literature, on the effects of different agricultural management practices on different environmental, agronomic and ecosystem variables.

The analysis presents the results of about 100 published studies that have examined the effect of at least one treatment comparison on at least one response variable, totaling more than 1500 data points and 750 interactions. These interactions either show an increase, a decrease or no effect in the strength or frequency of the interaction.

While such a statistical analysis is not mandatory for a valuation exercise, it helped to inform the development of the valuation framework and the interpretation of the valuation results. The results of this vote counting should however be taken as indicative only, as this statistical method is increasingly criticized as a rigorous scientific tool (Koricheva, & Gurevitch, 2013).

While further meta-analytical research is needed to fully understand the trade-offs among different rice management approaches, we have used this method to obtain a general overview of how different farm management practices can influence different environmental, agronomic and ecosystem variables in rice production.

Table 2.3.1 shows the valuation framework and gives an overview of all benefits and costs that were included in the study. The table is structured according to impacts versus dependencies, and realized versus hidden costs and benefits (e.g. externalities) and the affected stakeholder groups (e.g. producer vs wider society). It further states which benefits and costs were described qualitatively in the form of a narrative, which ones could be quantified with primary research data and which ones were estimated or modelled. Last but not least, it points to those that were included in the monetary valuation.

**Table 2.3.1.** Benefits and costs related to rice agro-ecosystems, detailing dependencies and impacts - some items can be both dependency and impact as they are both produced and used in the same system, realized versus hidden costs and benefits (i.e. externalities), and affected stakeholder groups. The table further shows which benefits and costs have been valued qualitatively, which ones have been quantified in biophysical terms (on the basis of primary research or modelled data) and which ones have been valued in monetary terms.

| | | **Dependencies and impacts** | | **Realized vs hidden costs and benefits** | | **Affected stakeholders** | | **Qualitative description** | **Biophysical quantification** | | **Monetary valuation** |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | | **Dependency** | **Impact** | **Visible** | **Invisible** | **Producer** | **Society** | | **Primary research data** | **Modelled data** | |
| **Benefits** | Rice grain (Revenue for rice grain of food) | | x | x | | x | | x | x | | x |
| | Rice straw (Nutrient value) | | x | (x) | x | x | | x | | x | x |
| | Rice husk (Energy value) | | x | (x) | x | x | | x | | x | x |
| | Pest control | x | x | | x | x | | x | x | | |
| | Nutrient cycling and soil fertility | x | x | | x | x | | x | x | | |
| | Water purification | x | | | x | x | x | x | | | |
| | Cultural heritage | | x | | x | | x | x | | | |
| | Recreational and tourism opportunities | | x | x | x | | x | x | | | |
| | Flood prevention | x | x | | x | | x | x | | | |
| | Water recharge | | x | | x | | x | x | | | |
| | Habitat provisioning | x | | | x | | x | x | x | | |
| | Dietary diversity | | x | | x | x | x | x | | | |

| | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| **Costs** | Water pollution (Pesticide and herbicide run-off) | | x | | x | | x | x | x | | x |
| | Water pollution (Eutrophication) | | x | | x | | x | x | | x | x |
| | Land pollution | | x | | x | | x | x | | | |
| | Air pollution (Rice straw burning for disposal) | | | | | | x | x | | x | x |
| | Air pollution (Rice husk burning for energy) | | | | | | x | x | | x | x |
| | Air pollution (Fertilizer) | | x | | x | | x | x | | x | x |
| | Water consumption | x | | | x | | x | x | x | | x |
| | GHG emissions | | x | | x | | x | x | x | | x |
| | Labor | x | | x | | x | | x | | | |
| | Fertilizer | x | | x | | x | | x | x | | x |
| | Pesticides | x | | x | | x | | x | x | | |

## 2.4 Biophysical and monetary valuation

Placing monetary values on the costs or benefits that arise due to different management practices takes place in three distinct steps. This process is guided at all times by an overarching research question, which outlines the aim of the monetary valuation, why the valuation is needed, and who the target audience is.

The first step, which measures the changes in physical conditions, has been performed in the academic literature used for this study. This includes the identification of the drivers for change, such as fertiliser or pesticide inputs. Additional to extracting this data in a standardized way across all five case study countries, a vote counting analysis was done to synthesize these results. This statistical review of primary research, i.e. peer reviewed literature, on the effects of different agricultural management practices on different environmental, agronomic and ecosystem variables, presents the results of 70 published studies that have examined the effect of at least one treatment comparison on at least one response variable, totaling more than 1500 data points. The interactions either show an increase, a decrease or no effect in the strength or frequency of the interaction. For more specific information on how the different treatment comparisons and response variables were grouped and which assumptions were made, please refer to the detailed vote-counting analysis report.

The second step requires the biophysical modelling of the impact, or impacts, that are caused by changing physical conditions. This includes identifying factors such as the endpoint of nutrient run-off, e.g. freshwater ecosystems, and quantifying the change in the biophysical indicator that is to be valued, such as the change in the quality of human health, measured in disability adjusted life years (DALYs) (see below for more details).

The final step involves the economic modelling component of the valuation. This step involves the identification of the final recipient of the impact, such as the local populations who experience the negative effects of eutrophication, and then selecting an appropriate valuation technique to monetize the change biophysical conditions.

When assigning monetary values to the costs and benefits arising due to different management practices in this study, it is essential that these steps are followed so that valuations can be viewed and communicated in a standardised way. The aim of this framework is to promote the understanding of monetary valuations, which reflect key spatial and temporal characteristics.

In this study, the biophysical modelling assigns the costs and benefits of the impacts to either human health, or ecosystems, arising from different management practices. Human health is measured in terms of DALYs. This metric quantifies the burden of disease on human populations, and can be thought of as one year of healthy life lost. The measure includes both the years of life lost due to illness (mortality), and the years of healthy life lost due to disability (morbidity). The valuation approach uses a willingness-to-pay (WTP) survey, which elicits values from society based on changes in factors like reduced income due to ill health, the pain and discomfort caused, as well as decreased life expectancy.

The costs or benefits of the impacts on ecosystems are quantified in terms of the change in ecosystem functioning, and then valued in terms of the change in the monetary value of the ecosystem services provided. Ecosystem functioning is measured as the change in net primary production (NPP) within ecosystems outside of the farm gate. Currently, impacts on the farm have not been considered. The monetary valuation approach involves conducting a meta-analysis of primary valuation studies of provisioning, regulating, and cultural ecosystem services. The approach allows the quantification and valuation of ecosystem services that are impacted due to changes in environmental quality. This can be due to the emission of air land and water pollutants, or to changes in water availability. Provisioning ecosystem services, such as rice and rice husk production, coming from within the farm gate have been valued using direct market pricing.

Please see the valuation methodology documents for more detail on any of the above topics.

The final results of the biophysical and economic valuation are presented as follows in section 3.

## 2.5 Scenario comparison

What if a country would decide to change from one management practice to the other throughout the entire country? What if the Philippines would change from continuous flooding in irrigated lowland systems to intermitted flooding on their entire rice production area? What if Senegal would practice SRI all throughout the country? While these are hypothetical questions and unlikely to occur in reality, they help to illustrate the magnitude of the effects.

In the last step, we upscale management practices from field to country level. All results – costs and benefits – are given on a per hectare basis. Knowing the rice farming area in each country, and the percentage of irrigated lowlands, rainfed lowlands and rainfed upland systems, one can calculate the production area in each rice growing environment. Multiplying this area by the difference in impact between two management practices, one can calculate the gains, losses or savings related to an environmental impact or ecosystem service when changing from one scenario to the other.

**Table 2.5.1.** Rice growing area in each case study country (Bogdanski et al, 2015a)

| | **Area under rice production in ha** |
|---|---|
| **Philippines** | 4,000,000 |
| **Cambodia** | 3,100,000 |
| **Senegal** | 135,129 |
| **Costa Rica** | 55,709 |
| **California** | 227,838 |

# 3. Results and discussion

The results and discussion part is structured in the following way: The nine main sections present different ecosystem services or environmental impacts related to rice farming in relation to rice production. After a short introduction, the respective valuation methodologies are briefly explained. More detailed information can be obtained from Trucost's Valuation Methodology document, which was developed specifically for this project.

The next level comprehends a listing of all management practice or system interactions that are related to this environmental impact or ecosystem service. How does the impact or service change when two different management practices or systems are applied? For instance, what happens to water quality if the system of rice intensification is implemented as compared to conventional management? At the same time, this section shows what happens to rice yields when the management practice or system is changed. Is there a trade-off or a synergy with regards to the environmental impact or ecosystem service? For instance, do yields increase or decrease while water quality improves?

Within each practice comparison, we further distinguish between irrigated lowland, rainfed lowland and rainfed upland systems. These are then further subdivided by country.

While this structure calls for comparisons between different production systems and between countries, the study authors did deliberately not do this as the values of benefits and costs derived from rice farming are country and system specific. Comparing one with the other would be mixing apples and oranges. Country and systems results are therefore not compared to each other, but rather listed independently to show the diversity of rice farming values in different contexts.

## 3.1 Increase in rice yield versus maintenance of water quality

In most parts of the world, rice agriculture is heavily dependent on agricultural inputs. Synthetic fertilizers are used to boost yields, while pesticides and herbicides are applied to address pest outbreaks and weed manifestation. Weed outbreaks can cause losses in rice production and thereby lower the income of farmers, while it is still unclear whether pesticides (particularly insecticides) actually increase rice yields (Heong et al 2015). Agricultural expansion and intensification can often lead to a change in the ratio of predatory invertebrates to herbivorous invertebrates. One of the main reasons is the misuse of pesticides. Not only is rice production itself affected but also the adjoining waterways, their wildlife such as fish and birds and the supply of drinking water. Finding alternative ways to address pests is therefore very important. Another reason is the increased use of fertilizer. Increasing fertilizer use often leads to higher disease incidence and a greater abundance of herbivorous insects and mites. This, in turn, often leads farmers to apply higher levels of pesticides and thereby reduce ecosystem efficiency and reduce water quality (Horgan and Crisol 2013; Spangenberg et al 2015).

Excess application of synthetic or organic nitrogen (N) or phosphorous (P) fertilizers to agricultural land is an important cause of eutrophication in nearby water bodies. Eutrophication can affect rivers, lakes, reservoirs and coastal waters. The enriched waters, when warmed in summer, can lead to blooms of algae, which have short lifespans, and decay via a process that consumes dissolved oxygen in the water. These algae blooms can be so severe that all available dissolved oxygen is consumed resulting in hypoxia, which kills fish and other organisms (Anderson et al., 2002). Harmful algae blooms (HABs) occur when toxic algae grow in the water (Ibid). These blooms can give off an unpleasant smell, reduce water clarity and harm the health of animals that consume the water. Algae blooms can occur in both inland and coastal waters.

Nitrate run-off from agricultural fields to surface water, or leaching to groundwater, can also pose health risks for humans where nitrate concentrations exceed critical thresholds. The World Health Organisation (2011) recommends a limit on nitrate and nitrite concentration in drinking water of 50mg/l and 3mg/l respectively.

This study sought to assess and valuate trade-off resulting from pesticide and herbicide use on rice yields, on the one hand, and water quality, on the other. It furthermore aimed to assess the trade-off between fertilizer use on yield and on water quality. The value of rice production as food for the farmer was estimated on the basis of the country specific revenue for rice grain received per tonne of paddy rice.

The pesticide and herbicide damage cost was estimated per kilogram of pesticide and herbicide active ingredient applied to agricultural soil. The biophysical data was first converted from litres per hectare to kilograms per hectare (where necessary) using the density of the pesticide. The valuation coefficients were then applied to estimate the monetary impact of pesticide and herbicide inputs per hectare.

The ecosystem impact valuations are based on continental scale modelling of the dispersion of pesticides and herbicides applied to agricultural soil and a country specific valuation of the complement of ecosystems contained within each country that are affected by each pesticide or herbicide. This method is described in further detail in Trucost's methodology document.

The Trucost economic modelling methodology provides valuations for the impact to human health of pesticides and herbicide. The human health impact valuation is based on continental scale modelling of the dispersion of pesticides and herbicides and their health impact expressed in DALYs. Each DALY is valued based on a global median Value of a Life Year (VOLY) calculated via a method taking account of income elasticity.

In order to value the potential eutrophication and drinking water contamination impacts of agricultural fertiliser use, it was necessary to estimate the quantity of nitrogen and phosphorous that was transported from the field via run-off and leaching. Combined run-off and leaching has been estimated by first estimating the nitrogen and phosphorus balance of the rice paddy over the growing period and then calculating the water balance.

The Trucost Eutrophication methodology provides valuation coefficients for the impact of nitrogen and phosphorus emissions to water and is applied to value the impact of fertiliser use on ecosystems, water quality and water treatment costs. This valuation is based on the estimated change in secchi depth (a measure of water clarity and eutrophic state) per kilogram of nitrogen or phosphorus deposited in a hypothetical lake constructed for each country based on country specific data. The change in secchi depth is valued based on a hedonic pricing study undertaken in the USA (further details are provided in the Appendix).

The Trucost Eutrophication methodology also provides valuation coefficients for the impact of nitrogen and phosphorus emissions to water and is applied to value the impact of fertiliser use on human health. This valuation is based on the estimated excess water treatment cost and the number of DALYs lost due to unsafe drinking water (where water is not treated) per kilogram of nitrogen or phosphorus emitted to water bodies (further details are provided in the Appendix). The valuation of water treatment cost impacts is based on a USA study and health effects are valued based on a global value per DALY gained or lost.

No data was available on the purchase cost of herbicides and pesticides applied in the studies identified in the literature review. As such, average market prices for key pesticide and herbicide products were estimated based on a limited search of advertised market prices on an online business to business marketplace. Alibaba.com is one of the largest online business-to-business trading websites in the world and is targeted at small to medium enterprises. Market prices were averaged across multiple suppliers and converted to prices per kilogram of active ingredient based on the active ingredient concentration and density of each product.

Likewise, no data was available on the cost of chemical fertilisers applied in the studies identified in the literature review. As such, study specific fertiliser inputs were valued on the basis of the estimated average price per kilogram of nitrogen, phosphorous or potassium for a country representative fertilizer. No data was available on the cost of organic fertiliser inputs and thus these costs could not be valued. However in many cases, the organic inputs used were residues or rice cultivation (such as rice straw or rice straw compost) that may be accessed at negligible cost.

Data was available for irrigated lowland systems (IL) and rainfed lowland systems (RL)

### a. Omitting the use of herbicides (IL)

In an experiment in irrigated lowland systems in the **Philippines** (Bhagat, et al 1999), the use of the herbicide pretilachlor (Sofit) led to an average revenue of US$2652 for one hectare of rice, while the value was US$2030 when no herbicide was applied. This equals a reduction of 23%.

The herbicide damage cost from applying pretilachlor (Sofit) in this Philippines case study was estimated to be US$0.2 per hectare. Herbicide inputs costs were on average US$1.5 per hectare.

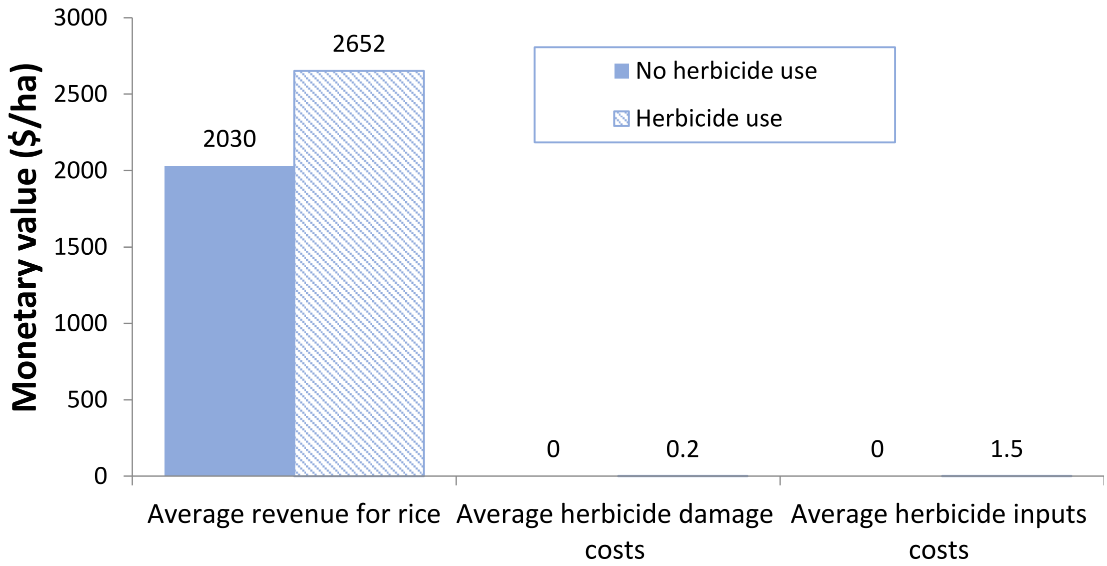


**Figure 3.1.1.** Comparison between the effects of herbicide use and no weed control on the **revenue** for rice grain, the herbicide damage costs and the herbicide input costs in US$ per hectare in irrigated lowland systems in the Philippines.

In an experiment in **Senegal** (Rodenburg et al. 2014), the use of the herbicides propanil and 2,4-D led to an average value of US$3727 for one hecatre of rice, while the value was US$1148 when no herbicide was applied. This equals a reduction of 69%.

Water quality increased in all cases when no herbicides were used. The herbicide damage cost from applying in this Senegalese study was estimated to be US$1 per ha, while no herbicide application caused no costs – a reduction of 100%.

The price for the herbicide amounted to US$350 per hectare, which was 350 times the value of the herbicide damage cost. While the herbicide input costs incurs to the farmer, the herbicide damage cost incurs to the society.

Labour costs were estimated based on a different study undertaken by Krupnik et al (2012) in Senegal. The labour costs for herbicide application according to standard farmer practice in the wet season were given at 2.2 mean person days (that are 8 hours) per hectare and season at a cost of 2.20 Euros per unit or 2.6 US dollars. This means that the costs for herbicide spraying were US$ 5.26 per hectare and season.

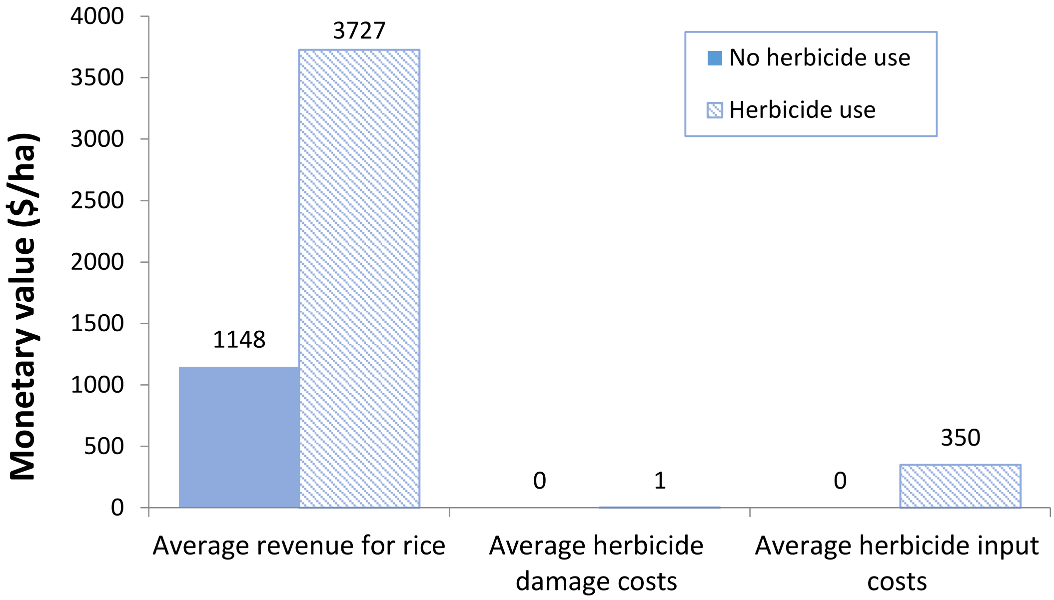


**Figure 3.1.2.** Comparison between the effects of herbicide use and no weed control on the **revenue** for rice grain, the herbicide damage costs and the herbicide input costs in US$ per hectare in Senegal.

In an experiment in **California** (Gibson et al 2001), the use of the herbicide Propanil led to an average value of US$2629 for 1 ha of rice, while the value was US$47 when no herbicide was applied. This equals a reduction of 98%.

Water quality increased in all cases when no herbicides were used. The herbicide damage cost from applying herbicides in this Californian study was estimated to be US$ 0.5 per ha, while no herbicide application caused no costs – a reduction of 100%.

The price for the herbicide amounted to an average cost of 85US$ per hectare, which was 170 times the value of the herbicide damage cost. While the herbicide input costs incurs to the farmer, the herbicide damage cost incurs to the society.

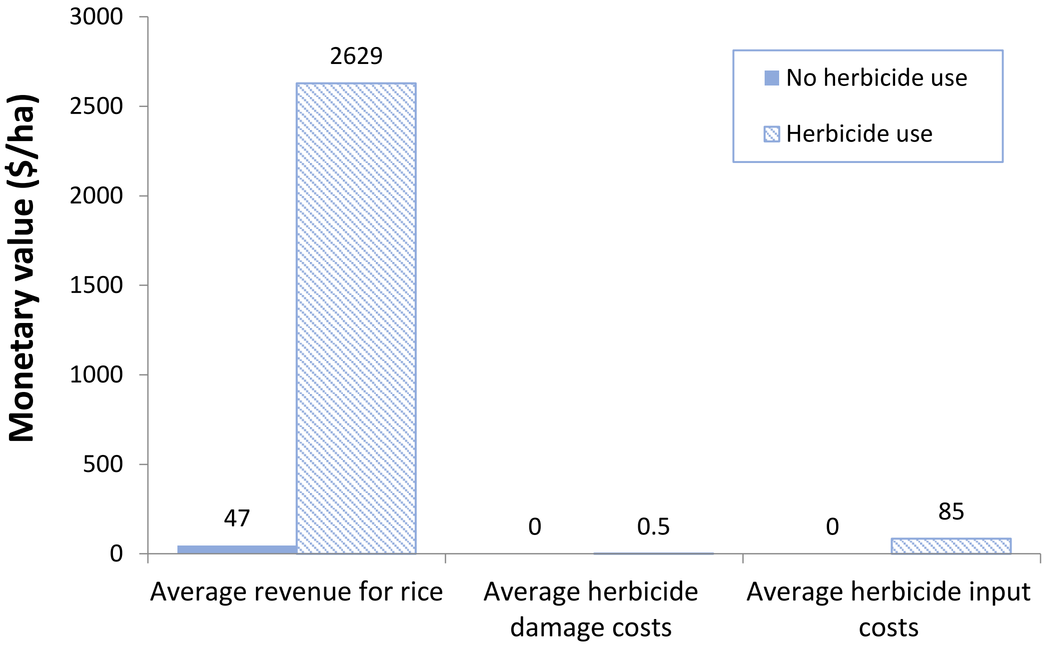


**Figure 3.1.3.** Comparison between the effects of herbicide use and no weed control on the **revenue** for rice grain, the herbicide damage costs and the herbicide input costs in US$ per hectare in California.

We present these three experiments separately to show the large difference of effects between the uses of different herbicides in three different locations. While these data points only represent three different experiments, the message is very clear: The effect of herbicides on water quality varies widely depending on the type of chemical used, and the context they are applied in. If adjoining waterways are close by, the herbicide damage costs are likely to be higher, than when no water bodies are located close to the spraying site.

If the **Philippines** stopped using the herbicide petrilachlor in the entire irrigated lowland area, practicing no weed management at all, the rice producer community would lose a total of US$ 1,915,760,000 through yield decreases. At the same time, the rice producer community would save US$4,620,000 in herbicide inputs costs.

On the other hand, society would not have to pay the water pollution related ecosystem and health costs of US$616,000. However, this is not even one percent of the loss that the rice farming community would face through yield decreases.

If **Senegal** stopped using the herbicides propanil and 2,4-D, practicing no weed management at all, the rice producer community would lose a total of US$ 243,947,610 through yield decreases. At the same time, the rice producer community would save US$ 33,106,500 in herbicide inputs costs. And US$ 497,543 in labour costs for herbicide spraying.

On the other hand, society would not have to pay the water pollution related ecosystem and health costs of US$94,590. However, this is not even one percent of the loss that the rice farming community would face through yield decreases.

If **California** stopped using the herbicides propanil, practicing no weed management at all, the rice producer community would lose a total of US$ 588,277,716 through yield decreases. At the same time, the rice producer community would save US$ 19,366,230 in herbicide inputs costs.

On the other hand, society would not have to pay the water pollution related ecosystem and health costs of US$113,919. However, this is not even one percent of the loss that the rice farming community would face through yield decreases.

Upscaling herbicide damage costs from field to country level as done here can only give a very rough indication of the damage cost at country level. As explained above, herbicide damage costs depend largely on the distance of the field to the next waterway. Hence values are very context specific.

Furthermore, irrespective of the context, the results of applying a monetary valuation to herbicide damage seem to extremely low, given that there is increasing evidence in the global literature that there may be direct human health costs, and the longer term development of resistance or reduction of ecological infrastructure to prevent or impede weed infestations. These and the following results related to herbicide and pesticide use therefore need to be treated with a lot of caution. For example, a recent publication by the y the International Agency for Research on Cancer (IARC) announced that glyphosate, a widely used herbicide, also in rice, is probably carcinogenic to humans (Guyton et al 2015).

### b. Omitting the use of pesticides (IL)

When no pesticides were used to control for pests, yields declined significantly in almost all instances in irrigated lowland systems in California and in the Philippines as detailed in the vote-counting analysis.

In one experiment in the **Philippines** (Kreye et al 2009), the value of one hectare of rice decreased 97% from US$1135 to US$32 on average when the soil fumigant Dazomet which acts as a fungicide was not applied.

As to be expected, water quality increased in all cases when no fumigants were used. The average pesticide damage cost from applying Dazomet in this Philippines case study was estimated to be US$882 per hectare while the herbicide free treatment had no associated costs.[1]

The value of one hectare of rice in **California** (Wu & Wilson, 1997) decreased 22% from US$2,439 on average to US$1,842 when the pesticide carbofuran, used to control insects, was not applied. There was not sufficient data to calculate the pesticide damage cost in California.

If **the Philippines** stopped using the pesticide Dazomet, practicing no pest management at all, the rice producer community would lose a total of US$ 3,397,240,000 through yield decreases.

On the other hand, society would not have to pay the water pollution related ecosystem and health costs of US$2,716,560,000.

If **California** stopped using the pesticide carbofuran, practicing no pest management at all, the rice producer community would lose a total of US$136,019,286 through yield decreases.

### c. Decreasing the rate of herbicide use (IL)

In irrigated lowland systems, water quality improved in all cases significantly when herbicide use was reduced. However, in 60% of all cases lower herbicide inputs led to a decrease in crop yields

[1] The estimated price for the fumigant amounted to 9014 US$ per hectare, which was more than 8 times the value of the revenue for rice grain producer price – an unrealistic estimate. One can assume, that if bought and used at such large quantities, Dazomet can be purchased at a significantly lower price than that one suggested in the methodology document.

while in 40% there was no difference between the two treatments. As weeds strongly compete with rice plants, these results are not surprising.

For country disaggregated data from a study in the **Philippines** (Bhagat et al 1999), the revenue for rice grain for one hectare of rice was US$2611 when herbicide use (petrilachlor) was reduced by half and US$2693 when the full dose was applied – a reduction of 3%. The herbicide damage cost was US$ 0.34 with higher inputs and decreased by 71% to US$ 0.1 when the herbicide rate was reduced. The herbicide input costs were US$ 2.73 and US$ 1.35, respectively.

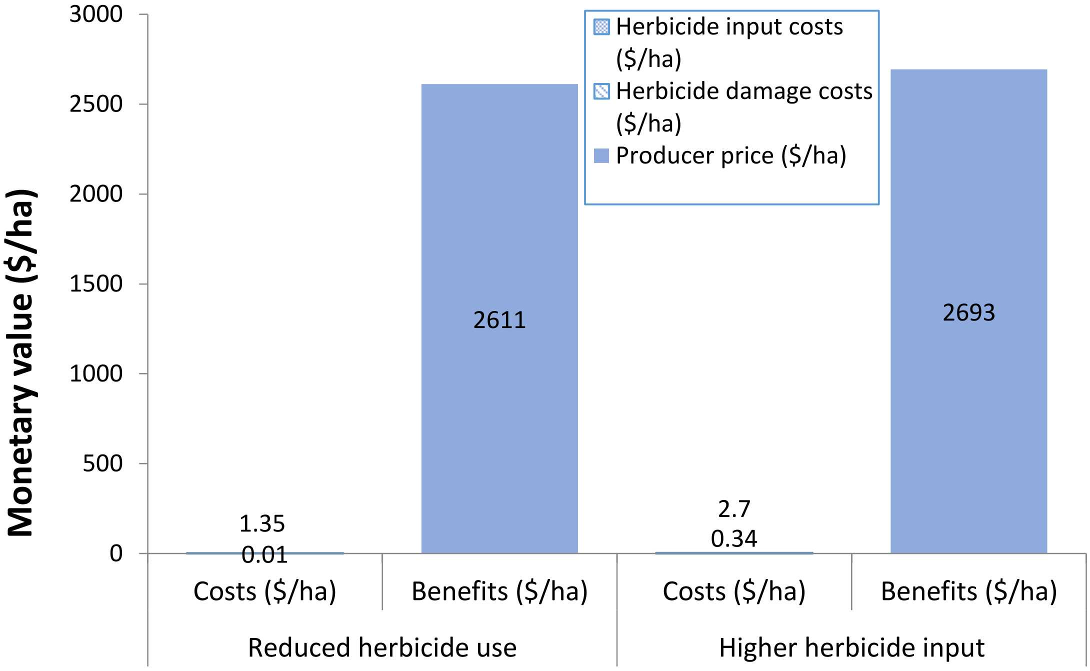


**Figure 3.1.4.** Comparison between the effects of reducing the rate of herbicide use on the **revenue** for rice grain, the herbicide damage costs and the herbicide input costs in US$ per hectare in the Philippines.

For data from irrigated lowland systems in **California** (Gibson et al 2001), the average revenue for rice grain for one hectare of rice was US$1776 when herbicide use (Propanil) was halved or quartered and US$3482 when the full dose was applied – a reduction of 50 %. The herbicide damage cost was US$0.71 with higher inputs and 0.29 with fewer inputs. The average herbicide input cost was US$ 122 for the higher rate, and US$ 49 for the lower pesticide rate.

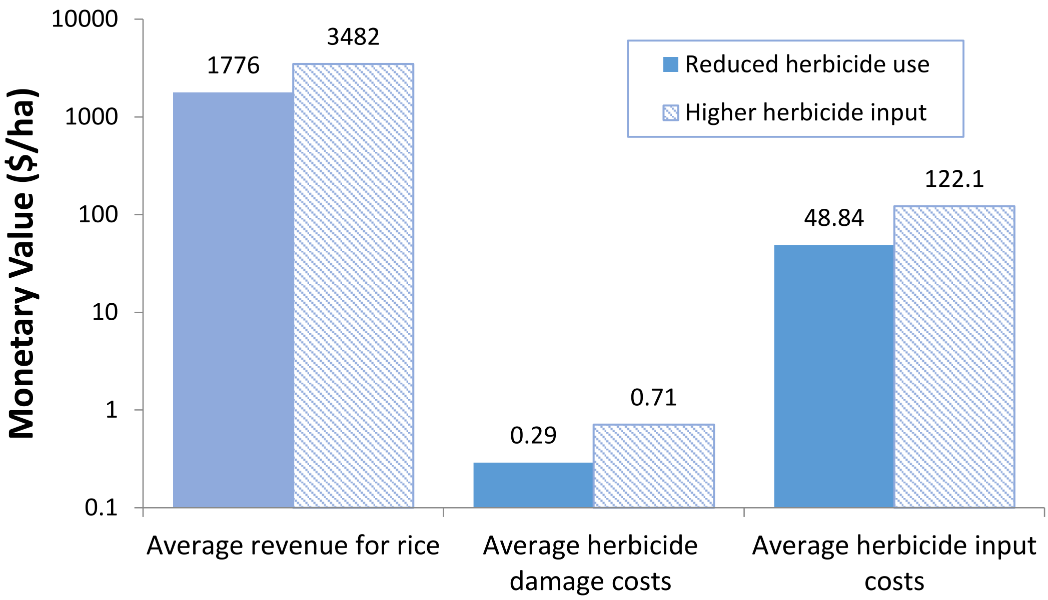


**Figure 3.1.5.** Comparison between the effects of reducing the rate of herbicide use on the **revenue** for rice grain, the herbicide damage costs and the herbicide input costs in US$ per hectare in California The y-axis was changed to log-scale to better portray the different impacts.

If the **Philippines** reduced the herbicide petrilachlor in the entire irrigated lowland area by half, the rice producer community would lose a total of US$ 252,560,000 through yield decreases. At the same time, the rice producer community would save US$4,250,400 in herbicide inputs costs.

On the other hand, society would not have to pay the water pollution related ecosystem and health costs of US$708,400. However, this is not even one percent of the loss that the rice farming community would face through yield decreases.

If **California** reduced the herbicide propanil in the entire irrigated lowland area by half or to a quarter, the rice producer community would lose a total of US$388,691,628 through yield decreases. At the same time, the rice producer community would save US$16,700,525 in herbicide inputs costs.

On the other hand, society would not have to pay the water pollution related ecosystem and health costs of US$1,293,600. However, this is not even one percent of the loss that the rice farming community would face through yield decreases.

### d. Combining hand weeding with biological control compared to herbicide use (IL)

In irrigated lowland systems in **Senegal** (Riara et al 1997), water quality improved when biological weed management together with manual weeding was practiced instead of herbicides, as expected. There was no statistical significant difference in yield in all cases which shows a clear advantage of those weed management practices that build on ecosystem services – maintaining water quality while delivering the same yields as when herbicides are used.

This is also reflected in the monetary values. Biological and manual weed control led to an average hectare value of US$2,927 while herbicide use led to a value of US$2,941 - a reduction of not even one percent when no herbicides were used. The herbicide damage cost was calculated at US$1 per hectare.

The herbicide input cost was US$518 per hectare, and thereby 518 times higher than the herbicide damage cost.

Labour costs were estimated based on a different study undertaken by Krupnik et al (2012) in Senegal. The labour costs for herbicide application according to standard farmer practice in the wet season were given at 2.2 mean person days (that are 8 hours) per hectare and season at a cost of 2.20 Euros per unit or 2.6 US dollars. Costs for hand weeding were given at 11.3 mean person days per hectare and season, at a unit cost of 1.11 Euros or 1.26 US dollars. This means that hand weeding costs are US$ 14.24 and the costs for herbicide spraying are US$ 5.26.

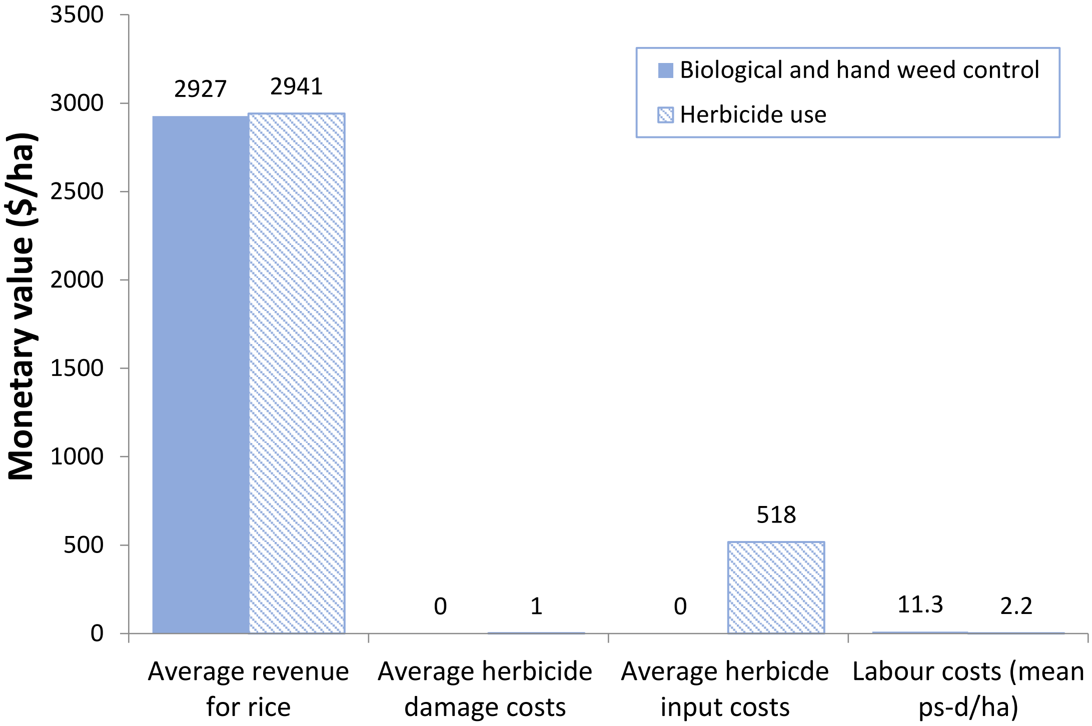


**Figure 3.1.6.** Comparison between the effects of herbicide use and combined hand and biological control on the **revenue** for rice grain, the herbicide damage costs, the herbicide input costs and labour costs in US$ per hectare in Senegal.

If **Senegal** changed from herbicide use to a combination of manual and biological weed control in the entire irrigated lowland area, the rice producer community would lose a total of US$ 1,324,260 through yield decreases. At the same time, the rice producer community would save US$ 48,997,620 in herbicide inputs costs. The savings in herbicides costs are 37 times the value of the revenue for rice grain losses. However, these herbicide costs are based on a rough methodology which builds on global averages. Hence, the results need to be treated with caution. The difference in labour costs would amount to US$ 849,418 an increase in costs for those that employ labour on their farms and a gain for those rural workers that do not own their own land. On the other hand, society would not have to pay the herbicide damage costs of US$94,590.

### e. Manual weeding compared to herbicide use (RL)

A study undertaken in **rainfed lowland systems** in **Cambodia** (Rickman et al 2001) compared the effect of manual weeding to herbicide use. Unsurprisingly, water quality improved when manual weeding was practiced. Yields increased in half of all cases, and showed no difference in the other half. The fact that hand weeding was more successful than herbicide use had to do with the fact that the post-emergent herbicide thiobencarb failed to significantly control some of the weeds.

The average income for the harvest of one hectare was US$1279 for manually weeded fields, and US$1062 when herbicides were used – a reduction of 17%. The herbicide damage cost could not be calculated as information on the quantity of herbicides used was missing in the literature.

### f. Omitting mineral fertilizer use (IL)

The reviewed studies did not report on the effects of fertilizer use on water quality. Whether fertilization led to eutrophication or not could therefore not be answered based on primary research data. To determine the costs of potential eutrophication, the effects from irrigated lowland systems were modelled.

For data from a study in **irrigated lowland systems in Senegal** (Kanfany et al 2014), omitting mineral fertilizer made revenues drop from US$2,736 to US $1,880. The fertilizer input costs were US$72 per hectare. According to data from Krupnik et al (2012), farmers would not need to spend 1.60 mean person days per hectare at a unit cost of 2.29 Euros or 2.6 US dollars – US$ 4.16 in total.

For data from another study in Senegal (Rinaudo et al 1983), omitting mineral fertilizer made revenues drop from US$1,624 to US $904. The fertilizer input costs were US$20 per hectare. According to data from Krupnik et al (2012), farmers would not need to spend 1.60 mean person days per hectare at a unit cost of 2.29 Euros or 2.6 US dollars – US$ 4.16 in total.

When averaging the results from both studies in Senegal, omitting mineral fertilizer led to a decrease in revenues from US$ 2,134 to US $1,392. The fertilizer input costs would be reduced by US$46 per hectare, and labour costs for fertilizer application by US$ 4.16 per hectare. Eutrophication costs were zero.

If Senegal would no longer use mineral fertilizers in all its irrigated lowland systems, farmers would lose a total of US$ 70,185,780 in revenues. At the same time they would gain a total of US$ 4,351,140 of avoided costs for fertilizer inputs. They would also save a total of US$ 393,494 in labour costs for fertilizer application. Eutrophication costs were zero.

In the Senegalese cases, there were no eutrophication costs, either because there was no surplus of nutrients that could run off and /or because run-off nutrients did not reach the nearest fresh water body. Furthermore, Quiblier et al., 2008 state that application rates of N are generally low in Senegal and N-fertilizers are not the main cause of eutrophication in Western Africa.

For data from a study in the **Philippines** in **irrigated lowlands** (Haefele et al 2011), the average revenue for rice grain for one hectare of rice in was estimated to be US$1414 when mineral fertilizers were used and US$1359 when no fertilizer was added. There was no data detailing the amount of fertilizer used, and hence also no information to calculate eutrophication costs.

While there was no study which showed the effects of N-fertilzer use on eutrophication, a study in three provinces of the Northern Philippines recorded high levels of nitrates, sometimes exceeding international maximum acceptable levels in drinking wells. However, the study concluded that there were generally low nitrate concentrations in groundwater/drinking water in rice growing areas, which caused no risk for health. The authors argue that this probably due to the relatively large volatilization losses and fast chemical and microbial degradation under aerobic conditions in the tropics (Bouman et al 2002).

Also for the other three case study countries, there was no study that provided data on the effects of N and P-fertilization on water quality. However, in **Costa Rica**, researchers found that water running off from the rice paddies in the irrigated production systems contain high concentrations of P and N. Pérez-Castillo et al. (2013) monitored water quality of water running off from rice fields using an index based on temperature, pH, oxygen saturation percentage, electrical conductivity, biochemical oxygen demand, suspended solids, nitrate content and total Ps content. They confirmed that quality of water leaving the rice fields was poor with levels of N and P high enough to cause eutrophication of the bordering natural wetlands. However, this study did not go as far to actually measure eutrophication. The authors do suggest mitigation measures to reduce the eutrophication potential, though: Eutrophication from nutrient runoff alters the local wetland plant ecology, encouraging the growth of monocultures of cattail (Typha dominguensis). Dense cattail stands in adjoining wetland have been found to act as a buffer to absorb nutrients, especially P, thus reducing the nutrient load from entering into connecting waterways (Varnell et al., 2010).

In **California**, there is rather a problem with dissolved organic carbon than with N or P fertilizers: The legislation banning rice straw burning to mitigate its impacts on air quality may have led to an important win-win in the Sacramento region but also necessitated alternative forms of rice straw management. Incorporating this material into soils as a source of organic carbon has become one of the principle management options but provides two to three times the levels of dissolved organic carbon as contrasted to burning. However as there are no established water quality measures of dissolved organic carbon levels for California at the moment, thus no management strategies to control these levels have been recommended (UCCE, 2012).

### g. Reducing mineral fertilizer rates (IL)

As mentioned above, the reviewed studies did not report on the effects of fertilizer use on water quality. Whether fertilization led to eutrophication or not could therefore not be answered based on primary research data, but had to be modelled.

For data from a study in **irrigated lowland systems** in the **Philippines** (Kreye et al. 2009), the average revenue for rice grain for one hectare of rice in was estimated to be US$210 when the standard rate of mineral fertilizers (NPK) were used and US$98, when the fertilizer dose was reduced by 50 or 25%.

For other data from the same study (Kreye et al. 2009), the revenue for rice grain for one hectare of rice in was estimated to be US$246 on average, when mineral fertilizers were used (NPK) and US$72, when N-fertilizer was omitted. The fertilizer input costs were US$94 per hectare with NPK, and US$ 48 with PK only.

For data from another study in the Philippines (Corton et al. 2000), the average revenue for rice grain for one hectare of rice in was estimated to be US$3091 when the standard rate of mineral fertilizers were used and US$2992 when fertilizer use was reduced by 20 to 30%. The fertilizer input costs were US$80 per hectare with NPK, and US$ 60 when N was reduced.

When averaging the results from the three cases in the Philippines, the revenue for rice grain for one hectare of rice in was estimated to be US$817 for the standard rate of NPK and US$707 when fertilizer use was reduced. The fertilizer input costs were US$ 92 per hectare for the standard rate and US$ 71 with reduced inputs.

For data from a study in **Costa Rica** (Molina and Rodriguez 2012), the average revenue for rice grain for one hectare of rice in was estimated to be US$3757 when the standard rate of mineral fertilizers were used and US$3348 on average when fertilizer use was reduced by 25,50 or 75 %.

For data from a study in **Senegal** (Kanfany et al. 2014), the average revenue for rice grain for one hectare of rice in was estimated to be US$2736 when the standard rate of mineral fertilizers were used and US$2367 on average when fertilizer use was reduced by 25,50 or 75 %. Fertilizer inputs costs would be reduced from US$ 72 per hectare to US$ 36 per hectare.

In all of presented cases, there were no eutrophication costs, either because there was no surplus of nutrients that could run off and /or because run-off nutrients did not reach the nearest fresh water body.

If the Philippines were to reduce mineral fertilizers in the entire country in irrigated lowland systems as done in the pilot studies by Kreye et al (2009) and Corton et al (2000), the farming community would lose US$ 338,800,000 through yield reductions. At the same time they would save US$ 64,680,000 in fertilizer input costs.

If Costa Rica was to reduce mineral fertilizers in the entire country in irrigated lowland systems as done in the pilot study by Molina and Rodriguez (2012), the farming community would lose US$ 7,746,869 through yield reductions. At the same time they would save US$ 681,876 in fertilizer input costs.

If Senegal was to reduce mineral fertilizers in the entire country in irrigated lowland systems as done in the pilot study by Kafanay at (2014), the farming community would lose US$ 34,903,710 for yield reductions. At the same time they would save US$ 3,405,240 in fertilizer input costs.

### h. Mineral compared to organic fertilizer (IL)

There is still ongoing discourse whether organic fertilizers can reduce agricultural N and P losses to ground and surface water compared to mineral fertilizers. For the sake of this study, we made the assumption that mineral nutrient input, compared to organic nutrient input is more likely to result in contamination of water with nutrients, in particular with nitrates or ammonium. We base this assumption on a study by Kramer (2006) which showed that annual nitrate leaching was on average five times higher when mineral fertilizers were applied than when organic ones were used.

Decomposition of organic matter results in simpler inorganic N forms such as ammonium NH4+ and nitrate NO2-. These are soluble in soil water and readily available for plant uptake. The ammonium form is attracted and held by soil particles, so it does not readily leach through the soil with rainfall or irrigation water. Nitrates, on the other hand, are not attached to soil particles and do move downward with soil water and can be leached into groundwater or run off into surface waters.

In order to compare the effect on yield and water quality between mineral and organic fertilizer use, one would need comparators that depart from the same nutrient content. However, hardly any study compared exactly the same levels of nutrients from organic with nutrients from mineral fertilizer.

Furthermore, in the absence of primary data, one needs a nutrient run-off model that takes the assumptions made by Kramer (2006) into account, presuming that annual nitrate leaching is on average five times higher when mineral fertilizers are applied than when organic ones are used. Current models such as the one by described by Koch and Salou (2015) which is used for this analysis do not make a distinction between organic and mineral fertilizers however. We therefore decided to exclude this comparison from the analysis.

### i. Winter flooding compared to no winter flooding (IL)

Water logging makes P more soluble. In a study by Johnston et al (1965) soils under flooded rice cultivation lost 530 g P per hectare and year, compared with only 80 to 200 g p per year when lucerne was planted (a non-flooded crop).

This might make winter flooded rice paddies more likely to lead to eutrophication than non-winter flooded fields. However, in the absence of primary data on eutrophication, our eutrophication model does not account for the difference in solubility under waterlogged conditions, and hence we cannot make any conclusions in this regard.

## 3.2 Increase in rice yields versus reduction in water use

Worldwide, about 80 million hectares of irrigated lowland rice provide 75% of the world's rice production. This predominant type of rice system receives about 40% of the world's total irrigation water and 30% of the world's developed freshwater resources. The dependence on water of the rice farming sector is a huge challenge as freshwater resources are becoming increasingly depleted due to competing water uses from the residential and industrial sector and as rainfall is increasingly erratic due to climate change and variability. More efficient water use is therefore a must, yet it carries a number of trade-offs as this study has shown.

This study sought to assess and valuate trade-off resulting from improved water management on rice yields, on the one hand, and water consumption, on the other. The value of rice production as food for the farmer was estimated on the basis of the country specific revenue for rice grain received per tonne of paddy rice.

The Trucost water consumption methodology estimates a country specific monetary value for the health and ecosystem impact per cubic metre of water used for rice cultivation. These valuation coefficients were applied to the total water input to the rice field (water cost) where this was reported in the case study literature. To be more specific, the Trucost economic modelling methodology provides valuations for the impact of water consumption on ecosystems. The impacts on ecosystems due to water consumption included in this methodology are limited to the effect of water scarcity on net primary productivity of ecosystems. Net primary productivity is used as a proxy for ecosystem health. Country specific valuations are estimated based on the estimated water scarcity in each country (based on data from the World Resources Institute) and country specific values for the complement of ecosystems contained within each country.

The Trucost economic modelling methodology provides also valuations for the impact of water consumption on human health. The impacts on human health due to water consumption included in this methodology are limited to those linked to the lack of water for irrigation of agriculture for food production and the lack of access to safe water for consumption and sanitation. Country specific valuation coefficients are estimated based on the estimated water scarcity in each country (based on data from the World Resources Institute) and a global value per DALY lost or gained. This valuation coefficient has also been used to calculate the benefits to human health from any water that is returned to ecosystems from the rice field. Data was available for irrigated lowland systems (IL) and rainfed lowland systems (RL).

### a. Improved versus conventional irrigation management (IL)

Improved irrigation management has become a central topic for research. While the primary aim of improved irrigation management is to reduce water consumption, rice yields, where possible, should not be compromised. As the vote counting analysis of five case study countries has shown, in two third of all cases, yields were not affected when water consumption was reduced. However, in the remaining third, the results showed decreased yields in the vote-counting analysis (see figure 3.2.1 and related vote-counting project report). The outcome was strongly related to the type of improved water management that was practiced though. While Alternate Wetting and Drying (AWD) did not have an effect on yields in most cases, aerobic soils production showed to significantly compromise rice yields.

In terms of water savings, almost two third of all studies found that improved water management led to significant water savings, and a third of the studies found no large differences in water consumption compared to continuously flooded systems. This was explained by a range of context specific factors such as low water tables, low percolation rates and small water inputs, in the first place.

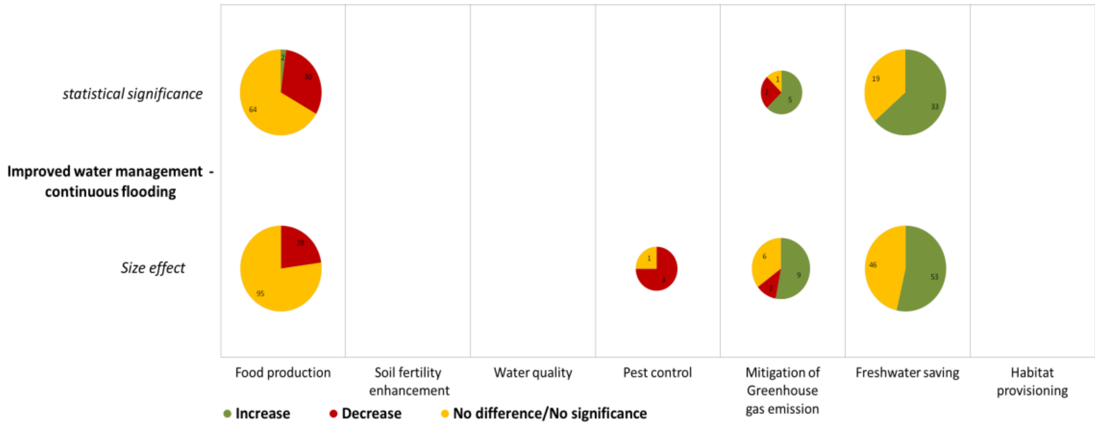


**Figure 3.2.1.** Results from the vote-counting analysis for improved water management versus continuous flooding.

One needs to be aware that water saving measures in these studies have been confined to field level. For comprehensive conclusions regarding water savings, one would need to implement a water accounting framework that includes measurements that go beyond the local level alone. None of the studies included in this analysis did such a comprehensive assessment.

Furthermore, one also needs to take the trade-offs into account that might be linked to water savings. For example, water saving regimes will increase the weed biomass as flood irrigation suppresses weeds. Our study showed that pest and weed control indeed decreased in the majority of all cases although the sample size was not large enough to draw any firm conclusions. On the positive side, GHG emissions tend to decrease with improved water management. Nonetheless, due to an insufficient sample size, this could not be demonstrated in this analysis. There was also not sufficient data that demonstrated the effects of improved water management on habitat provisioning, but evidence from other countries clearly shows effects on this response variable as natural habitat for aquatic organisms and water birds diminishes when continuous flooding is not provided.

When looking at country disaggregated data from several studies in irrigated lowlands systems (Boumann et al 2005, Peng et al 2010, Tabbal et al 2002, Lampayan et al 2014, Baghat et al 1999, Wassmann et al 2000, Bronson 1997, Belder 2004, Corton et al 2000, Wiangsamut et al 2013) the value of rice production in the **Philippines** was estimated to be US$2970 per hectare when fields were continuously flooded, and US$2605 when improved water management practices were applied. The monetary valuation for environmental and health related water consumption costs in the Philippines resulted in an average cost of US$337 for continuously flooded systems and US$199 for improved water management per hectare of rice production.

According to disaggregated data from Senegal (de Vries et al 2010), the value of rice production in **Senegal** was estimated to be US$3205 per hectare when fields were continuously flooded, and US$2,938 when improved water management practices were applied. The monetary valuation for environmental and health related water consumption costs in Senegal resulted in an average cost of US$784 for continuously flooded systems and US$554 for improved water management per hectare of rice production.

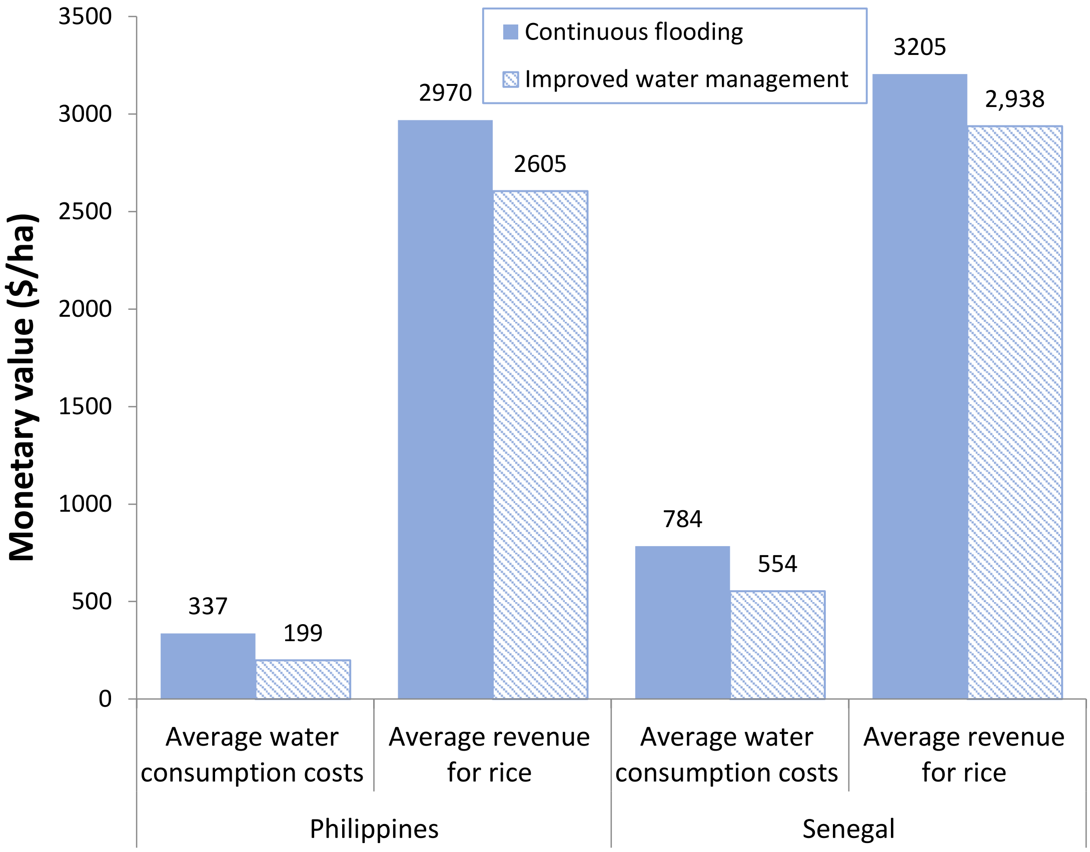


**Figure 3.2.2.** The figure shows the results of the monetary valuation of improved water management compared to continuous flooding. Values for the **revenue** for rice grain per hectare and the water consumption costs per hectare are given. Water consumption costs refer to costs for the environment and human health – not the actual costs of irrigation water – as further detailed in the introduction to this section.

If the **Philippines** were to change all their irrigated lowland systems (currently 70% of the entire rice growing area) from continuous to improved water management practices, they/the society would save US$ 425,040,000 in water consumption related health and environmental costs. At the same time, the rice producer community would lose a total of US$1,124,200,000 through yield losses. The loss of the farmer community would be almost three times the value of the water consumption externality costs.

If **Senegal** was to change all their irrigated lowland systems (currently 70% of the entire rice growing area) from continuous to improved water management practices, they/the society would save US$ 21,755,769 in water consumption related health and environmental costs. At the same time, the rice producer community would lose a total of US$25,255,530 through yield losses.

### b. System of Rice Intensification (SRI) (IL and IR)

The System of Rice Intensification (SRI) includes intermittent flooding as part of a production package. The vote counting analysis showed (see figure 3.2.3) that for irrigated lowland systems yields were not affected by the water saving regime when compared to conventional management with continuous flooding. In a few cases, yields even increased.

Regarding water consumption, the majority of studies did not find a difference between SRI and conventional systems, although the sample size of included treatment comparisons was not large enough to draw firm conclusions regarding SRI. Surprisingly, in a few cases, water use even increased when SRI was practiced.

For rainfed lowland systems, the vote-counting analysis showed that in half of the cases yields increased when SRI was implemented. In the other half, yields remained the same. Water saving was not recorded as rainfed systems are not irrigated.

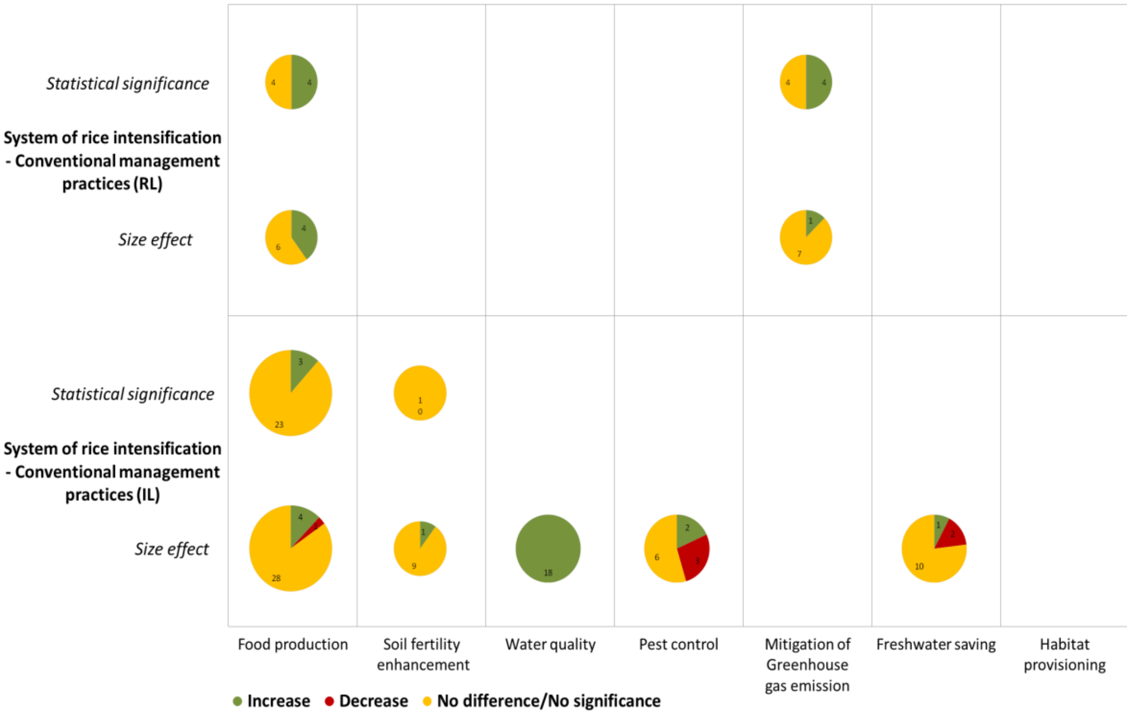


**Figure 3.2.3.** Results from the vote-counting analysis for SRI versus conventional management for RL and IL systems.

For country disaggregated data from **irrigated lowland systems** in **Senegal** (Krupnik et al 2010, Krupnik et al 2012a, Krupnik et al 2012b), conventional management led to a revenue of US$2302 per hectare, when and US$2422 when SRI was implemented. Water consumption for conventional management was on average 801US$ per hectare while it was slightly lower for SRI with US$626. While the produce price increased, the water consumption costs decreased when SRI as implemented – a clear win-win situation.

Data from Miyazato et al (2010) collected in **irrigated lowland systems** in the **Philippines** led to revenue for rice grainsrevenues of US$1124 per hectare when conventional management was practiced and US$1692 when SRI was implemented. Revenue for rice grainsRevenues therefore increased by almost 50% when SRI was practiced. Water consumption for both systems was not recorded in this study.

Data from Dumas-Johansen (2009), Koma (2002), Ly et al (2012), Ly et al (2013) and Satyanarayana et al (2007) collected in **rainfed lowland systems** in **Cambodia** led to a value of rice production of US$1099 per hectare when conventional management was practiced and

US$1422 when SRI was implemented. Water consumption for both systems was not recorded as there is no irrigation in rainfed systems.

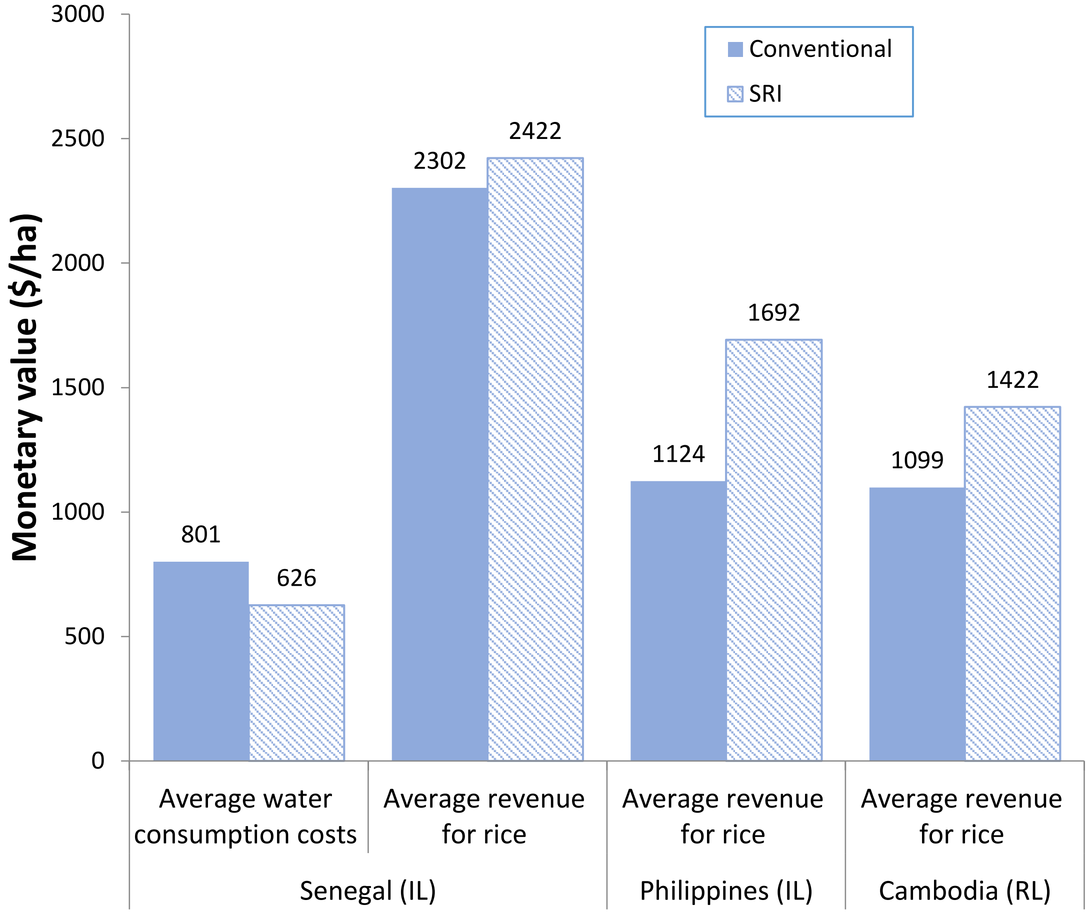


**Figure 3.2.4.** The figure shows the comparison of conventional management and SRI for Senegal, Philippines and Cambodia in terms of average revenue for rice and environmental and health costs of water consumption.

If **Senegal** was to change all its irrigated lowland systems (currently 70% of the entire rice growing area) from conventional management to SRI, the society would save US$ 11,350,800 in water consumption related health and environmental costs. At the same time, the rice producer community would gain a total of US$16,553,250 through yield increases.

If the **Philippines** were to change all their irrigated lowland systems from conventional management to SRI, the rice producer community would gain a total of US$ 749,760,000 through yield increases.

If **Cambodia** was to change all its rainfed lowland systems from conventional management to SRI, the rice producer community would gain a total of US$ 801,040,000 through yield increases.

### c. Direct seeding versus transplanting (IL)

In direct seeding, seeds are directly broadcasted in the rice field after land preparation. This decreases the total preparation and growing period of rice compared to traditional transplanted rice. In this way direct seeding has the potential to decrease water inputs. The data presented only shows research done in the **Philippines** for the irrigated lowland system (Corton et al 2000; Tabbal et al 2002). In two out of 10 cases (20%) an increase in water saving has been found when comparing direct seeding to transplanting, while no difference is shown for eight out of 10 cases (80%) (size effect). No statistical significant data is given on water use.

In 11 out of 12 (92%) cases no differences in yield have been found for direct seeding compared to transplanting rice (effect size). In the other case yield decreased (8%). Statistical data showed a significant decrease in yield in two out of two (100%) cases.

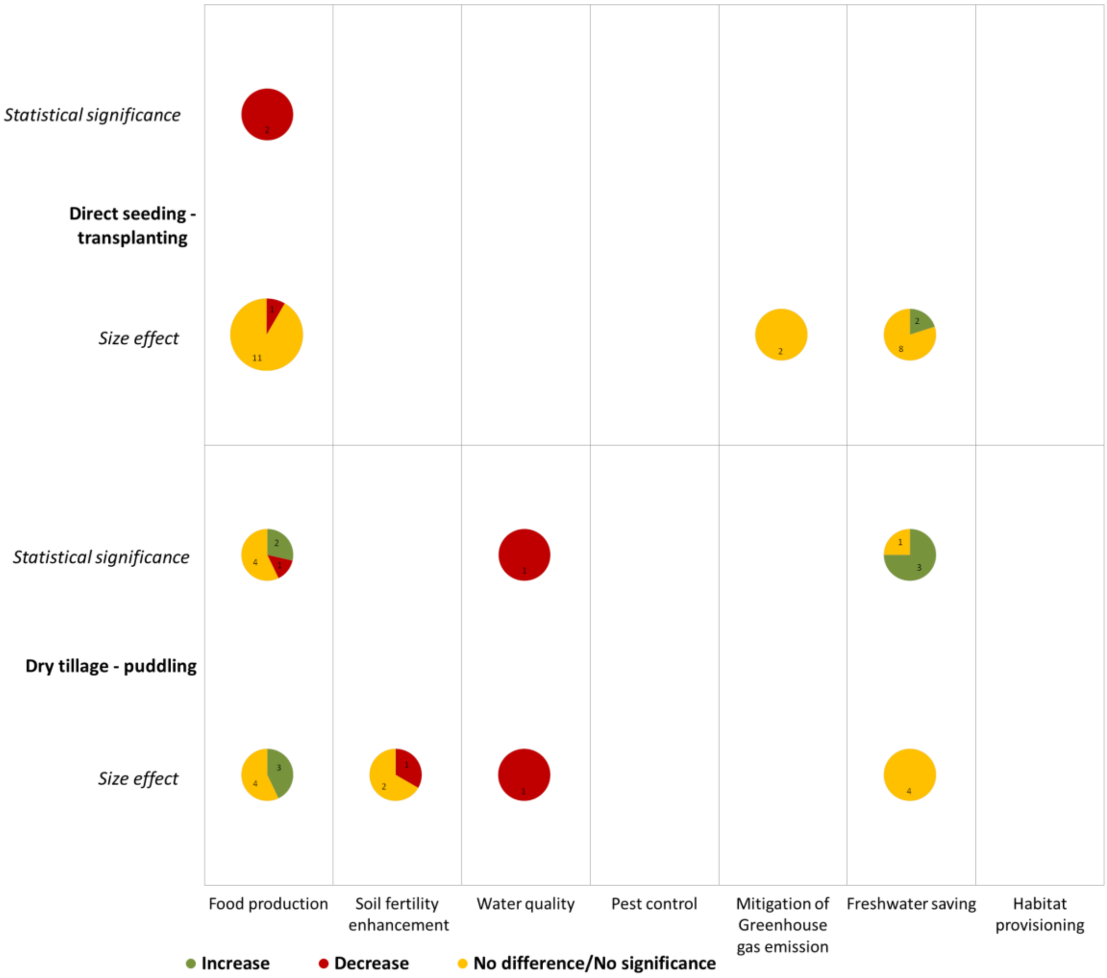


**Figure 3.2.5.** Results from vote-counting analysis showing the impact on rice yields and freshwater saving when direct seeding is practiced as compared to transplanting, and when the soil is dry tilled as compared to puddling.

When transplanting was practiced in **the Philippines,** this led to an average revenue of US$3047 per hectare and US$3340 when direct seeding was practiced. Health and environmental related water consumption costs for transplanting were on average 111US$ per hectare while they were lower for direct seeding with 93US$.

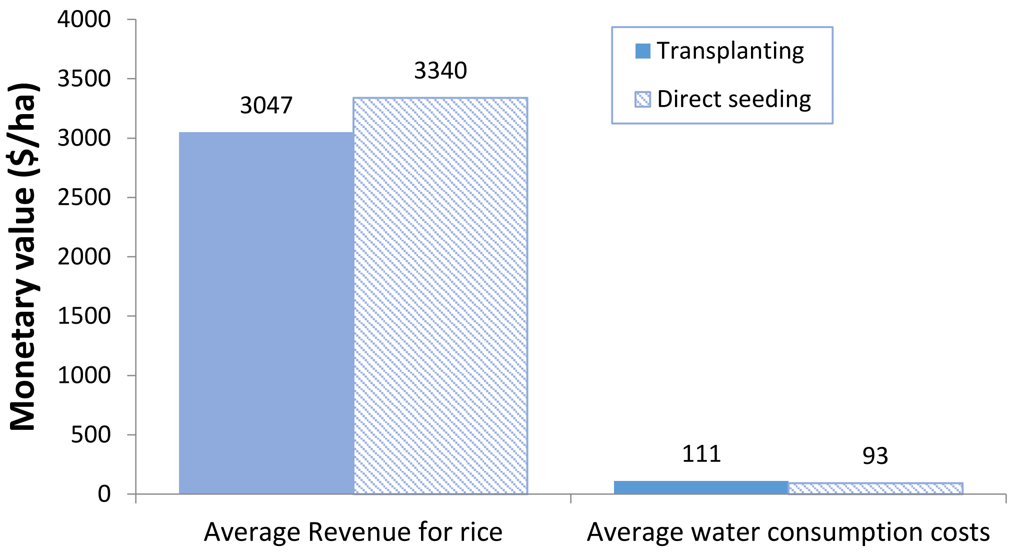


**Figure 3.2.6.** Comparison of effects of transplanting and direct-seeding on **revenue** for rice grain and water consumption costs in US$ per hectare.

If the Philippines were to change all their irrigated lowland systems from transplanting to direct seeding, the rice producer community would gain a total of US$902,440,000 through yield increases. At the same time, society would reduce health and ecosystem related water consumption costs by 55,440,000 – a clear synergy.

### d. Dry tillage – puddling (IL)

Research on dry tillage compared to puddling has been analyzed for Philippines (Sudir-Yadav et al 2014). Puddling is plowing the rice field under flooded conditions. Dry tillage has therefore the potential to safe water. According to statistical significant data, there was an increase in water saving in three out of four cases (75%) for dry tillage over puddling. In the other case (25%) no significant effect has been found. There was no significant decrease or increase in yield in four out of seven cases (57%). In two out of seven cases (29%) there was an increase in food production for dry tilled and in one out of seven cases a decrease.

When puddling was practiced in **the Philippines,** this led to an average revenue of US$2430 per hectare and US$2262 when dry tillage was practiced. Health and environmental related water consumption costs for puddling were estimated to be US$489 per hectare for puddled systems, and US$423 for those systems that were dry tilled.

If the Philippines were to change all their irrigated lowland systems from puddling to dry-tillage, the rice producer community would lose a total of US$ 517,440,000 through yield decreases. At the same time, society would reduce health and ecosystem related water consumption costs by 175,560,000 – a trade-off.

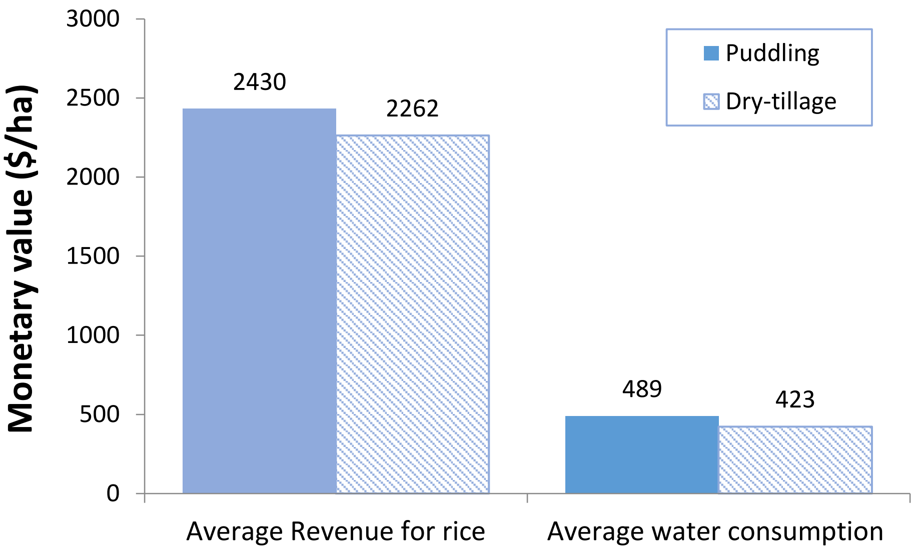


**Figure 3.2.7.** Comparison of effects of puddling and dry-tillage on **revenue** for rice grain and water consumption costs in US$ per hectare.

## 3.3 Increase in rice yields versus reduction of air pollution

Air pollution from rice agriculture can originate from rice residue burning, be it for disposal or for energy, and the use of mineral fertilizers and manure.

In Asia alone, 60% of the continent's 550 million tons of rice straw are being burnt in the field each year. Air pollution can be easily addressed by substituting straw burning with an alternative management practice. There are several alternative management options to burning residues, with effects on several response variables. Straw might be rolled with a heavy roller to crush the straw into the soil surface, it might chopped and then incorporated using a chisel plow or disc, or it can baled and removed.

Application of different chemical fertilizers and manures is a major source of ammonia ($NH_3$) emissions. The rate and total amount of $NH_3$ emission are related to different parameters such as climatic conditions, soil characteristics and kind of fertilizer. The use of manure has not been tested in any of the case study literature papers. The analysis is therefore restricted to emissions from mineral fertilizers.

This study sought to assess and valuate trade-off resulting from different types of residue and fertilizer management on rice yields, on the one hand, and air pollution, on the other.

The value of rice production as food for the farmer was estimated on the basis of the country specific revenue for rice grain received per tonne of paddy rice.

Quantification of the composition of rice straw combustion emissions was based on a study by Akagi et al (2011) which provides emissions factors for a range of biomass types, including crop residues which were assumed to be equivalent to rice straw. Where rice straw was reported to be burned for energy in the source studies, the total quantity of rice straw produced was valued.

The ecosystem impact valuations were based on continental scale modelling of the dispersion of emitted chemicals to rural air and a country specific valuation of the complement of ecosystems contained within each country that are affected by each pesticide or herbicide. The Trucost Air, Land and Water Pollutants methodology provides valuation coefficients for a range of organic and inorganic pollutants emitted to rural air. These coefficients are based on continent scale modelling of pollutant dispersion for organic pollutants and metals, and European modelling adjusted for population density for inorganic pollutants. Health effects are quantified as the number of DALYs lost per unit of emission and valued based on a global average value per DALY.

Ammonia emissions from rice fields per hectare were estimated based on the formulas developed by Koch and Salou (2015). The ammonia emission factors applied are not country or crop specific, but include specific emissions factors for different fertilizer types and are applied to the treatment specific mix of chemical fertilizers or the regional average mix of fertilizers.The Trucost Air, Land and Water Pollutants methodology provides valuation coefficients for the impact to human health of ammonia emissions to air. This valuation is based on a European study of the human health effects of air pollution and has been adjusted for each study country based on population density. A global value per DALY lost or gained has been used to value the health effects of ammonia emissions to air. Further detail on this methodology is provided in the Appendix.

Data from peer reviewed journal in relation to yields and emissions from air pollution was available for irrigated lowland systems (IL) in California, Senegal, Cambodia, Costa Rica and the Philippines.

### a. Incorporation, or rolling versus burning of rice straw (IL)

The results of the vote counting analysis showed that in the majority of all cases in **California** (Cintas & Webster 2001, Eagle et al 2000, Linquist 2006), yield was not affected when alternative crop residue management practices to rice straw burning were chosen and significant air pollution was avoided – a clear synergistic effect. However, this was only the case when sufficient mineral fertilizers were applied to the field. When no fertilizers were used, yield dropped by half. Under these conditions, incorporation of rice straw proved to bring the yields back up to almost maximum levels – however only after a couple of years.

These positive effects are clearly linked to the effect of rice straw incorporation or rolling on nutrient cycling and soil fertility. Residues incorporated or rolled into the soil increase microbial activity, they help to prevent erosion, positively affect soil structure and add carbon and organic matter to the soil. However, it takes time until these effects show as incorporating rice straw into the wet soil results in temporary immobilization of N through high C levels. In general, it needs to be noted that effects of rice straw addition on soil fertility are poorly studied – also this analysis did not have a large enough sample size to draw firm conclusions.

An important trade-off linked to changing from straw burning to other management practices is the incidence of pest and disease outbreaks. While straw burning is often used as a cost-effective pest and disease control practice, all other residue management strategies need alternative pest control mechanisms. California, for instance, reverted to residue incorporation in combination with winter flooding to suppress weed growth – which elsewise would have compromised yields.

There was no primary research data on air pollution related to straw burning, however potential emissions were modelled based on study a by Akagi et al (2011).

The average value of rice production was estimated to be US$3,668 per hectare when rice straw was burned, and US$3,711when rice straw was incorporated or US$3,677 when rolled into the soil. The rice straw nutrient value was estimated to be US$137 per hectare when straw was incorporated and US$136 when straw was rolled into the soil. The costs of air pollution related to rice straw burning were US$6,769, while rice straw incorporation and rolling – as expected - led to no air pollution costs from burning. Figure 3.1 shows the valuation results for the different straw treatments.

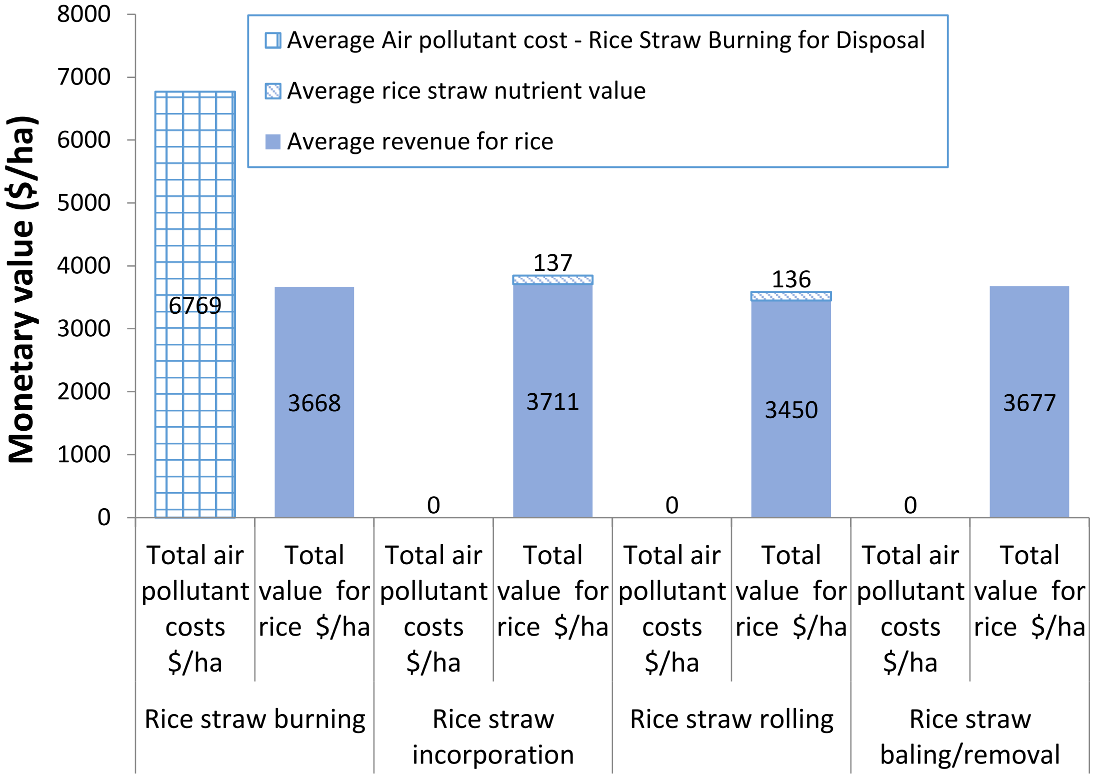


**Figure 3.3.1.** Valuation results comparing straw burning to incorporation, rolling and baling/removal in terms of value of rice production, the value of nutrients in rice straw and the costs of air pollutants from straw burning.

Air pollution costs related to rice straw burning are extremely high. The California rice straw-burning ban enacted in the 1990's put significant pressure on farmers to find alternative measures to remove or incorporate rice straw waste from farm fields. In the 1980's more than 95 percent of rice fields were burned as a means of reducing the highly resistant to decomposition rice straw. The area of rice straw burnt had a small but significant impact on risk of asthma hospitalization and on morbidity as studied in Butte country California (Jacobs et al 1997).

If California had continued to burn rice straw on 95 percent of their rice fields as done in the 80s, it would cause society environmental and health related air pollution costs of US$1,462,197,750.

When California changed to rice straw incorporation, the rice producer community gained a total of US$9,288,602 from yield increases while omitting the high air pollution costs – a win-win situation. The rice farmer community also gained US$29,593,918 from the nutrient value of rice straw when incorporated into the soil, which – as a positive externality - is currently not accounted for. One might argue that this benefit it already reflected in the revenue, as improving soil fertility increases yields over time. Yet apart from the indirect impact on yields, sustainably managed soils also contribute to the wider benefit of the society, especially future generations, who will farm the same soil in the future.

The situation looks different when straw is rolled into the soil. Compared to the rice straw burning scenario, the farming community would have lost US$ 47,091,052 if the straw had been rolled into the soil – a clear trade-off.

While there were no experiments (and data) in the other case study countries that compared rice straw burning to alternative management practices, there seems to be a large potential to decrease externality costs related to air pollution from straw burning in these countries. In the Philippines, which rice production area is 20 times the area of California, there is total production of rice straw is around 9 Million tons per year. Craig Jamieson from the International Rice Research Institute (personal communication) estimates that around 95% of rice straw in the Philippines is just burned in the field. While he does not have precise figures for what happens to the other 5%, he points to uses such as mushroom production, soil incorporation (especially in upland rice) and composting.

In Costa Rica, the majority of rice straw is incorporated. Burning of rice straw now requires a permit, which has reduced burning to about 15 -20%. Very little rice straw is used for forage (Personal communication, Roger Madriz, Research Director of the National Rice Cooperation (CONARROZ)).

In the Northern parts of Senegal, in the Senegal River Valley, 80 percent of the rice straw residues are burned. A reason for the burning of rice straw is that cattle are wandering around rice fields, releasing their dung in the fields. As dung contains seeds, cattle are seen as a major vector for dissemination of wild rice (which is considered a weed). Straw residues are burned, so cattle do not spent too much time in the field grazing on these residues. The remaining 20 percent of the rice straw residues are either fed to animals or buried in the field as fertilizer (UNFCCC, n.d.). However, other sources cite different numbers for the Senegal River Valley: 5% are burnt, 70% are sold for cattle feed, 25 % are left in the field for grazing cattle (Makhfousse, personal communication, 2015). In the Northern department of Podor, 2 % is burnt, 97 % are sold for cattle feed, and 1% is incorporated in the soil (Makhfousse, personal communication, 2015). The feed producers treat their rice straw with urea to improve the nutrition value. In the South of Senegal, in the Casamance region, almost no rice straw is burned. The residues are either left in the field for grazing or the straw is buried to improve soil fertility (UNFCCC, n.d.) The production of fodder is slowly starting; it is a good source of revenue for the farmers (Makhfousse, personal communication, 2015). 1 kg is sold for about 5 Euro cents, in times of high demand even for about 9 Euro cents (Makhfousse, personal communication, 2015).

### b. Removing and baling versus burning of rice straw (IL)

Removing and baling of rice straw is another promising practice that has found particular interest within the global discourse on bio-economy. Rice straw is often thought to be a free (waste) resource available to produce energy, be it for bioelectricity, biogas or even liquid fuels for transport. Ongoing studies clearly show however, that logistics are most likely to be too expensive to make rice straw energy a profitable business. Rice husks, on the other hand, have been shown to be a valuable resource when used directly at the milling sites – as raw material which can fuel part of the milling or rice-drying operations (see next section on energy from rice husks). Some integrated farming systems that have both crops and livestock, also rely on rice straw as bedding material or as (supplementary) animal feed, albeit of low nutritional value.

When comparing removing and baling compared to burning, there was no significant difference in yields between the different treatments in most cases according to the vote counting analysis (see vote counting document).

This is also reflected in the valuation exercise. The average value of rice production in California was estimated to be US$3,668 when fields were burned and US$3,450 when straw was removed.

The costs of air pollution related to rice straw burning were US$6,769 per hectare, while baling and removal led to zero air pollution costs. Costs might be generated through fossil fuel emissions when straw is transported to other sites for other uses.

As already illustrated in the previous section, of California had continued to burn rice straw on 95 percent of their rice fields as done in the 80s, it would cause society environmental and health related air pollution costs of US$1,462,197,750 per year.

As for yields – if California's rice farmers had baled and removed the straw in comparison with rice straw burning, the farming community would have gained US$ 1,944,126 – a clear trade-off.

### c. Using rice husks for energy production

There were no studies that calculated the air emissions associated with burning rice husk for energy. The results are therefore based on modelling. When taking the example from California based on Cintas & Webster (2001), Eagle et al (2000), and Linquist et al (2006) which was presented in the previous section, and assuming that additional to rice straw burning, also rice husks are burnt to provide energy, the air pollution costs increase considerably. Depending on the amount of rice husk produced per hectare, air pollutant costs from husk burning for energy were estimated to be US$1370 (rice straw incorporation scenario).

At the same time, the benefits also increase by providing rice husks as a renewable source of energy. The rice husk energy value was US$534 per hectare (from electricity).

If all Californian rice farmers had the same rice husk yields and if all of them were to use their entire rice husk for energy production, the gain would be US$ 121,663,214 when husks were used for electricity production. Air pollutant costs from rice husks combustion would amount to US$ 312,138,060.

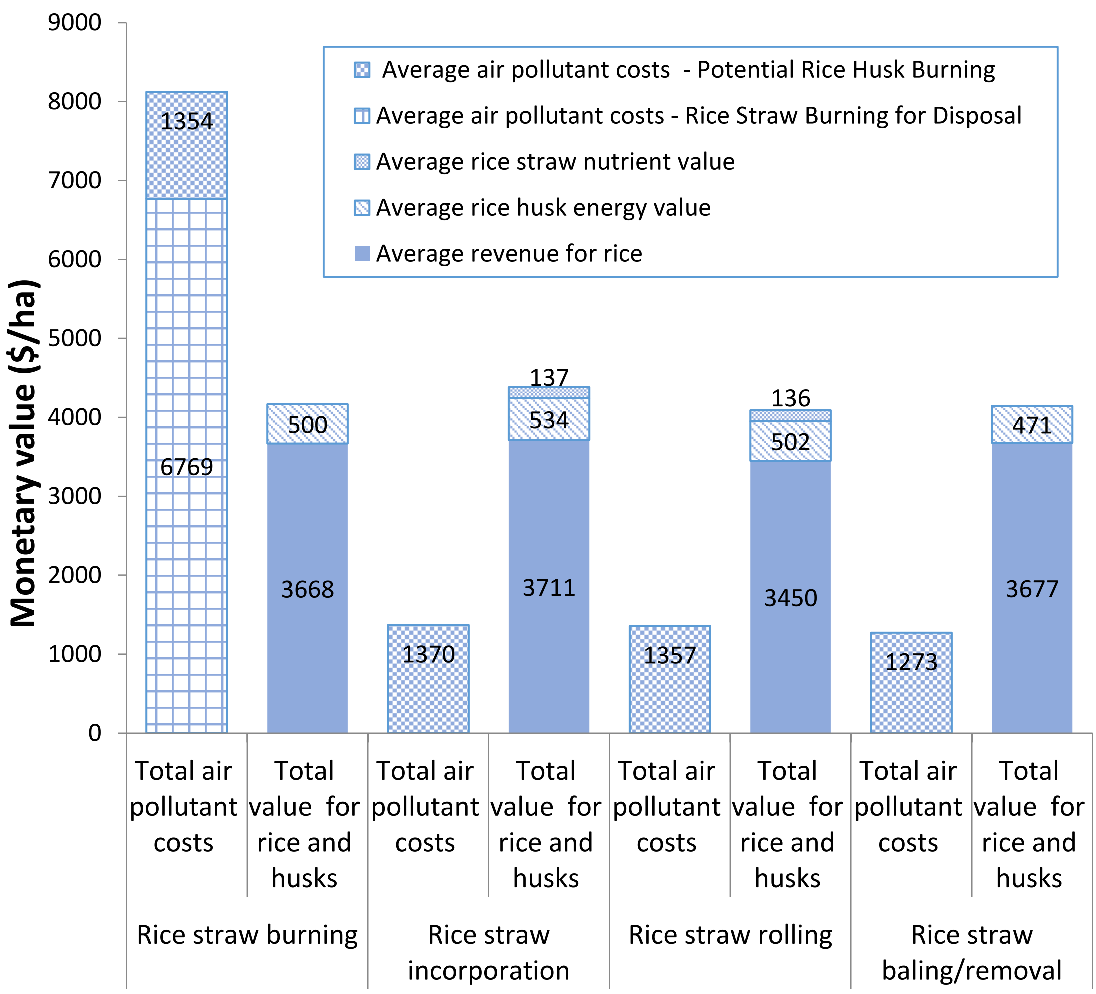


**Figure 3.4.1.** Valuation results comparing straw burning to incorporation, rolling and baling/removal in terms of benefits and costs for five variables: **Revenue** for rice in US$/ha, rice husk energy value US$/ha, rice straw nutrient value US$/ha, air pollution costs from rice straw burning for disposal in US$/ha and air pollution costs from rice husk burning for energy in US$/ha.

### d. Omitting mineral fertilizer use (IL)

There were no studies that recorded ammonia emissions from fertilizer applications. Emissions were modelled based on the Trucost Air, Land and Water Pollutants methodology as mentioned in the introduction.

For data from a study in **irrigated lowland systems in Senegal** (Kanfany et al 2014), omitting mineral fertilizer made revenues drop from US$2,736 to US $1,880. The fertilizer input costs were US$72 per hectare. Ammonia emission costs dropped by US$25 per hectare.

For data from another study in Senegal (Rinaudo et al, 1983), omitting mineral fertilizer made revenues drop from US$1,624 to US $904. The fertilizer input costs were US$20 per hectare. Ammonia emission costs dropped US$34 per hectare.

When averaging the results from both studies in Senegal, omitting mineral fertilizer led to a decrease in revenues from US$ 2,134 to US $1,392. The fertilizer input costs would be reduced by US$46 per hectare. Ammonia emission costs would drop by US$ 30 per hectare.

If Senegal would no longer use mineral fertilizers in all its irrigated lowland systems, farmers would lose a total of US$ 70,185,780 in revenues. At the same time they would gain a total of US$ 4,351,140 of avoided costs for fertilizer inputs.

Society would avoid costs from ammonia emissions worth US$ 2,837,700

For data from a study in the **Philippines** in **irrigated lowlands** (Haefele et al 2011), the average revenue for rice grain for one hectare of rice in was estimated to be US$1414 when mineral fertilizers were used and US$1359 when no fertilizer was added. There was insufficient data to calculate fertilizer input and ammonia emission costs.

If the Philippines would no longer use mineral fertilizers in all their irrigated lowland systems, farmers would lose a total of US$ 46,200,000in revenues.

### e. Reducing mineral fertilizer rates (IL)

For data from a study in the **Philippines** (Kreye et al. 2009), the average revenue for rice grain for one hectare of rice in was estimated to be US$221 when the standard rate of mineral fertilizers (NPK) was used and US$91, when the N-fertilizer dose was reduced by 100,50 or 25%. Accordingly, fertilizer input costs dropped from US$95 per hectare to US$67 per hectare.

Ammonia emission costs from the standard dose of mineral fertilizer were estimated to be US$215 per hectare, while reduced rates led to costs of US$88 per hectare.

If rice farmers in the Philippines would follow this example, society would avoid ammonia emission costs of $US 391,160,000. At the same time, rice farmer would lose $ 400,400,000 worth of yields, and gain $US 86,240,000 in N-fertilizer inputs.

For data from another study in the Philippines (Corton et al. 2000), the average revenue for rice grain for one hectare of rice in was estimated to be US$3091 when standard rates of mineral fertilizers (NPK) were used and US$2992 when N-fertilizer use was reduced by 20 to 30%. Accordingly, fertilizer input costs dropped from US$80 per hectare to US$60 per hectare.

Ammonia emission costs from the standard dose of mineral fertilizer were estimated to be US$200 per hectare, while reduced rates led to costs of US$135 per hectare.

If rice farmers in the Philippines would follow this example, society would avoid ammonia emission costs of $US 200,200,000. At the same time, rice farmer would lose $ 304,920,000 worth of yields, and gain $US 61,600,000 in N-fertilizer inputs.

For data from a study in **Costa Rica** (Molina and Rodriguez 2012), the average revenue for rice grain for one hectare of rice in was estimated to be US$3757 when the standard rate of mineral fertilizers were used and US$3348 on average when fertilizer use (either N, P or K) was reduced by 25, 50 or 75 %. Accordingly, fertilizer input costs dropped from US$123 per hectare to US$101 per hectare.

Ammonia emission costs from the standard does of mineral fertilizer were estimated to be US$33 per hectare, while reduced rates led to costs of US$24 per hectare.

If all rice farmers in irrigated lowland system in Costa Rica would follow this example, society would avoid ammonia emission costs of US$ 170,469. At the same time, rice farmer would lose US$7,746,869 worth of yields, and gain US$ 416,702 in fertilizer inputs.

For data from a study in **Senegal** (Kanfany et al. 2014), the average revenue for rice grain for one hectare of rice in was estimated to be US$2736 when the standard rate of mineral fertilizers (NPK) was used and US$2367 on average when fertilizer use (NPK) was reduced by 25, 50 or 75 %. Accordingly, fertilizer input costs dropped from US$72 per hectare to US$36 per hectare.

Ammonia emission costs from the standard dose of mineral fertilizer were estimated to be US$25 per hectare, while reduced rates led to costs of US$13 per hectare.

If all rice farmers in irrigated lowland system in Senegal would follow this example, society would avoid ammonia emission costs of US$ 1,135,080. At the same time, rice farmers would lose US$ 34,903,710 worth of yields, and gain US$ 3,405,240 in fertilizer inputs.

## 3.4 Increase in rice yields versus the production of energy from rice husks

While there were no peer reviewed studies that explicitly looked at the relation between yield increases and energy production from rice husks, one can assume that there is no trade-off between the two. While researchers experiment with the integration of rice husks into the soil to increase soil organic matter (personal communication, Rodenburg) and ultimately yields, husks are most often burnt at the milling site. In some instances however, rice husks are used to fuel part of the milling or rice-drying operations, or sold to the industry.

In Costa Rica, the Research Director of the National Rice Cooperation (CONARROZ), Roger Madriz, stated that between 60 to 65% of the rice husks are used for energy generation (personal communication). The cement producer CEMEX, for instance, uses rice husks as fuels for its production. They successfully implemented a United Nations-certified Clean Development Mechanism (CDM) that substitutes conventional fossil fuels such as petroleum coke, fuel oil, and natural gas with more environmentally friendly local biomass products such as rice husks and wood chips. The biomass is now used in cement kilns to serve as fuel during calcination, an important component of the cement-making process in which raw materials are heated to temperatures of nearly 1,500 degrees Celsius.

In the absence of data on rice husks use, we modelled the benefits and costs of rice husk use. The value of rice husk per tonne as a substitute for energy is described in the Trucost methodology document. According to IRRI (n.d.) rice husk has a high average calorific value of 3410 kcal/kg and therefore are a good, renewable energy source. However, because of the high silica contents rice husk is very abrasive and wears conveying elements very quickly.

Assuming that one can obtain one ton of rice husk from five tons of rice grain, and the energy equivalent of one ton of rice husk is around 500kwh, one can calculate the value based on the electricity cost. In the United States, one ton of husk used for electricity production was estimated to be worth US$67, in Costa Rica US$95, in Cambodia US$170, in Senegal US$118 and in the Philippines US$110.

Based on average yield data in each country, and assuming each of the case study countries would use their rice husk for electricity production, the energy value of rice husk per country and year would be: US$ 347,820,000 for Cambodia, US $4,022,190 for Costa Rica, US$ 343,200,000 for the Philippines, US$ 12,437,273 for Senegal, and US$ 29,003,777 for California.

If countries would use their entire rice husk for energy, this would add between 3 and 30% of value on top of the revenue for rice grain for rice:

- In Cambodia, rice farmers would gain an additional 30%.
- In Costa Rica, rice farmers would gain an additional 3%.
- In the Philippines, rice farmers would gain an additional 12%.
- In Senegal, rice farmers would gain an additional 12%.
- In California, rice farmers would gain an additional 3%.

**Table 3.4.1.** Rice production overview for the five case study countries

| | Average paddy yield (t/ha) | Average rice husk yield (t/ha) | Rice production area (ha) | Total rice production (t) | Total rice husk production (t) | Value of total rice production (US$) | Value of total rice husk production (US$) |
|---|---|---|---|---|---|---|---|
| Cambodia | 3.3 | 0.66 | 3,100,000 | 10,230,000 | 2,046,000 | 1,142,300,000 | 347,820,000 |
| Costa Rica | 3.8 | 0.76 | 55,709 | 211,694 | 42,339 | 136,663,434 | 4,022,190 |
| Philippines | 3.9 | 0.78 | 4,000,000 | 15,600,000 | 3,120,000 | 2,948,400,000 | 343,200,000 |
| Senegal | 3.9 | 0.78 | 135,129 | 480,203 | 105,401 | 102,337,440 | 12,437,273 |
| California | 9.5 | 1.9 | 227,838 | 2,164,461 | 432,892 | 1,006,474,365 | 29,003,777 |

## 3.5 Increase in rice yields versus reduction of GHG emissions

Global estimates attribute about 89 percent of rice global warming potential to $CH_4$ emissions which are due to flooding practices in irrigated and rainfed lowland systems (Linquist et al, 2006). To a much smaller degree, the production and application of N-fertilizers contributes to the rice global warming potential. And also emissions from rice straw burning impact global climate change. In addition to rice production being a major emitter of GHGs, rice systems also sequester carbon via soil organic carbon in top soil. Yet overall, rice production is a net producer of greenhouse gas emissions.

This study sought to assess and valuate the trade-off resulting from water management, residue management, fertilizer application and the choice of rice varieties on rice yields, on the one hand, and GHG emissions, on the other. The value of rice production was estimated on the basis of the country specific revenue for rice grain received per ton of paddy rice.

Primary data on GHG emissions as reported in the peer reviewed studies was used to model the GHG emission costs. The cost of GHG emissions was valued following the Trucost Greenhouse Gas methodology which provides a valuation coefficient for $CO_2$ equivalent emissions based on the social cost of carbon.

When no primary data was available, Trucost modelled potential GHG emissions by estimating nitrous oxide volatilization from chemical fertilizers, and methane emissions from rice fields. The burning of crop residues is not generally considered a net source of $CO_2$ emissions as the carbon released is assumed to be reabsorbed in the next growing season, however residue burning is considered a source of non-$CO_2$ greenhouse gases (IPCC, 2006). The quantity of non $CO_2$ greenhouse gas emissions released per tonne of crop residue burned was therefore also estimated. The estimations are based mainly on global assumptions. While they give a general idea of the order of magnitude of emissions in relation to other benefits and costs from rice production, they have to be treated with caution when comparing the differences between management practices and systems as they are not practice specific.

Data was available for irrigated lowland systems (IL) and rainfed lowland systems (RL).

### a. Improved versus conventional irrigation management (IL)

Practices that reduce water consumption and hence reduce the flooding time of irrigated lowland systems tend to reduce GHG emissions. Maintaining the same yields under reduced flooding times is a challenge, but has been successful for some of the tested practices.

Under alternate wetting and drying (AWD) fields are flooded and the water is left to evaporate and to infiltrate the soil until a critical level. Under aerobic soils, also known as aerobic monocultures, the crop is usually dry direct seeded and soils are kept aerobic throughout the growing season. Supplementary irrigation is applied as necessary and adapted rice cultivars that are responsive to fertilizers and with higher yield potential than upland rice varieties are used (Kreye et al 2009). When considering both of these improved water management practices together, more than half of all cases led to reduced GHG emissions - for statistically significant data, 5 out of 9 cases (56%) showed a reduction in GHG emissions, 1 showed no difference (11%) and 2 (33%) showed an increase in GHG emissions.

Estimates on the effect on yields vary. Using improved water management practices, in 64 out of 96 (67%) cases, yields remained the same, while in 32 cases (33%) yields decreased according to statistically significant data. The decrease is mostly due to the aerobic rice systems as a long-term experiment in the Philippines shown. Aerobic rice yields were consistently lower than in conventional, flooded rice, and yield differences increased over eight seasons of continuous cropping (Peng et al 2006). Yield failures, or zero harvest, occur occasionally and were attributed to 'soil sickness': potentially the combined effect of allelopathy, nutrient depletion, buildup of soil-borne pests and diseases and soil structural degradation (Ventura & Watanabe 1978).

When looking at country disaggregated data from several studies in irrigated lowlands systems (Boumann et al 2005, Peng et al 2010, Tabbal et al 2002, Lampayan et al 2014, Baghat et al 1999, Wassmann et al 2000, Bronson 1997, Belder 2004, Corton et al 2000, Wiangsamut et al 2013) the value of rice production in the **Philippines** was estimated to be US$2970 per hectare when fields were continuously flooded, and US$2605 when improved water management practices were applied.

There was no primary research data for GHG emissions in this context. The difference in GHG emission costs between the two treatments can therefore not be calculated. When applying the TRUCOST model to calculate *potential* GHG emissions from nitrous oxide volatilization of chemical fertilizers, and methane emissions from rice fields, average GHG emissions costs were in the range of US$590 per hectare. This value was obtained for both treatments since the model is not detailed enough to differentiate between different types of irrigation management, and can therefore only show the order of magnitude, not the difference between the practices.

According to disaggregated data from Senegal (de Vries et al 2010), the value of rice production in **Senegal** was estimated to be US$3205 per hectare when fields were continuously flooded, and US$2,938 when improved irrigation management practices were applied.

There was no primary research data for GHG emissions in this context. The monetary valuation for modelled GHG emissions in Senegal resulted in an average cost of US$527 for both practices, a figure that can only be used to indicate the order of magnitude for emissions from rice fields in this context.

If the **Philippines** were to change all their irrigated lowland systems (currently 70% of the entire rice growing area) from continuous to improved irrigation management practices, the rice producer community would lose a total of US$1,124,200,000 through yield losses. No data on the difference between practices in GHG emissions is available.

If **Senegal** was to change all their irrigated lowland systems (currently 70% of the entire rice growing area) from continuous to improved irrigation management practices, the rice producer community would lose a total of US$25,255,530 through yield losses. No data on the difference between practices in GHG emissions is available.

### b. System of Rice Intensification (SRI) versus conventional water management (IL and RL)

The System of Rice Intensification (SRI) includes intermittent flooding as part of a production package. The system advises transplanting of young (eight to ten days old) single rice seedlings, with care and spacing, and applying intermittent irrigation and drainage to maintain soil aeration. In addition the use of a mechanical rotary hoe or weeder to aerate the soil and control weeds is encouraged.

In the SRI system, 'intermittent flooding' - irrigation to field capacity and managing high soil moisture without anaerobic conditions is managed through visual inspection of soil and attempts to maintain a moist soil surface. However, such flooding regimes are prone to yield losses where water is not carefully monitored and particularly at the vulnerable rice flowering stage. In general, yields are the same with SRI as under continuously flooded systems or may even increase. For statistically significant data, yields remained the same in 23 out of 26 cases (88%), and increased in the remaining three (12%). There was no primary research data on GHG emissions.

For country disaggregated data from **irrigated lowland systems** in **Senegal** (Krupnik et al 2010, Krupnik et al 2012a, Krupnik et al 2012b), conventional management led to a revenue of US$2302 per hectare, when and US$2422 when SRI was implemented.

As primary research data for GHG emissions for SRI and conventionally managed systems was not available, GHG emissions were modelled. GHG emission costs were estimated to be US$1352 per hectare for conventional systems and US$ 1383 for SRI.

If **Senegal** was to change all its irrigated lowland systems (currently 70% of the entire rice growing area) from conventional management to SRI, the society would have to pay US$ 2,932,290 in GHG emission costs. At the same time, the rice producer community would gain a total of US$16,553,250 through yield increases.

These modelled results have to be questioned however. Primary data from GHG emission tests in SRI systems have usually shown the opposite –SRI does decrease emissions when compared to conventional systems. However, these overall emission reductions stem from reduced water inputs and hence reduced methane emissions, while nitrous emissions are usually higher due to the use of organic fertilizer such as rice straw (for instance see Gathorn-Hardy et al n.d.). The GHG emission model used here does not distinguish between different amounts of water input however, and therefore only displays the GHG emission increases from nitrous oxides/rice straw.

Data from Miyazato et al (2010) collected in **irrigated lowland systems** in the **Philippines** led to revenue for rice grainsrevenues of US$1124 per hectare when conventional management was practiced and US$1692 when SRI was implemented. Revenues therefore increased by almost 50% when SRI was practiced. GHG emission costs were estimated to have an order of magnitude of US$563 per hectare.

While the concept of SRI was originally developed under irrigated conditions, these systems have also been adapted to **rainfed lowland (RL)** paddies. The SRI in lowland rainfed systems differ from the conventional management system in several parameters, but the focus of included research studies is on modified water and nutrient management. In these studies, SRI fields are moist during transplanting and drained several times during the growing season. Trade-offs are likely to occur between $CH_4$ emissions when the fields are flooded and $N_2O$ emissions when fields are drained.

There was only very limited data on SRI in RL, all of which stem from **Cambodia**. Therefore, results have to be treated with caution. When comparing SRI to conventional management, there was no significant difference in GHG emissions in 50% of all cases and an increase in mitigation potential (and a decrease in emissions) in the other 50%. In terms of yield, there was no significant difference in yield in 50 % of the cases and an increase in the other 50%.

Data from Dumas-Johansen (2009), Koma (2002), Ly et al (2012), Ly et al (2013) and Satyanarayana et al (2007) collected in rainfed lowland systems in Cambodia led to a value of rice production of US$1099 per hectare when conventional management was practiced and US$1422 when SRI was implemented.

For this system comparison in Cambodia, primary research data for GHG emissions was available. The monetary valuation for GHG emissions in Cambodia's RL paddies resulted in an average cost of US$690 for conventionally managed systems and US$586 for SRI per hectare of rice production – a reduction in costs of 15%.

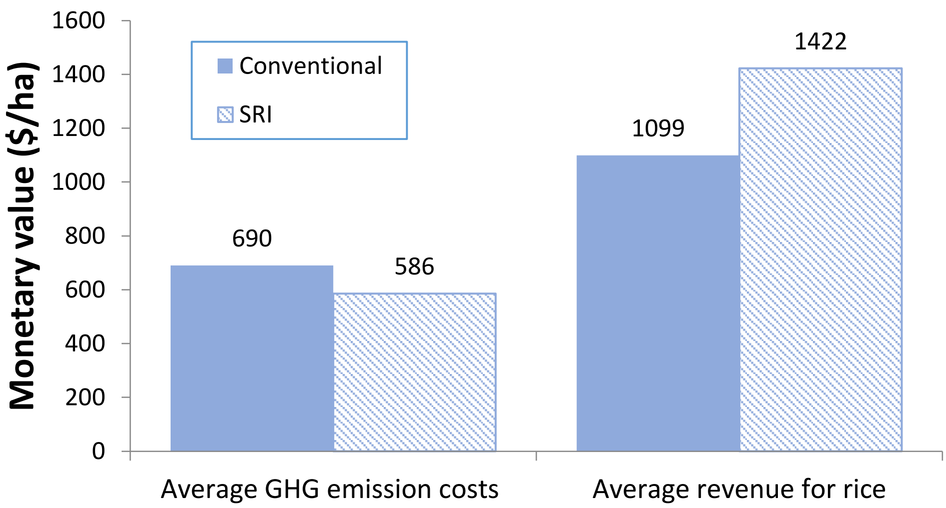


**Figure 3.5.1.** Valuation results comparing conventional management to SRI in terms of GHG emission costs and rice grain revenues in rainfed lowland systems in Cambodia. Monetary valuation for GHG emissions is based on primary research data.

If all rice farmers in rainfed lowland systems in Cambodia would change to SRI, they would increase the produce price value of rice by US$ 801,040,000. At the same time, society would have to spend US$ 257,920,000 less in GHG emission costs.

### c. Rice straw burning compared to straw incorporation (IL)

The $CO_2$ emitted through straw burning has less impact on climate change than the $CH_4$ emissions from rice cultivation (McCarty 2011) since $CH_4$ has 21 to 34 times the Global Warming Potential (GWP) of $CO_2$. Data on GHG emissions from rice straw burning and alternative residue management practices all from came from studies in **California** in **irrigated lowland systems (IL)** (Cintas and Webster 2011; Eagle et al 2000; Linquist et al 2006). When comparing straw incorporation with burning in irrigated lowland systems, GHG emissions declined in 4 out of 8 (50%) cases when straw was incorporated, and remained the same as in the case of burning in the other 4 (50%). No statistically significant data for GHG emissions was available for this practice comparison.

Incorporation of straw showed no difference in yield in 9 out of 14 cases (64%) for statistically significant data. In 4 cases out of 14, there were reduced yields when straw was incorporated as compared to burning it (29%). In one case, there was an increase in yields (7%). The decrease in yield may be attributable to allelopathy, as found by Pheng et al (2010) or to the phytotoxins present in rice residues (e.g. Bacon and Cooper 1985).Overall, the average yield was higher when straw was incorporated however. The mean value of rice production was estimated to be US$3,668 when straw was burned and US$3,711 when straw was incorporated.

There was no primary research data to estimate the average GHG emission costs in **California** for these experiments. When the model was used to estimate emissions rice straw burning led to average costs of US$863 and US$1,419 when straw was incorporated. GHG emission costs for straw incorporation were therefore significantly higher than when burnt, which might be surprising but can be explained by the high GWP of $CH_4$.If rice farmers in California would incorporate their rice straw in the entire production area, they would gain a total of US$ 9,797,034 compared to the straw burning scenario. They would cause a total of US$ 126,677,928 GHG emission costs to society, however.

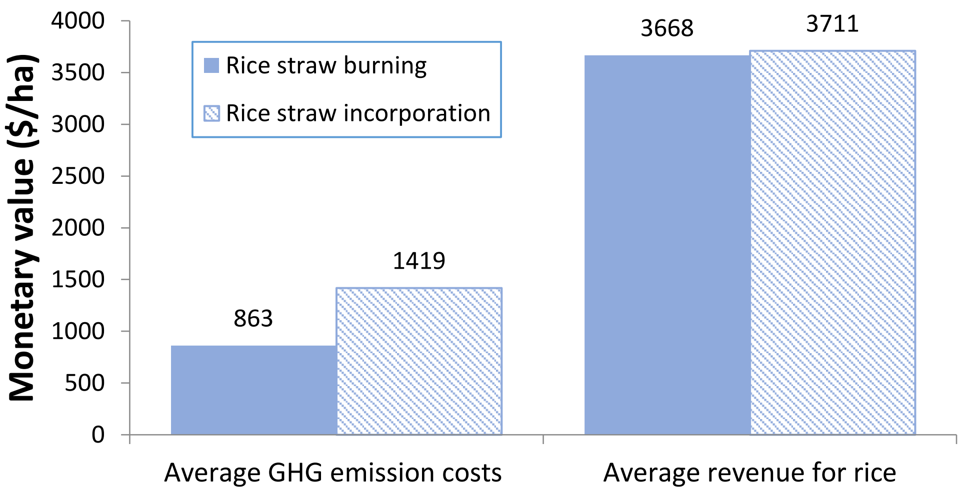


**Figure 3.5.2.** Valuation results comparing rice straw burning to rice straw incorporation in terms of GHG emission costs and average revenue for rice in irrigated lowland systems in California. Monetary valuation for GHG emissions is based on modeled data.

**d. Reducing the rate of synthetic fertilizer application (IL)**

Soil nitrogen additions can increase both the food production potential and the climate change impact of rice production. In addition to the global warming potential of synthesizing nitrogen fertilizers, increased net primary productivity of crop and weed biomass impacts the amount of carbon stored in dry cultivation systems and the amount of methane produced in flooded systems.

For data from a study in the **Philippines** (Kreye et al. 2009), the average revenue for rice grain for one hectare of rice in was estimated to be US$221 when the standard rate of mineral fertilizers (NPK) was used and US$91, when the N-fertilizer dose was reduced by 100,50 or 25%. Accordingly, fertilizer input costs dropped from US$95 per hectare to US$67 per hectare.

The modeled results for GHG emission costs for higher fertilizer inputs were $US594 and for lower inputs $US586.

If rice farmers in the Philippines would follow this example, society would avoid GHG emission costs of $US 24,640,000. At the same time, rice farmer would lose $ 400,400,000 worth of yields, and gain $US 86,240,000 in N-fertilizer inputs.

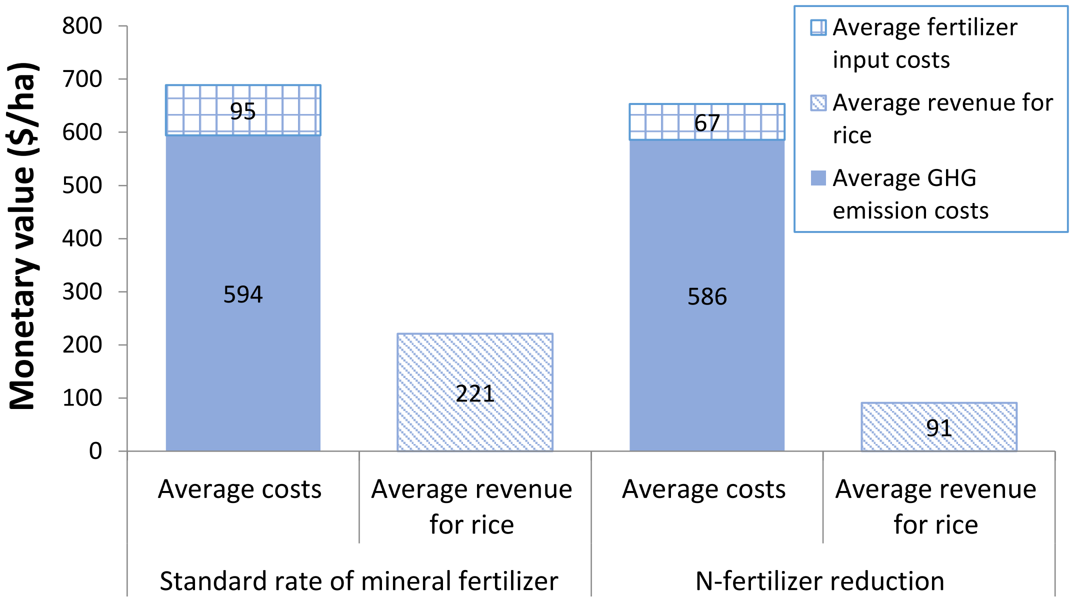


**Figure 3.5.3.** Valuation results comparing a reduction in fertilizer use in terms of GHG emission costs, fertilizer input costs and average revenue for rice in irrigated lowland systems in the Philippines. Monetary valuation for GHG emissions is based on modeled data.

For data from another study in the Philippines (Corton et al. 2000), the average revenue for rice grain for one hectare of rice in was estimated to be US$3091 when standard rates of mineral fertilizers (NPK) were used and US$2992 when N-fertilizer use was reduced by 20 to 30%. Accordingly, fertilizer input costs dropped from US$80 per hectare to US$60 per hectare.

This was reflected in the GHG emission costs. The modelled results for higher fertilizer inputs were US$593 and for lower inputs US$587.

If rice farmers in the Philippines would follow this example, society would avoid GHG emission costs of $US 18,480,000. At the same time, rice farmer would lose $ 304,920,000 worth of yields, and gain $US 61,600,000 in N-fertilizer inputs.

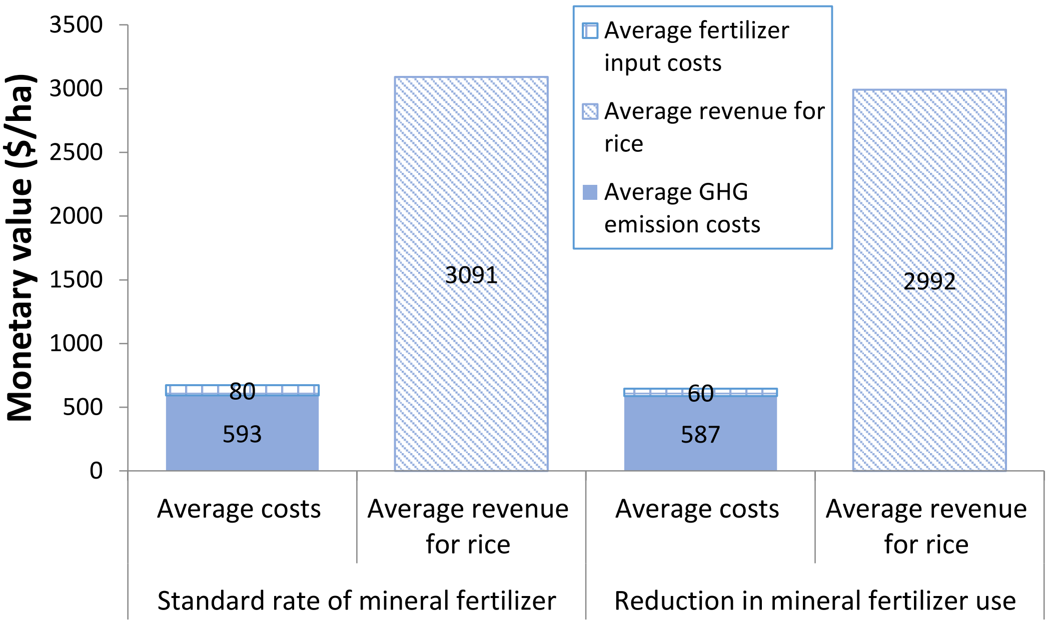


**Figure 3.5.4.** Valuation results comparing a reduction in fertilizer use in terms of GHG emission costs, fertilizer input costs and average revenue for rice in irrigated lowland systems in the Philippines. Monetary valuation for GHG emissions is based on modeled data.

For data from a  study in **Costa Rica** (Molina and Rodriguez 2012), the average revenue for rice grain for one hectare of rice in was estimated to be US$3757 when the standard rate of mineral fertilizers were used and US$3348 on average when fertilizer use (either N, P or K) was reduced by 25, 50 or 75 %.  Accordingly, fertilizer input costs dropped from US$123 per hectare to US$101 per hectare.

There was no primary research data on GHG emissions. Modelled results showed a slight decrease from high to low fertilizer inputs from US$195 to US$191.

If all rice farmers in irrigated lowland system in Costa Rica would follow this example, society would avoid GHG emission costs of US$ 75,764. At the same time, rice farmer would lose US$7,746,869 worth of yields, and gain US$ 416,702 in fertilizer inputs.

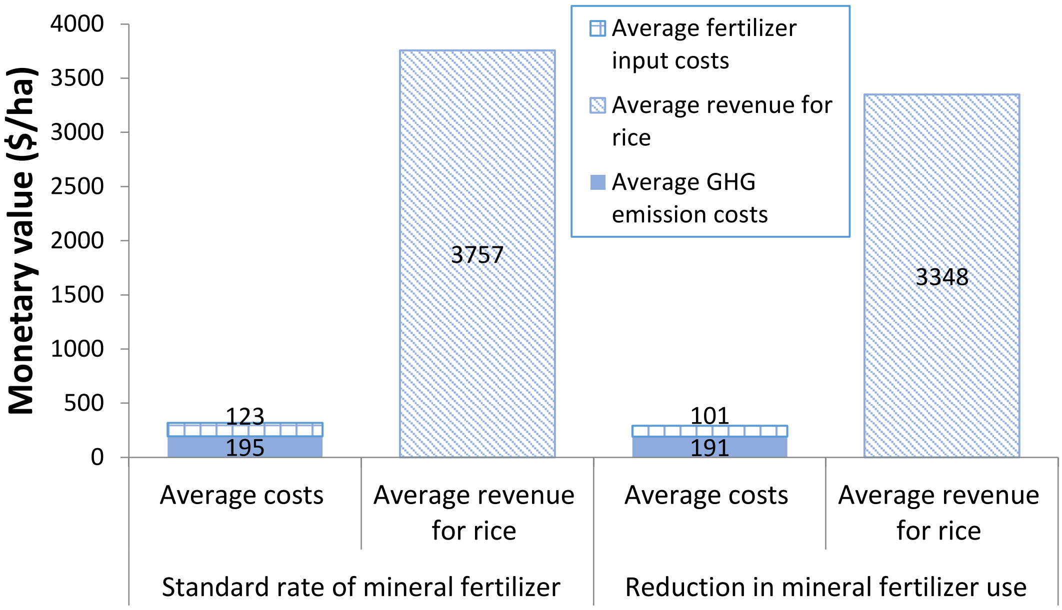


**Figure 3.5.5.** Valuation results comparing a reduction in fertilizer use in terms of GHG emission costs, fertilizer input costs and average revenue for rice in irrigated lowland systems in Costa Rica. Monetary valuation for GHG emissions is based on modeled data.

For data from a study in **Senegal** (Kanfany et al. 2014), the average revenue for rice grain for one hectare of rice in was estimated to be US$2736 when the standard rate of mineral fertilizers (NPK) was used and US$2367 on average when fertilizer use (NPK) was reduced by 25, 50 or 75 %. Accordingly, fertilizer input costs dropped from US$72 per hectare to US$36 per hectare.

There was no primary research data on GHG emissions. Modelled results showed a slight decrease from high to low fertilizer inputs from US$527 to US$519.

If all rice farmers in irrigated lowland system in Senegal would follow this example, society would avoid GHG emission costs of US$ 756,720. At the same time, rice farmer would lose US$ 34,903,710 worth of yields, and gain US$ 3,405,240 in fertilizer inputs.

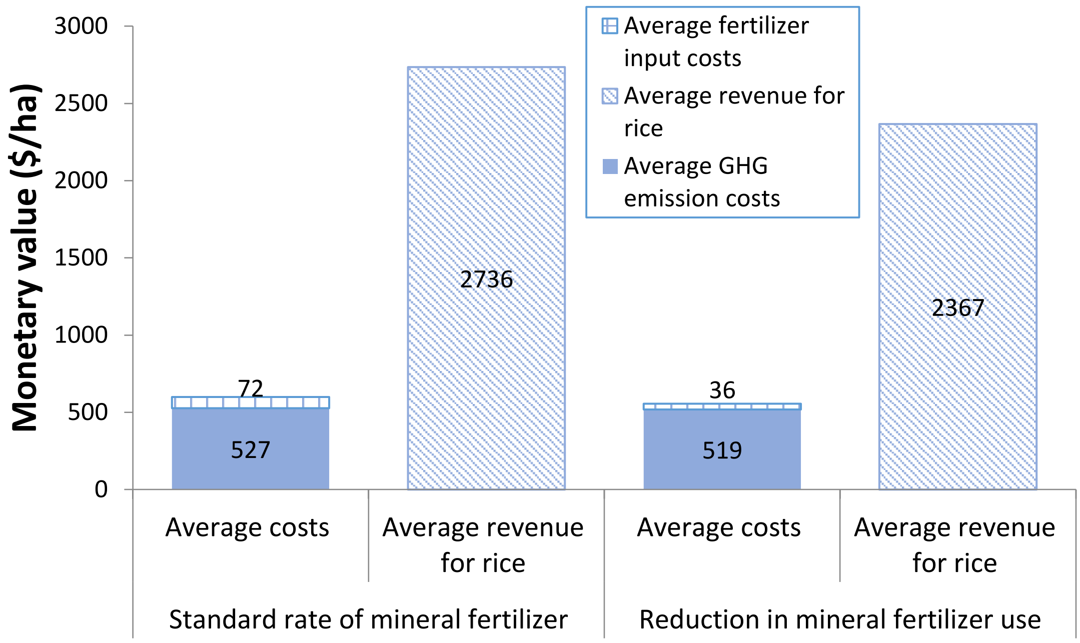


**Figure 3.5.6.** Valuation results comparing a reduction in fertilizer use in terms of GHG emission costs, fertilizer input costs and average revenue for rice per hectare in irrigated lowland systems in Senegal. Monetary valuation for GHG emissions is based on modeled data.

### e. Comparing synthetic fertilizer with a mix of organic and mineral fertilizer (IL)

When comparing the difference between synthetic fertilizer alone with a mix of organic and mineral fertilizer, the majority of cases did not show an effect (73% of all cases) and in the remaining cases showed a decrease in yield (27%). The yield decrease may be attributed to the high C/N ratio when carbonized rice husk was added to the system (Haefele et al 2011) or to phytotoxings present in rice residues (Pheng et al 2010).

In the **Philippines**, in a study by Schmidt et al (n.d.) the average value of a hectare of rice after applying a mix of organic and mineral fertilizer was estimated at US$4427 per hectare and US$4172 per hectare for synthetic fertilizer alone.

There was no primary research data on GHG emissions. Modelled results showed a strong increase in emission from US$563 to US$2980 when organic fertilizer in the form of rice straw was added.

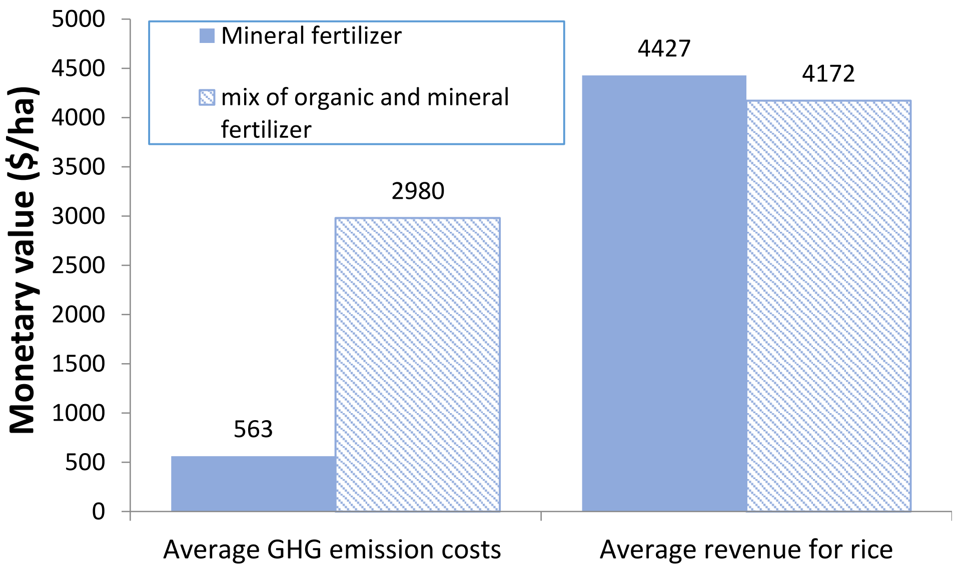

**Figure 3.5.7.** Valuation results comparing a reduction in fertilizer use in terms of GHG emission costs and average revenue for rice per hectare in irrigated lowland systems. Monetary valuation for GHG emissions is based on modeled data.

In another study in the Philippines by Haefele et al (2011) the average value of a hectare of rice after applying a mix of organic and mineral fertilizer was estimated at US$2021 per hectare and US$2102 per hectare for synthetic fertilizer alone.

There was no primary research data on GHG emissions. Modeled results showed a strong increase in emissions from US$213 to US$740 when organic fertilizer in the form of rice straw was added.

In another study in the Philippines by Sander et al (2014) the average value of a hectare of rice after applying a mix of organic and mineral fertilizer was estimated at US$2877 per hectare and US$3047 per hectare for synthetic fertilizer alone.

Primary research data on GHG emission was available for this study. GHG emissions were US$751 without and US$1496 with added rice straw.

In another study in the Philippines by Wassmann et al (2000) the average value of a hectare of rice after applying a mix of organic and mineral fertilizer was estimated at US$1824 per hectare and US$2043 per hectare for synthetic fertilizer alone.

Primary research data on GHG emission was available for this study. Average GHG emissions were US$67 without and US$1168 with added rice straw or green manure. Rice straw produced significantly higher emission costs (US$2069) than green manure (US$266).

In another study in the Philippines by Bronson et al (1997) the average value of a hectare of rice after applying a mix of organic and mineral fertilizer was estimated at US$1914 per hectare and US$2254 per hectare for synthetic fertilizer alone.

Primary research data on GHG emission was available for this study. GHG emissions were US$58 without and US$651 with added rice straw or green manure. Rice straw produced significantly higher emissions costs (US$760) than green manure (US$433).

When averaging the results from all studies in the Philippines, the average value of a hectare of rice after applying a mix of organic and mineral fertilizer was estimated at US$2410 per hectare and US$2589 per hectare for synthetic fertilizer alone.

When averaging the results for GHG emissions (primary research data only), GHG emission costs were US$676 per hectare without and US$801 per hectare with added rice straw or green manure.

The results can be explained by the fact that incorporation of organic fertilizer such as rice straw or husk under wet conditions such as in irrigated lowland systems leads to temporary immobilization of N and a considerable increase in $CH_4$ emissions (e.g. Dobermann & Fairhurst, 2002).

If all rice farmers of IL systems in the Philippines were to use a mix of organic and mineral fertilizer instead of mineral fertilizer only, they would initially lose US$ 551,320,000 worth of yields. However, these are short term effects. On the long term, a straw addition actually increases soil fertility, especially in N, P, K and Si (Dobermann & Fairhurst, 2002) and hence also yields. GHG emission costs would increase by US$ 385,000,000.

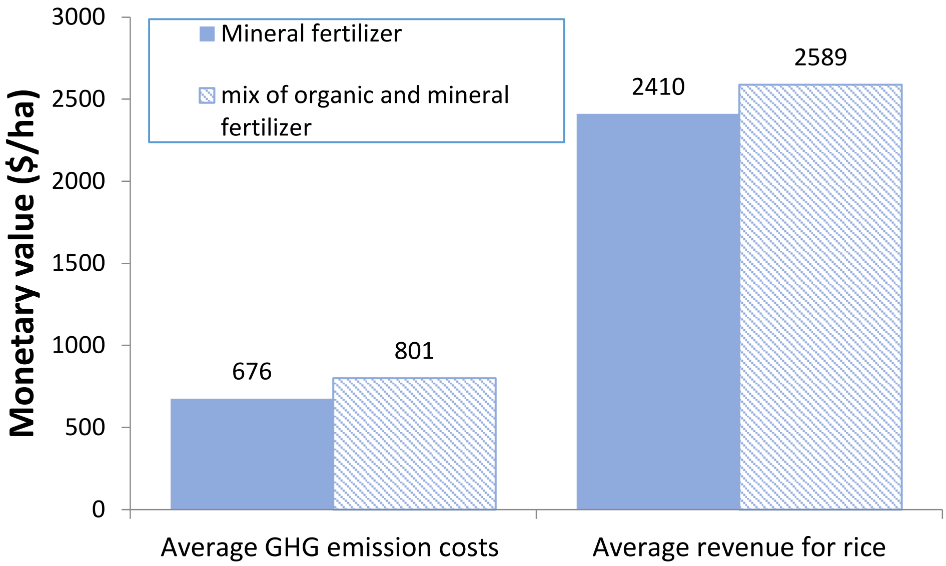


**Figure 3.5.8.** Valuation results comparing mineral fertilizer use with the combined use of mineral and organic fertilizer in terms of GHG emission costs and revenues in irrigated lowland systems in the Philippines. Monetary valuation for GHG emissions is based on primary research data.

In **Costa Rica (IL)**, in a study by Quiros (2006) the average value of a hectare of rice after applying a mix of organic and mineral fertilizer was estimated at US$5075 per hectare and US$4466 per hectare for synthetic fertilizer alone. Contrary to the other studies there was an increase in yield when organic fertilizers were added. Most probably because the fertilizer was not rice straw or husk, but the Velvet bean, *Mucuna pruriens*, instead.

There was no primary research data on GHG emissions. Modelled results can only give an order of magnitude as the GHG emission methodology does not account for emissions from *Mucuna pruriens*. Emission costs were US$193 per hectare.

### f. Comparing local to new rice varieties (IL)

In **irrigated lowland systems (IL)** in the **Philippines**, research by Wassmann et al (2000) compared cumulative $CH_4$ emissions from rice fields of improved varieties to the local IR 72 variety. Four different new varieties were tested over two seasons. As 3.5.10 shows the local variety has a higher cumulative $CH_4$ emission flux in both dry and wet seasons. The variety yielded higher compared to both varieties in the dry season. However, no significant effect has been found. For the wet season the local variety yielded significantly higher compared to IR65597 and significant lower compared to the Magat variety. There is a clear trade-off between the local variety with higher yields (although not statistically significant) and improved varieties with lower emissions.

The average value of one hectare of rice using the local variety resulted in US$1975, and US$1732 when averaging the values for all new varieties. There was primary research data on $CH_4$ emissions only. The local variety led to GHG emission costs of US$27 per hectare and the new varieties led to an average cost of US$19 per hectare. Figure 3.5.10 below shows the underlying biophysical data.

If farmers would plant these new varieties instead of the local variety all over IL in the entire country, farmers would reduce US$ 24,640,000 of GHG emission costs However, they would lose US$ 748,440,000 worth of yields. It is questionable however that the "local" variety would show the same results all throughout the country, as "local" variety usually refers to a plant that is adapted to local - not national - conditions.

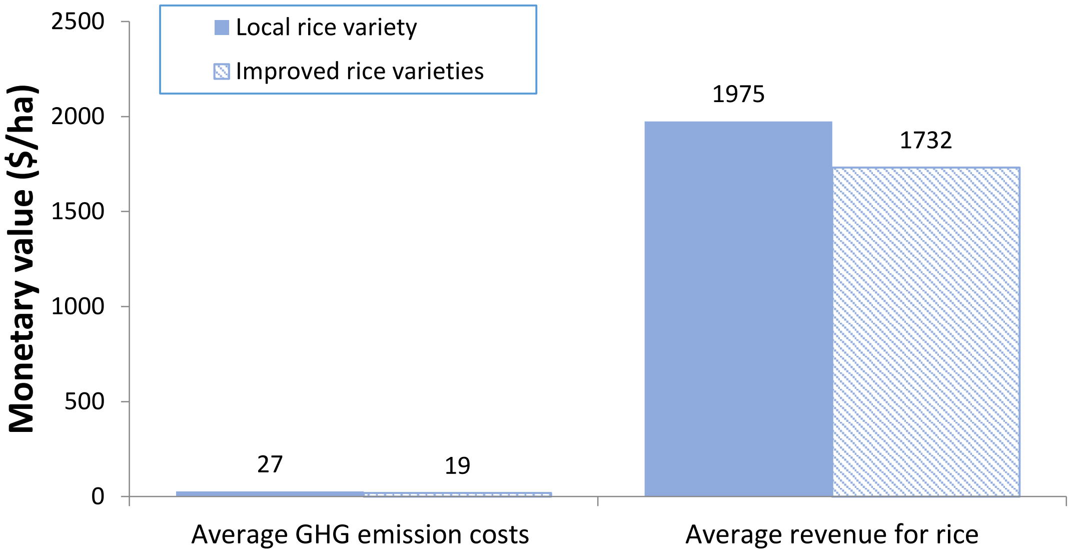


**Figure 3.5.10.** Valuation results comparing a reduction in fertilizer use between a local and new varieties in terms of GHG emission costs and average revenue for rice in US$ per hectare in irrigated lowland systems in the Philippines. Monetary valuation for GHG emissions is based on primary research data.

## 3.6 Increase in rice yield versus habitat provisioning

The practice of submerging soil for rice production has existed for many hundreds of years providing habitat for a wide range of organisms such as aquatic plants, fish and water fowl. These occur naturally or as a result of cultivating aquatic organisms within the available water (Halwart & Gupta, 2004). Rice-fish production can be done concurrently or as rotational crops and can involve many species beyond fish including crabs, prawns, turtles, and mollusks. The breadth of biodiversity in rice fields however, extends far beyond what is intentionally cultivated. A study in 1979 recorded 589 total species of organisms in a rice field in Thailand, of which 18 were species of fish and 10 were species of reptiles and amphibians (Halwart & Gupta, 2004). Thus, the value of aquatic biodiversity in flooded-rice production extends far beyond the traditional interpretation of rice cultivation.

Through the provision of habitat to a suite of different plant or animal species, rice agro-ecosystems also improve basic human nutrition and dietary diversity. Nutrients such as Vitamin A, B, calcium, iron or zinc, or different amino acids which are often lacking in the diet of rural people could be supplied by rice agro-ecosystems and their aquatic biodiversity (Burlingame et al 2006). Many rural households depend on monotonous diets that are too high in carbohydrates and too low in animal source foods and micronutrient-rich fruits, fish and vegetables. Access to a diversified diet is often constrained by lack of purchasing power, limited expertise and limited availability. Experience has shown that more diversified farming systems that contain horticultural or aqua cultural components are one way to improve households' availability and access to such animal source foods, fruits and vegetables (see section 3.9 a for more information).

While within the five case study countries there was only one reported case in the peer reviewed literature which documented and quantified the importance of habitat provisioning of rice systems for overwintering winter fowl (see narrative review, California) with its concomitant benefits for recreational activities such as hunting (i.e. cultural ecosystem services), there are plenty of studies around the globe that report on rice as an important wetland habitat.

### a. Pesticide free rice production[2]

There was no peer reviewed study in the case five study countries that quantified the differences between those rice systems that use agro-chemicals and those that do not in terms of habitat provisioning. However, grey literature sources extensively document rice-fish farming as a crucial livelihood activity for Asian rice farmers.

A recent literature review (Griffith, 2015, unpublished) on ecosystem services provided by aquatic organisms in global rice production systems lists more than 30 papers that have assigned a value to the provision of food by aquatic organisms in rice fields.

For example, in Cambodia, de Silva et al (2013) point out that fishing and foraging are a crucial source for food and seasonal income for parts of the year. In addition to vegetables, rice fields provide 50 to 250 kg of fish and other aquatic animals per family and year, with a value of about 100 to 150 USD per hectare (Hortle et al 2008). Often these are the primary sources of protein for the rural rice farming communities, and therefore of immense nutritional value - not just for the rice farmers alone, but also for landless members of the community.

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[2] As there were no peer reviewed studies in the five case study countries, we included data from other countries to show the importance of this ecosystem service in relation to rice yields alone.

Beyond the provision of food and income, aquatic species also provide important (biological) pest control services. Naturally occurring frogs or toads, or carnivorous fish keep rice pests at a low level. A study done in China reported 68% less expenses for pesticides and 24% less chemical fertilizer when rice-fish culture was practiced as compared to monocultures (Xie et al 2011).

A study done in Laos examined the use of aquatic organisms in rice fields and identified their roles in household economy (Yamada et al, 2004). The average amount of biological resources sold was the highest in the mountain villages, USD 85/year/household, followed by the hillside villages at USD 41/year/household, and lowland villages at USD 23 /year/household. These amounts represent 53 percent, 27 percent and 18 percent of total household cash income, respectively.

This and other studies have shown that the income from aquatic organisms can significantly complement, if not double the revenue from rice farming. A study by Muthmainnah (2015) showed that the income from aquatic organisms in rice paddies was indeed more than that of rice farming only (1100 USD/local community as compared to 800USD/local community).

Beyond the direct benefit of increasing the farmers' income through the cultivation of aquatic species, there are also seems to be an indirect benefit through the increase of rice yields. Halwart and Gupta (2004) analyzed data from five different countries from Asia, including the Philippines, and found that in 80% of the cases, the introduction of fish led to higher rice yields (by at least 2.5 percent) than without fish. The authors explain this increase by a decreased likelihood of weeds and stembores which inevitable leads to healthier rice plants.

### b. Winter flooding (IL)

Studies from California documented the different effect of winter flooding versus no flooding. For statistically significant data, there was an increase in yields when fields were flooded in 83% of all cases and no difference in 17% of the cases. The average value of rice production was estimated to be US$3,610 when fields were winter flooded, and US$3,442 when they were not.

For effect size data, habitat increased in two thirds of all cases when fields were winter flooded. In the other third of the cases there was no difference. There was no statistically significant data. Habitat provisioning was not valued in monetary terms.

Figure 3.6.1 gives biophysical values instead, based on the study of Day and Colwell (1998). On average, habitat provisioning increased with winter flooding from 7.91 water birds per hectare to 13.29 winter bird per hectare. Yields increased from 7.66 tons per hectare to 8.9 tons per hectare.

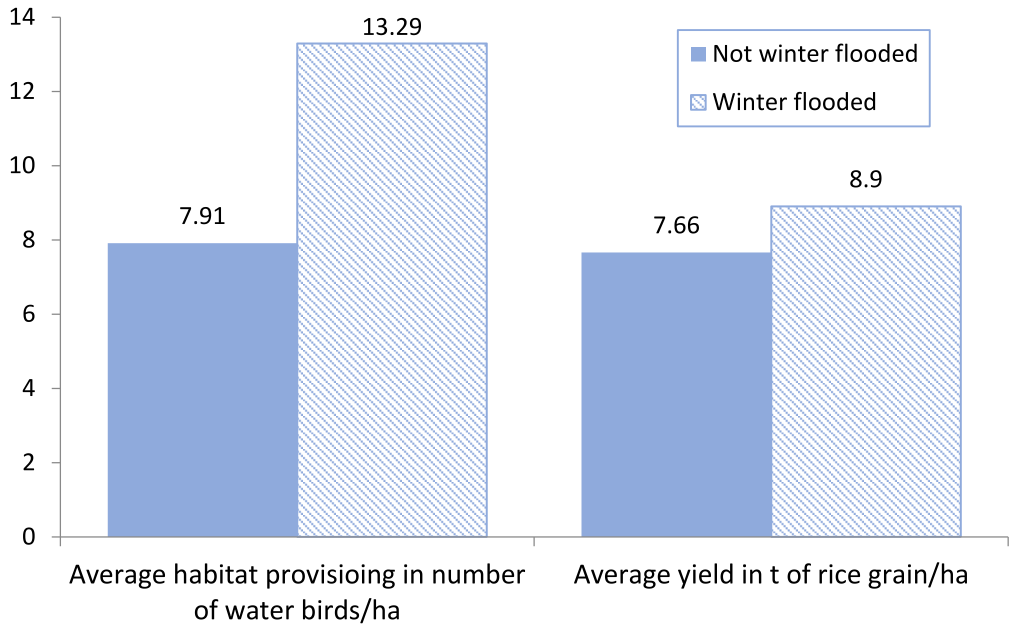


**Figure 3.6.1.** Habitat provisioning in number of water birds/ha and yields in t of rice grain/ha when winter flooded or not winter flooded in California.

## 3.7 Increase in rice yield versus improving nutrient cycling and soil fertility

Nutrient cycling and soil fertility, a regulating ecosystem service, underpin many other ecosystem services, and are hence one of the essential "inputs" for sustaining rice production as such. It is therefore logical that an increase in rice yield is closely linked to the well-functioning regulation of nutrient cycling and soil fertility, which depend on a suite of biological, chemical and physical processes over space and time. Studies on these essential ecosystem services are scare however. The few studies that exist compare the effect of different fertilizer regimes on crop yields and to a lesser degree on soil fertility. Research revolves around the question whether mineral fertilizers can be partially replaced by organic ones without compromising soil fertility and yields. Or to which degree certain organic amendments such as rice straw actually increase soil fertility. While the effect of rice straw incorporation does have a positive effect on soil fertility and yields on the long run as discussed above, the sample size of the vote counting analysis was too small to draw any clear conclusions from the case studies. The effect of other organic fertilizers such as green and animal manure (e.g the water fern azolla, or duck droppings), intercropping or crop rotations is hardly documented and quantified in any of the five case study countries. Data was available for irrigated lowland systems (IL), rainfed lowland systems (RL) and rainfed upland systems (RU).

Rice straw and rice husks are the only organic materials which are available to all rice farmers, if not burned or used for other purposes. Straw contains about 40% of the N, 30 to 35 % of the P, 80 to 85% of the K and 40 to 50% of the sulfur (S) that a rice plant takes up (Dobermann & Fairhust, 2002). However, the mere existence of these nutrients in the straw does not make them readily available for the soil[3]. To what extent nutrient cycling and soil fertility benefits from rice straw can be used to full capacity strongly depends on the state of the soil and the residue management practice. Straw can be burned and the ashes spread on the field, it can be removed or incorporated in the soil. The same applies for rice husks as the following examples clearly show.

### a. Using mineral and organic fertilizer together instead of mineral fertilizer only (IL, RL and RU)

For **irrigated lowland systems (IL),** the vote-counting analysis compared several experiments from Senegal and the Philippines with statistically significant results showing soil fertility impacts. In more than 80% of the cases there was no difference in soil fertility parameters between using mineral and organic fertilizer together or of mineral fertilizer only. It is difficult to draw any conclusions from these few results, however. Not only is the sample size very small, but also the duration of the experiments seems to be too short to see any changes take place after the organic amendment has been added to the soil. Yields declined in 20% of the cases, while in the other cases 80% there was no difference when fertilizers were mixed. Since a monetary valuation of soil fertility was not possible, we present the original biophysical quantification data for both yields and soil fertility.

In the **Philippines**, an experiment by Haefele et al (2011) compared different residue management practices. The researchers contrasted the application of mineral fertilizer with a mix out of mineral and organic fertilizer, i.e. carbonized or untreated rice husks. Figure 3.7.1 below shows the effect on yield and on different soil fertility parameters. The majority of soil fertility parameters increased with the addition of organic material: Total N content, soil available P, soil exchangeable K, and total organic carbon content increased and bulk density decreased when organic fertilizer

[3] In section 3.3. we used modelled data to determine the nutrient value of rice. This is a theoretical exercise as it can only determine the nutrient in the straw, but it cannot tell whether these nutrients are actually absorbed by the soils and the "value" materializes. Using primary research data from field experiments is therefore more appropriate in this regard.

was added on top of mineral fertilizer over a period of four years. Yields were slightly lower than when fertilized with mineral fertilizer only, which was due to the high C/N ratio of the carbonized rice husk presumably.

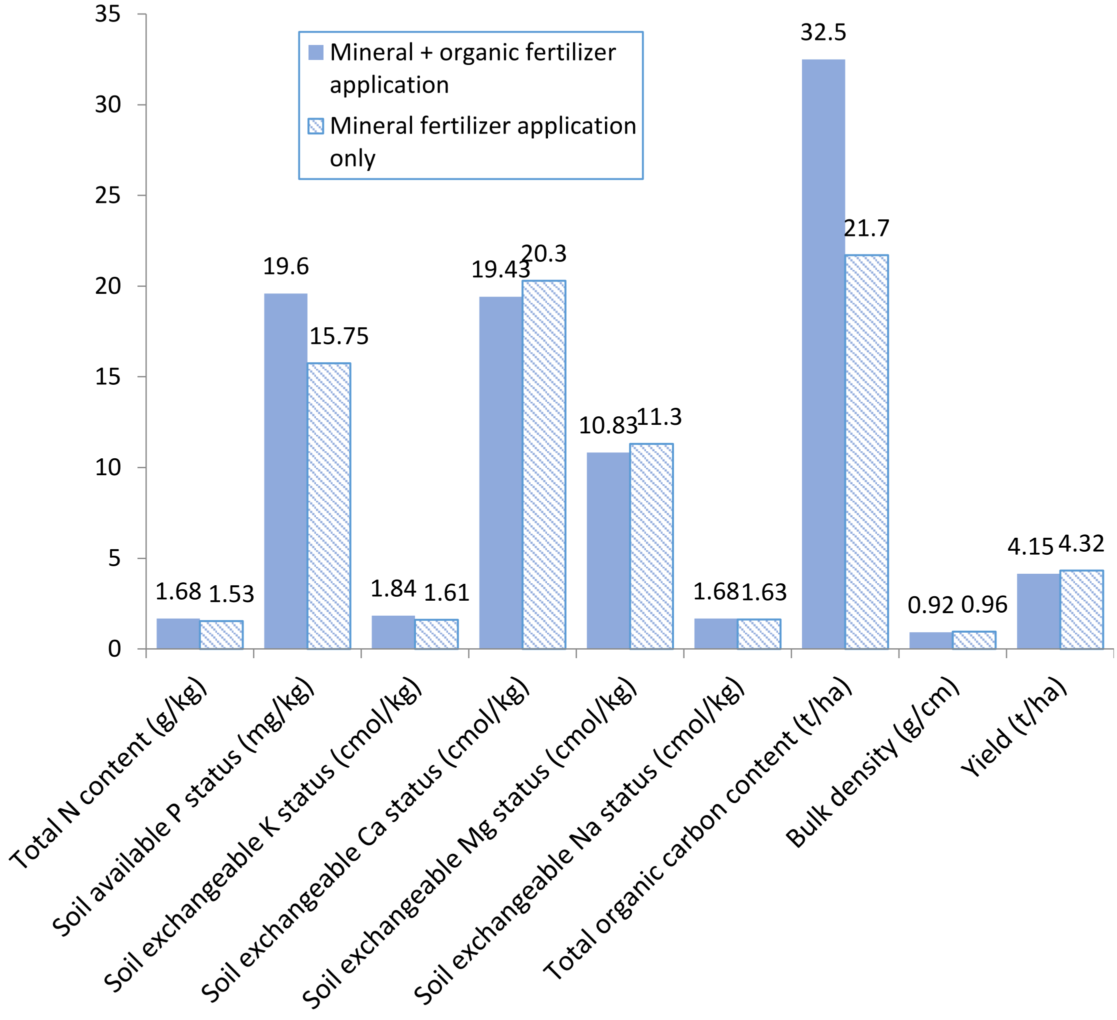


**Figure 3.7.1.** Treatment comparison between mineral fertilizer application only versus mineral and organic fertilizer, i.e. carbonized or untreated rice husks. The effect on yield and several soil fertility parameters were measured. Please note the metric of the Y axis relates to different measurement units. The Y axis is formatted in logarithmic scale.

**In Senegal**, an experiment done by Krupnik et al (2012a) compared the use of mineral fertilizer to the joint application of mineral fertilizer and rice straw over a period of three years, in total five growing seasons. Partial nutrient balances were calculated by subtracting output of nitrogen, phosphorus and potassium from the input. Nutrient inputs were measured from precipitation, irrigation, straw input and fertilizer. Output of nutrients were measured from straw export, grain export and drainage water. The joint application of mineral and organic fertilizer resulted in a positive nutrient balance, indicating a build-up of nutrients in the soil. Application of mineral fertilizer only shows a negative nutrient balance resulting in a depletion of soil nutrients.

N recovery efficiency indicates the portion of available nitrogen recovered in the biomass of the rice plant. The data shows higher N recovery efficiency for the combination of mineral and organic fertilizer application compared to mineral fertilizer application only. This means that the treatment under mineral and organic fertilizer regimes is better able to uptake and use the available N.

Yields increased slightly with the addition of straw, on average from 6.33 t $ha^{-1}$ to 7.06 t $ha^{-1}$. The differences in yield are minimal, as the experiment could only capture short term effects. Long term effects of straw incorporation are usually significant however. Studies have shown that where straw is incorporated and mineral fertilizer is used, reserves of N, P, K and Silicon (Si)are maintained or increased (Dobermann & Fairhust, 2002).

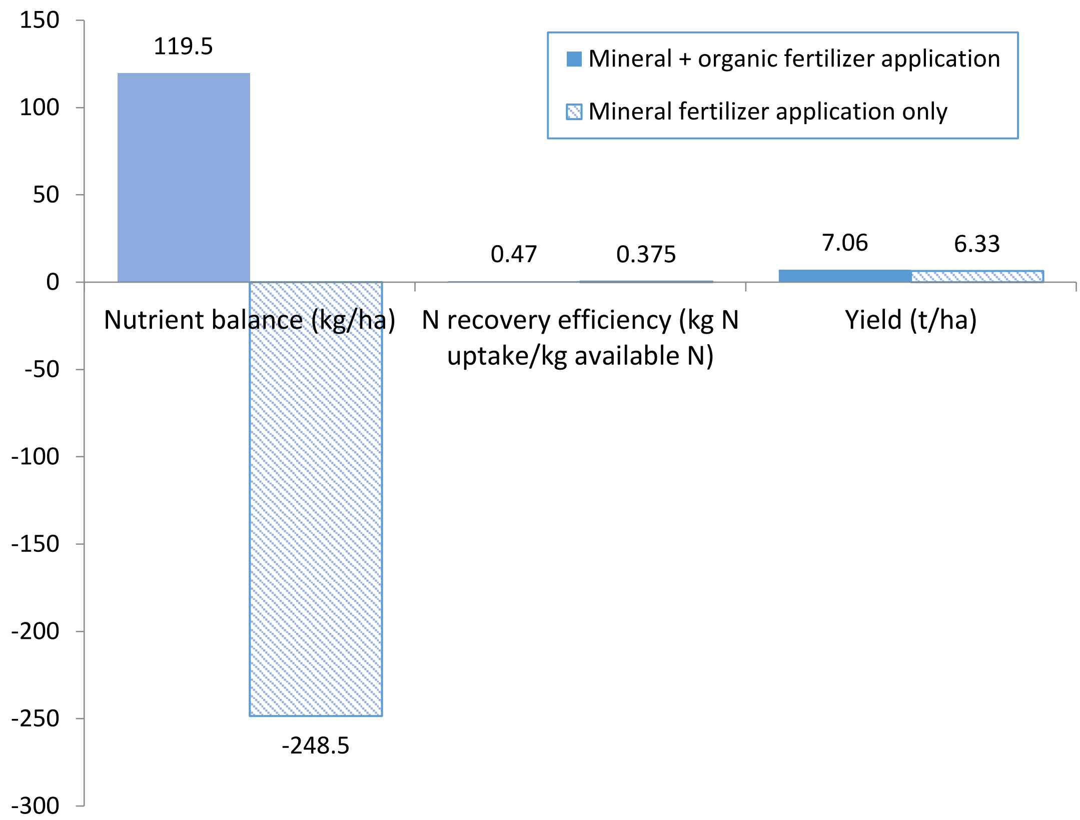


**Figure 3.7.2.** Biophysical results for yields and soil fertility parameters comparing two different fertilizer regimes – mineral and organic compared to mineral fertilizer only. The graph shows average numbers over five growing seasons.

**In rainfed lowland systems (RL),** for soil fertility, in 2 out of 4 cases there was an increase when fertilizers were mixed (50%) and no difference between the two treatments for the other 2 (50%) (effect size data). Soil fertility was not valued in monetary terms. Figure 3.7.3 below shows the biophysical values measured for both soil fertility and yields. All data is from **Cambodia**.

Average rice yields in a study by Vang et al (2010) were 10t/ha when mineral fertilizers were used, and 11.7 t/ha when both mineral and organic fertilizers (straw, cow manure and green manure) were applied. Soil fertility increased for all measure soil fertility parameters when fertilizers were mixed.

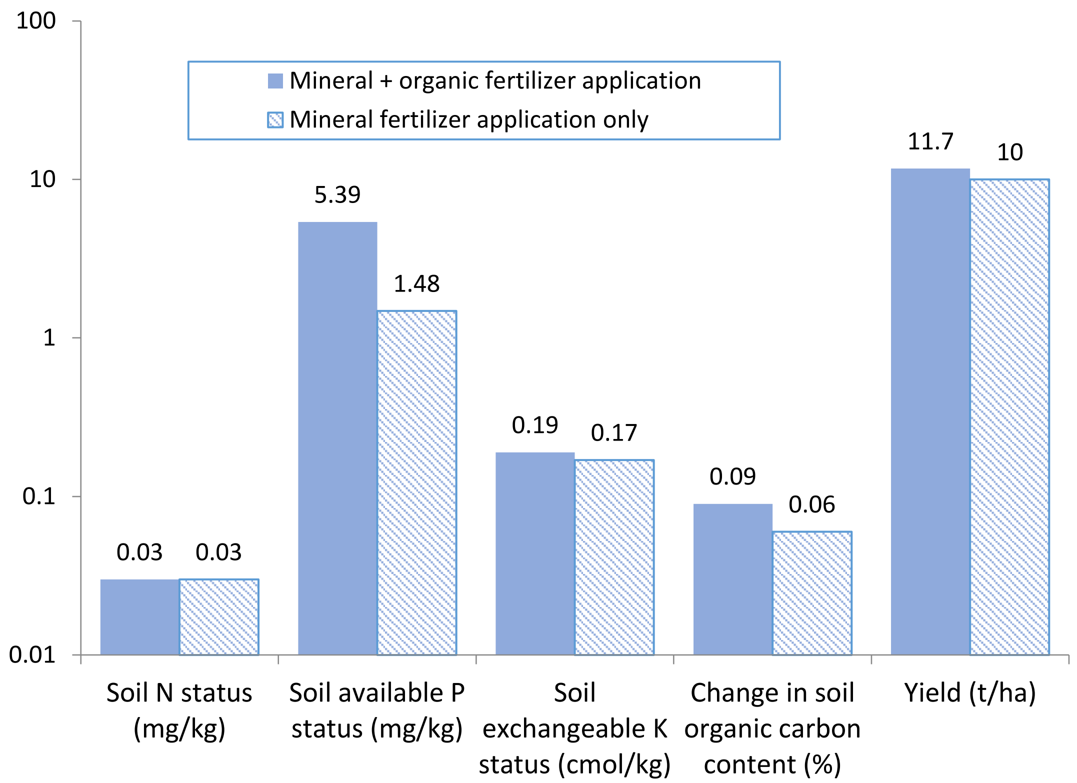


**Figure 3.7.3.** Biophysical results for yields and soil fertility parameters comparing two different fertilizer regimes – mineral and organic compared to mineral fertilizer only. The Y axis is formatted in logarithmic scale.

An experiment by Haefele et al (2011) compared different residue management practices **in rainfed upland systems (RU)** in the **Philippines.** The researchers contrasted the application of mineral fertilizer with a mix out of mineral and organic fertilizer, i.e. carbonized or untreated rice husks. There was an increase in soil fertility in 36% of all cases when fertilizers were mixed. There was no difference in 60% and a decrease in 4% of the cases when organic and mineral fertilizer were used together (statistically significant data). Yields declined in 15% of the cases, while in 85% of the cases there was no statistically significant difference between the two treatments.

As soil fertility could not value in monetary terms, we show the biophysical values for averaged soil fertility parameters and yield. Figure 3.7.4 below shows that the majority of all soil fertility parameters improved with the addition of organic material: Total N content, soil available P, soil exchangeable K, soil exchangeable Mg, soil exchangeable Na and total organic carbon content increased and bulk density decreased when organic fertilizer was added on top of mineral fertilizer over a period of four years. Yields improved slightly.

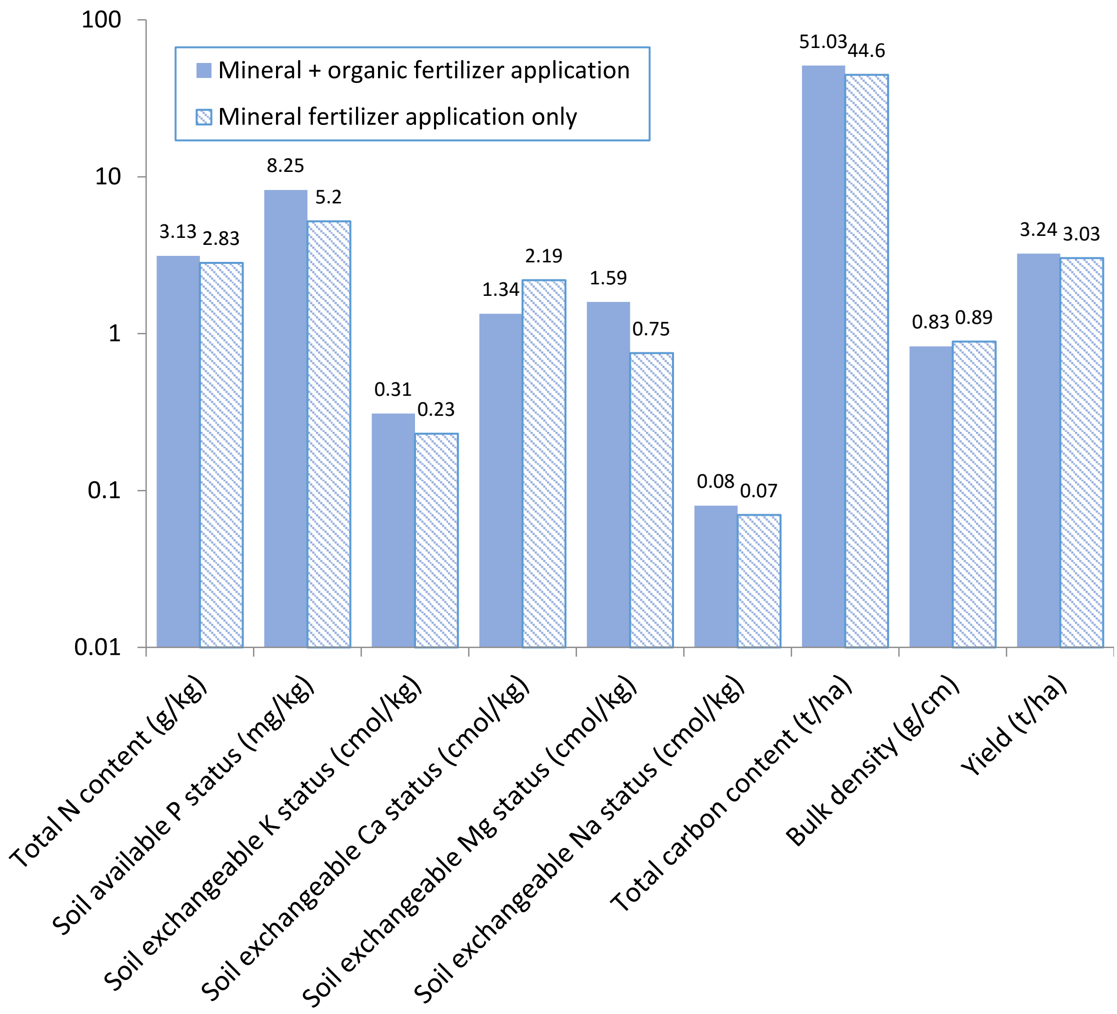


**Figure 3.7.4.** Treatment comparison between mineral fertilizer application only versus mineral and organic fertilizer together, i.e. carbonized or untreated rice husks, for rainfed upland systems in the Philippines. The effect on yield and several soil fertility parameters were measured. Please note the metric of the Y axis relates to different measurement units. The Y axis is formatted in logarithmic scale.

### b. Using organic fertilizer instead of no addition (IL, RL and RU)

Two studies compared the use of organic fertilizer to no addition in **irrigated lowland systems (IL).** In terms of nutrient cycling and soil fertility, there was a statistically significant increase in nutrient cycling and soil fertility in all cases (100%). As for yield, in 36 % of all cases there was an increase in yield while in the remaining 64% there was no difference. Soil fertility was not valued in monetary terms.

Figure 3.7.5 below shows the results of an experiment by Haefele et al (2011) in the **Philippines in irrigated lowland systems (IL).** The researchers contrasted the application of organic fertilizer, i.e. carbonized or untreated rice husks, with no addition. Unsurprisingly, the majority of all soil fertility parameters improved with the addition of organic material: Total N content, soil available P, soil exchangeable K, soil exchangeable Na and total organic carbon content increased and bulk density decreased when organic fertilizer was added to the soil over a period of four years. On average, yields decreased slightly, which was due to the high C/N ratio of the carbonized rice husk presumably.

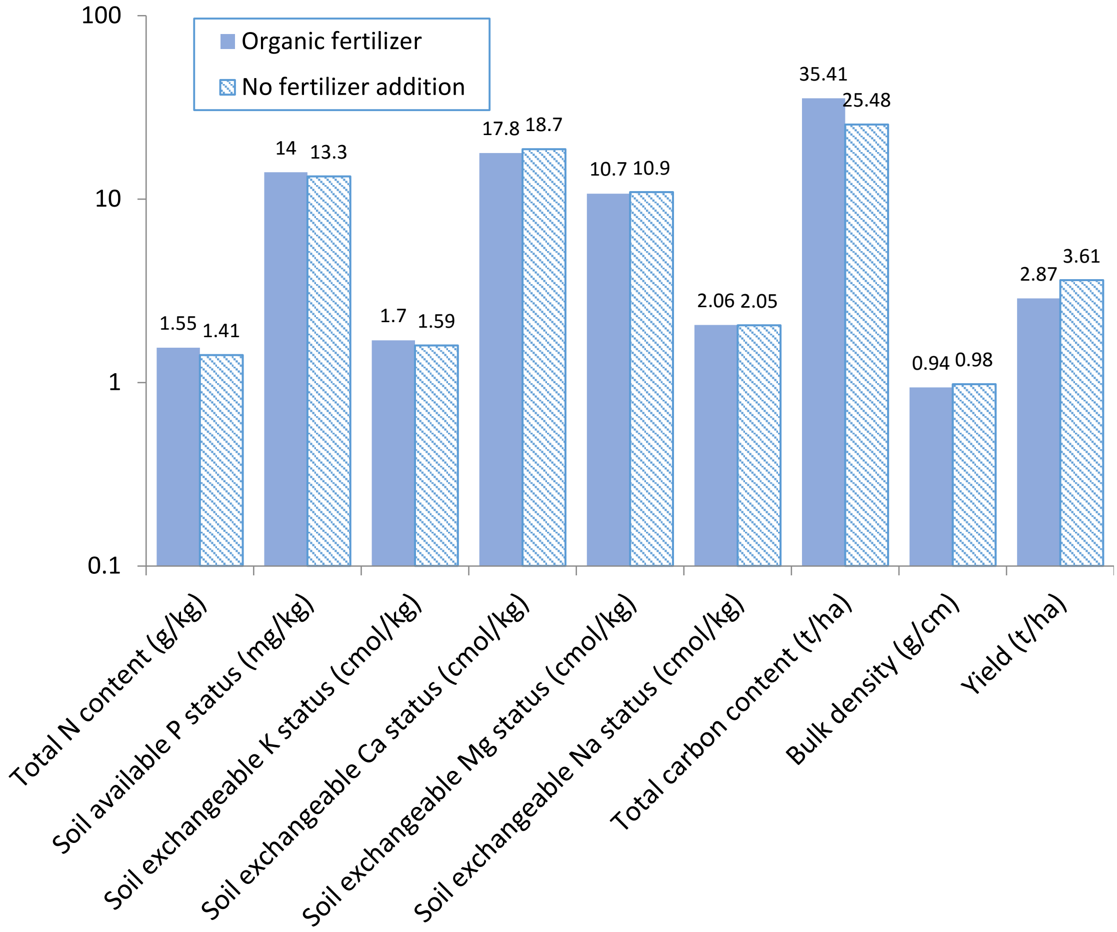


**Figure 3.7.5.** Treatment comparison between organic fertilizer application, i.e. carbonized or untreated rice husks versus no addition for irrigated lowland systems in the Philippines. The effect on yield and several soil fertility parameters were measured. Please note the metric of the Y axis relates to different measurement units. The Y axis is formatted in logarithmic scale.

Figure 3.7.6 shows the results of a study from Krupnik et al (2012a) in Senegal comparing straw input to no fertilizer application. Partial nutrient balances are calculated by subtracting output of nitrogen, phosphorus and potassium from the input. Nutrient inputs were measured from precipitation, irrigation and straw input. Output of nutrients were measured from straw export, grain export and drainage water. The nutrient balance of the treatment with straw incorporation is slightly positive, while the nutrient balance of no fertilizer application is about 400 kg/ha negative, resulting in the depletion of soil nutrients. Yield increased on average from 3.6 t $ha^{-1}$ to 4.17 t $ha^{-1}$ when straw was added to the soil.

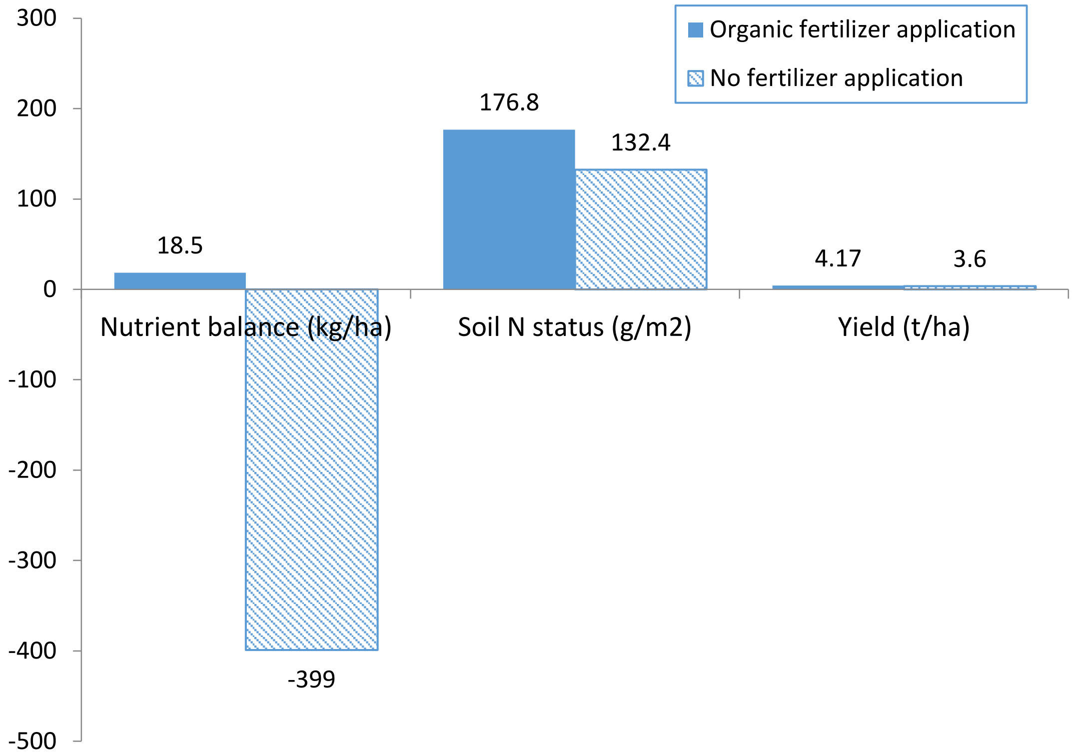


**Figure 3.7.6.** Biophysical results for yields and soil fertility parameters comparing organic fertilizer input (rice straw) to no fertilizer input. The graph shows average numbers over five growing seasons.

In **rainfed upland systems (RU),** one study in the **Philippines** (Haefele et al 2011) compared the use of organic fertilizer, i.e. carbonized or untreated rice husks, to no addition. In terms of nutrient cycling and soil fertility, they found that in for statistical significant data, in 31 out of 43 cases (72%) there was no difference when organic fertilizer was used. There was an increase in nutrient cycling and soil fertility in 12 out of 43 cases (28%). Soil fertility was not valued in monetary terms. As for yield, for statistical significant data, there was no difference in 19 out of 20 cases (95%), and an increase in 1 case (5%).

Figure 3.7.7 below compares the application of organic fertilizer with a treatment where no fertilizer was added. Unsurprisingly, the majority of all soil fertility parameters improved with the addition of organic material: Total N content, soil available P, soil exchangeable Ca and total organic carbon content increased when organic fertilizer was added to the soil over a period of four years. Yields increased slightly.

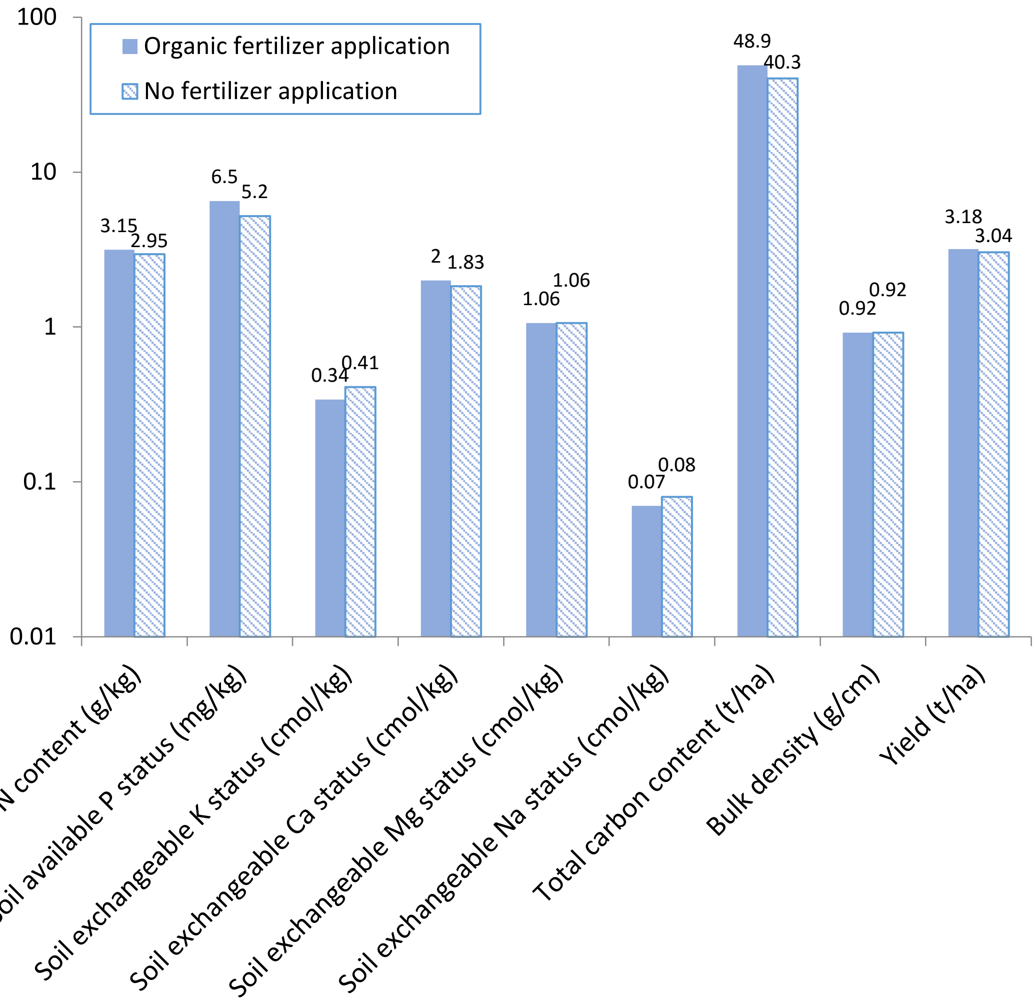


**Figure 3.7.7.** Treatment comparison between organic fertilizer application, i.e. carbonized or untreated rice husks versus no addition for rainfed upland systems in the Philippines. The effect on yield and several soil fertility parameters were measured. Please note the metric of the Y axis relates to different measurement units. The Y axis is formatted in logarithmic scale.

### c. Organic fertilizer with winter flooding and without winter flooding

In an experiment in **California** by Eagle et al (2001) in **irrigated lowland systems,** the researchers compared the N recovery efficiency (measured in kg/ha of total N in the plant) after rice straw incorporation when paddies were winter flooded or not.

The results showed that the N recovery efficiency increased from 150.7 kg/ha when fields were not flooded, to 159.7kg/ha when fields were flooded. At the same time, yields increased from 5.7 t/ha to 6.4 t/ha.

## 3.8 Increase in yield versus preventing pest and disease outbreaks

The occurrence of pest species as such is not a problem, it becomes problematic however when the natural ecological balance is disrupted leading to serious outbreaks due to exponential multiplication of the pests. Pest outbreaks can be prevented by strengthening the natural community of arthropod species of an agro-ecosystem in the first place, or by applying genetic, cultural, biological, mechanical or carefully measured chemical control mechanisms that suppress pest and disease populations before they become too numerous.

There were no peer reviewed studies in the five case study countries as documented in other places showing how a healthy ecosystem can increase the ecological resilience of rice agro-ecosystem to prevent pest and disease outbreaks. For example, the greatest single cause of Brown Planthopper outbreaks is pesticide use. Ample evidence shows that these planthoppers are an insecticide-induced resurgent pest, which has adapted itself even to rice varieties which were developed to be resistant against this rice pest (e.g. Settle et al. 1996, Heong et al, 2015). In a healthy rice system, the number of invading and reproducing plant hoppers is controlled by natural enemies (e.g. spiders, insectivorous bats, parasitic wasps), yet when such predators and parasitoids are reduced or absent through early pesticide spraying, invading pest populations grow exponentially, which results in pest outbreaks and consequent crop damage.

Work by Settle (1995) describes how the building of organic matter enhances the habitat of natural enemy communities, thereby supporting high levels of natural biological control. If organic matter is increased early in the growing season, abundant populations of detritus- feeding and plankton-feeding insects will be fostered, usually peaking and declining in the first third of the season. These insects have no impact on rice yields, either positive or negative- but their populations provide natural enemies of rice pests a "head start" – to build up their populations early in the season so as to be able to strongly suppress the pest populations which enter the paddy in mid-season. Pesticides early in the season will prevent the strong build-up of natural enemies, killing both them and their early-season food source.

Although this is an integral part of Integrated Pest Management, the little that has been published on the subject area was not published in any of the five case study countries.

The same holds true for the deliberate introduction of biological control agents. While this seems to be a promising way to avoid chemical plant protection measures, there has been no peer reviewed study in any of the five countries.

To some extent, literature from the five countries showed alternative management practices to chemical pest control such as the cultural mechanisms or mechanical practices. These – while not actively promoting a natural enemy community – passively provide a favorable environment for the development of an ecological infrastructure that can host natural enemy species. Other advantages are the usually low expenses, the low(er) environmental impacts and the fact that pests are not likely to develop resistances as compared to chemical interventions. On the negative side, some of the chemical free practices decrease yields. Also, many of these approaches require the adoption by the entire community in order to make them effective.

### a. Cultural control

Cultural control aims to modify production practices in a way that allow for better control of pests, weeds and diseases. At the same time, many of these control mechanisms also strengthen natural enemy communities over time, or make rice plants more resilience against pest attacks. However, a change of production practices may affect yield. Each type of intervention therefore needs to be weighted carefully.

**Rice straw burning** is a common measure control pests, weeds and diseases. However, as described in an earlier section, the air pollution linked to burning has led policy makers to forbid rice straw burning in many parts of the world. The alternative measures eliminate the incidence of air pollution, yet pest, weed and disease control can be a problem in these cases. In a study on stem rot disease *Sclerotium orzae* (Cintas & Webster 2001) where straw burning was replaced by either straw crushing and rolling into the soil, soil incorporation or by straw baling and removal, pest control decreased drastically in all cases (100%), both for effect size and statistically significant data. However, burning is also destroying the buildup of natural enemy populations, and is therefore not a recommended management practice to control pests sustainably. Pest control could not be valued in monetary terms, we therefore show the biophysical values (Figure 3.8.1).

As for yields, for statistically significant data, incorporation showed no difference in 64%. Yields were reduced when straw was incorporated as compared to burning it in 29% of all cases. The average value of rice production was estimated to be US$4748 when rice was burnt and US$4491 when rice was incorporated. The number of *Sclerotium* per gram of soil increased from 0.62 to 1.3, hence disease control more than doubled with burning.

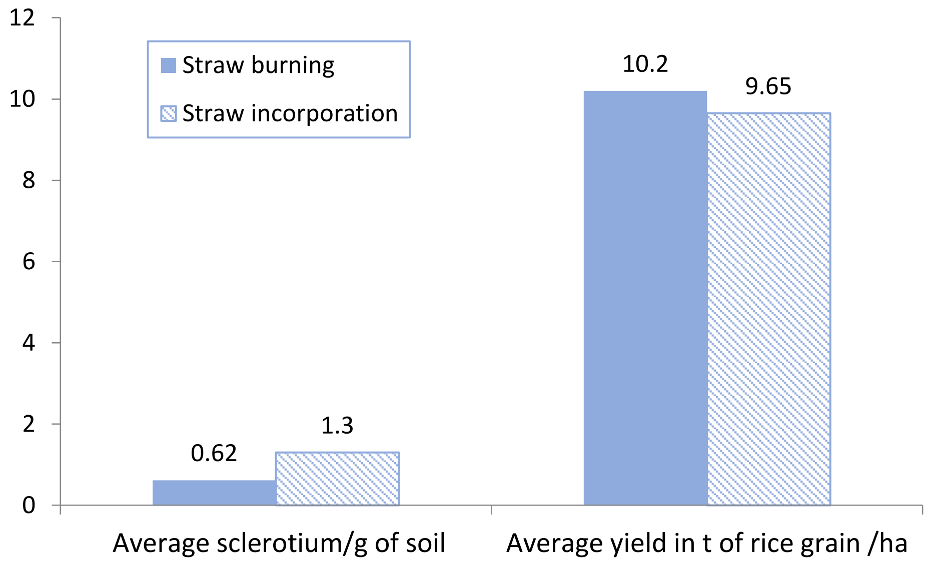


**Figure 3.8.1.** Comparison of yield in ton per hectare and number of *Sclerotium* per gram of soil when straw is burnt or when straw is incorporated.

Rolling showed no difference in 86% of all cases, a decrease in 12% and an increase in 12%. The average value of rice production was estimated to be US$4748 when rice was burnt, and US$4555 when rice straw was rolled into the soil. The number of *Sclerotium* per gram of soil increased from 0.62 to 2.03, hence disease control more than trippled with burning.

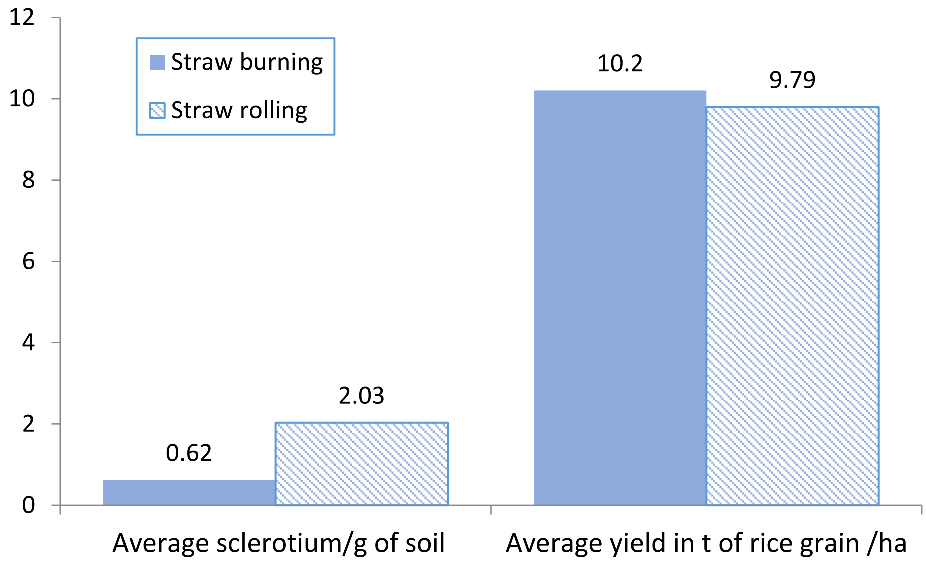


**Figure 3.8.2.** Comparison of yield in ton per hectare and number of *Sclerotium* per gram of soil when straw is burnt or when straw is rolled into the soil.

Baling and removal showed no difference in 86% of all cases, and a decrease in 14% of all cases. The average value of rice production was estimated to be US$4748 when rice was burnt, and US$4559 when rice straw was removed. The number of *Sclerotium* per gram of soil increased from 0.62 to 1.27, hence disease control doubled with burning as compared to straw removal.

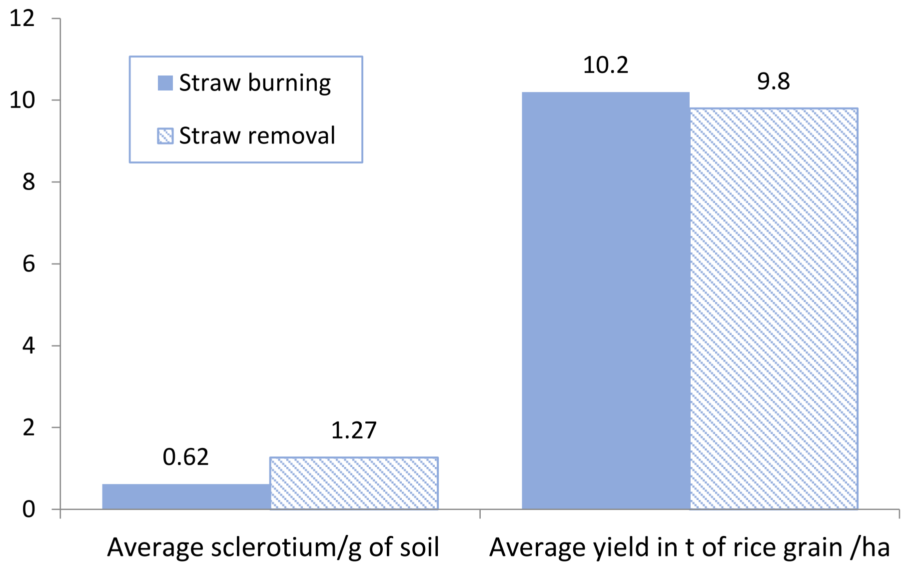


**Figure 3.8.3.** Comparison of yield in ton per hectare and number of *Sclerotium* per gram of soil when straw is burnt or when straw is removed.

The study by Cintas & Webster (2001) also found that **winter flooding** was the best alternative to straw burning in order to control the disease *Sclerotium orzae.* Statistically significant data showed that the disease incidence decreased in 83% of all cases.

As to be expected, this had a positive effect on yields as the disease was controlled better. For statistically significant data, yields were higher in 83% of all cases when fields were winter flooded.

The average value of rice production was estimated to be US$4698 when fields were winter flooded, and US$4459 when fields were not. The decrease in pest control was not valued in monetary terms.

We therefore displayed the data from the Cintas & Webster (2001) study in biophysical terms. The number of *Sclerotium* per gram of soil increased from an average of 0.89 to 1.76, hence disease control almost doubled with winter flooding as compared to no flooding.

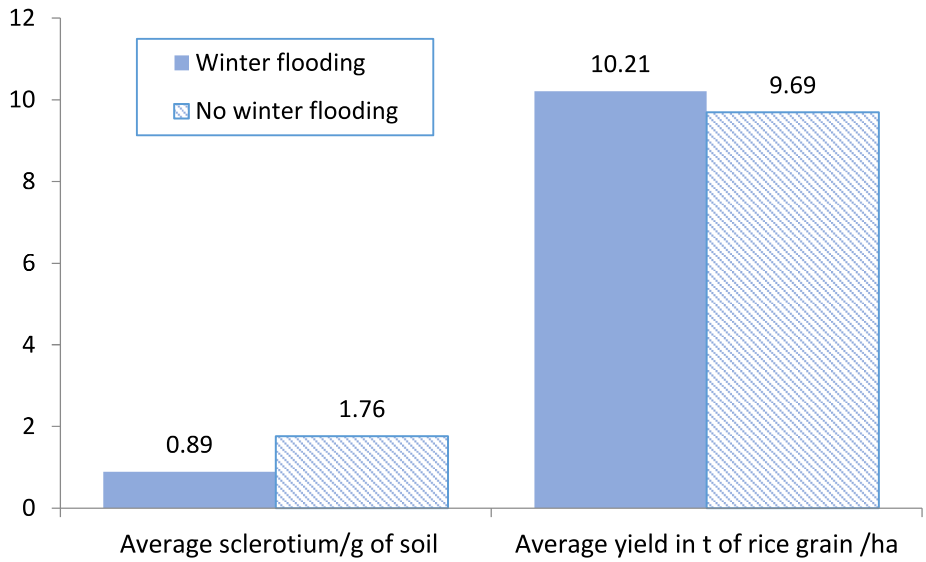


**Figure 3.8.4.** Comparison of yield in ton per hectare and number of *Sclerotium* per gram of soil when fields are winter flooded and when they are not flooded.

Likewise, weeds can be effectively controlled by **continuous flooding**. In an experiment in the Philippines (Bhagat et al. 1999), weed biomass increased when improved water management practices were applied instead of continuous flooding. In 75% of all cases, weed control was compromised when soils were no longer covered by water. Yields decreased in 50% of all cases; and no difference was shown for the other 50%.

The average value of rice production was estimated to be US$2659 when fields were flooded continuously and US$2338 when fields were management with improved water management practices. The decrease in weed control was not valued in monetary terms. Instead, we displayed the effect of continuous flooding (saturated soils) and improved water management (shallow water ponding) on weed growth in figure 3.8.5 below, and expressed the effect in biophysical measurements. Weed growth increased with improved water management practices, and therefore weed control decreased by almost three times. Yields, in this experiment dropped by about 10%.

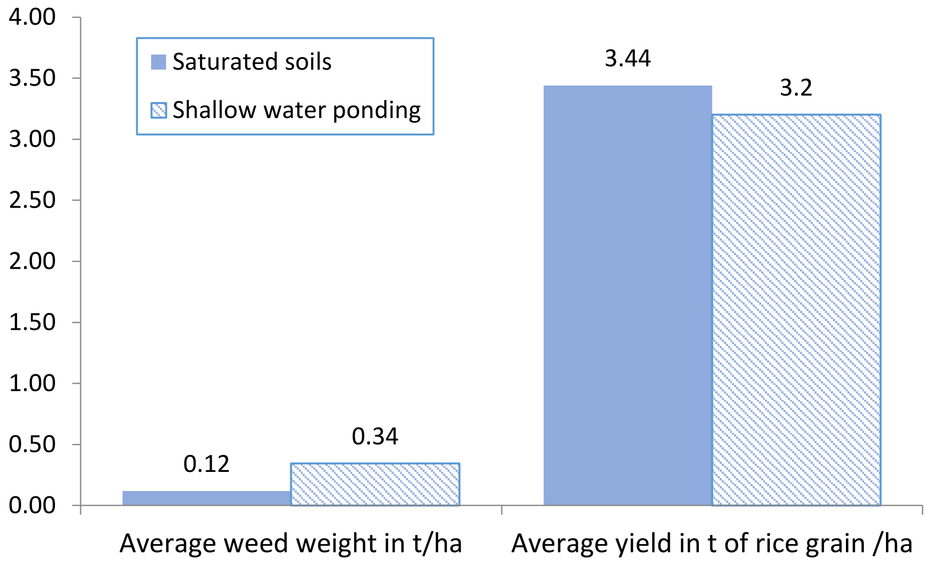


**Figure 3.8.5.** Comparison of yield in t per hectare and weight of weed in kilogram per hectare when fields are fully saturated and when they are only shallow ponded.

However, one needs to be aware that while flooding is a good control mechanism for some diseases or weeds, in other cases the opposite might be the case. Experience has shown that temporary drainage of rice paddies can successfully combat pest such as grasshoppers, water weevils, or whorl maggots by affecting their respiration (IRRI, n.d. b). There was no data in this respect in any of the five case study countries however.

Researchers have found that changing the nutrient composition of crops, the rate of fertilizer use can influence plant defenses (Chen & Ni, 2011). The research shows that there is more plant damage from pests in nitrogen-fertilized crops, as high nitrogen levels in plant tissue can decrease resistance and increase susceptibility to pest attacks. The use of **no or lower fertilizer rates** can therefore be highly effective in suppressing certain pests, yet yields may be lower. There was no data showing these effects in any of the five case study countries however.

While decreasing the use of fertilizers can increase the plant resistance to pests, fertile soils, on the other hand, have positive effects on pest and weed resistance of rice crops. Research in upland systems has shown that **improving soil fertility** through the introduction of leguminous shrubs and trees or the incorporation of crop residues and green manure, for instance, enables rice crops to better compete with parasitic weeds such as Striga (Elezein & Kroschel, 2003; Kayeke 2007). There was no data in this respect in any of the five case study countries.

Many other cultural mechanisms that suppress pests, weeds and diseases are available. Planting trap crops, crop rotations or intercropping are just a few more examples. Yet as our literature research has shown there is hardly any scientific evaluation of these practices available. Likewise, hardly any studies covered biological, mechanical or genetic control in the five case study countries.

### b. Genetic control

An experiment conducted in Senegal by Rodenburg et al. (2014) tested the weed competiveness off two high yielding local varieties: Sahel 108 and Sahel 202. Sahel 202 is known as a relative strong competitor against weeds, while Sahel 108 is identified as a relative weak weed competitor. The advantage of Sahel 108 is a shorter growing period 113 days versus 125 days for Sahel 202. The experiment was repeated over 3 years. Figure 3.8.6 shows an average decrease of weed biomass of Sahel 202 over Sahel 108. However, in none of the three seasons this difference has shown a significant effect. Also yield is higher for the Sahel 202 variety and this effect was significant in 2 out of 3 seasons.

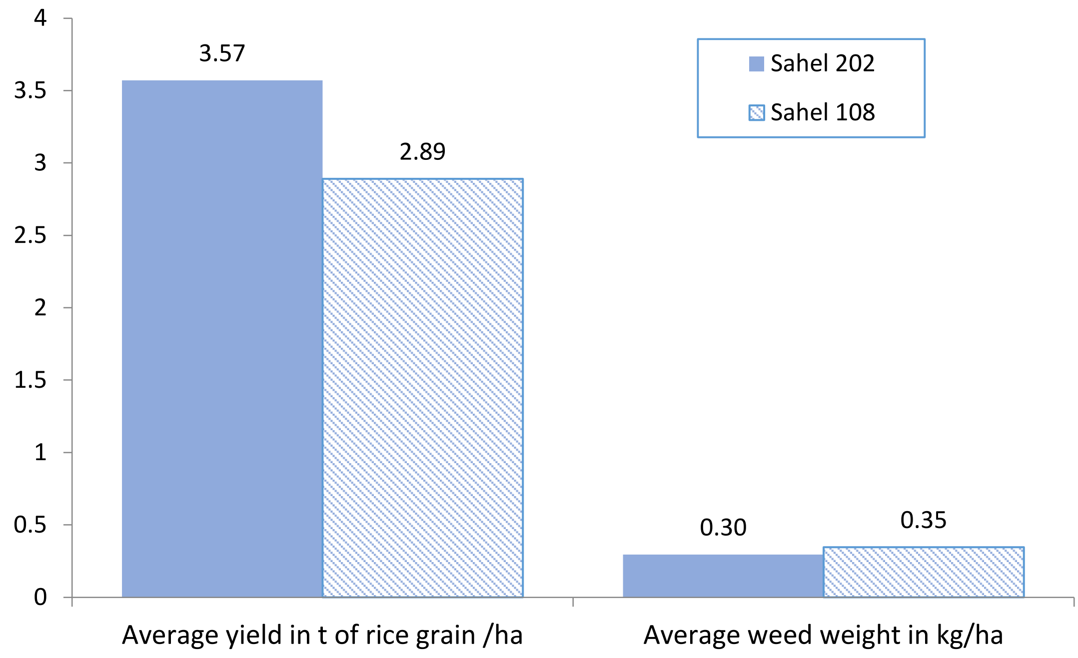


**Figure 3.8.6.** Comparison between two local varieties in Senegal – Sahel 108 versus Sahel 202 - in terms of weed resilience and yield.

### c. Manual, biological and chemical control

In irrigated lowland systems in **Senegal** (Riara et al 1987), weed control improved when biological weed management (through *Azolla*) together with manual weeding was used instead of herbicides or biological weed control only.

The use of herbicides outperformed the use of biological control only, the latter being the least efficient of all practices.

There were slight differences in yield between the different practices, with biological control combined with hand weeding showing the best results. However, there was no statistical significant difference between the different practices.

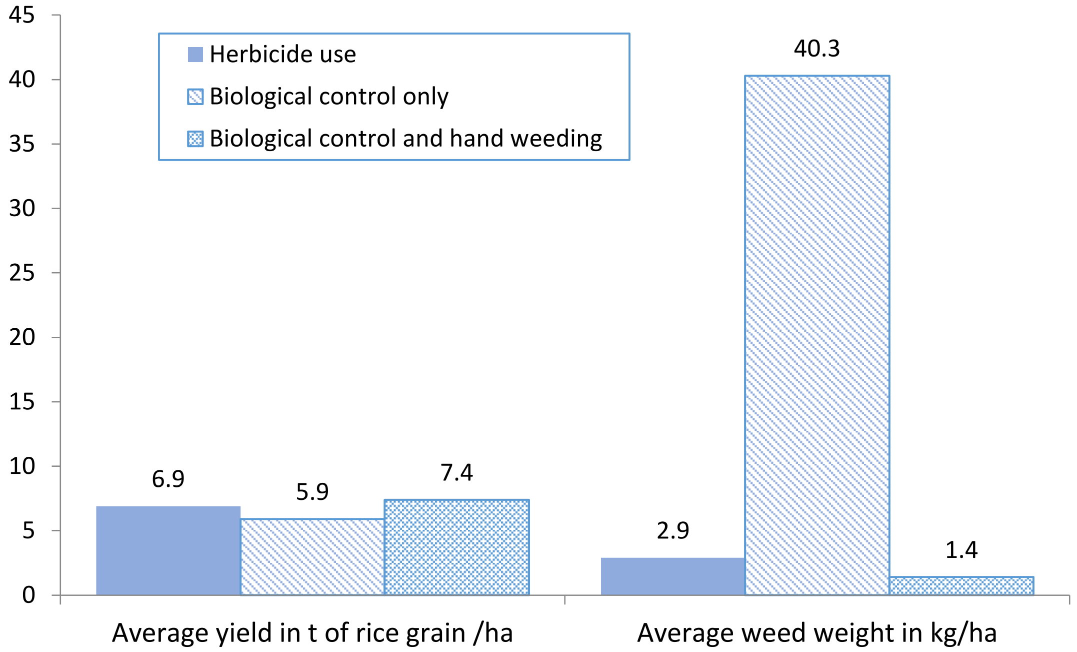


**Figure 3.8.9.** Comparison of yield and weed growth responses between herbicide use, biological weed control only and join hand weeding and biological control. Please note the metric of the Y axis relates to different measurement units. The Y axis is formatted in logarithmic scale.

A study undertaken in rainfed lowland systems in **Cambodia** (Rickman 2001) compared the effect of manual weeding to herbicide use. The study showed that manual weeding was more effective than herbicide use in terms of both yields and weed control.

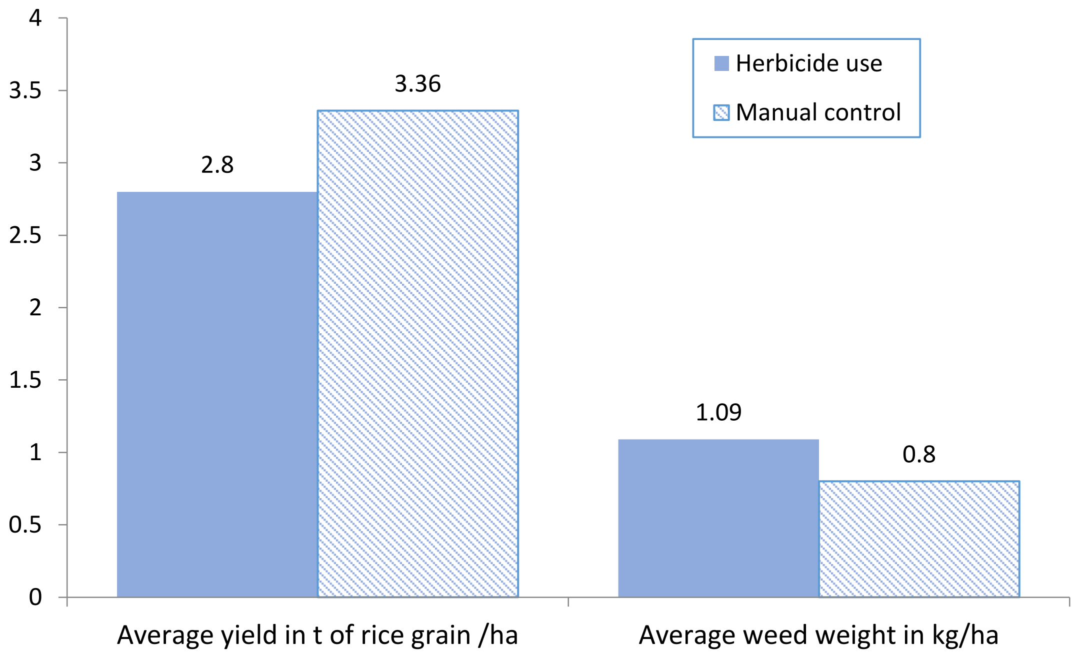


**Figure 3.8.10.** Comparison of yield and weed growth responses between herbicide use and hand weeding and biological control. Please note the metric of the Y axis relates to different measurement units.

## 3.9 Other invisible benefits related to rice production

Invisible benefits related to rice production are not only absent from national accounts, but also often overlooked in the primary research literature. This section will list and describe those ecosystem benefits that could not be identified in the case study literature, but are known to play an important role in rice-agro-ecosystems through information from other sources:

### a. Dietary diversity

Dietary diversity defined as the number of different foods or food groups consumed over a given reference period, can be enhanced by diverse rice-agroecosystems. Many rural households depend on monotonous diets that are too high in carbohydrates and too low in animal source foods and micronutrient-rich fruits, fish and vegetables. Access to a diversified diet is often constrained by lack of purchasing power, limited expertise and limited availability. Experience has shown that more diversified farming systems that contain horticultural or aqua cultural components are one way to improve households' availability and access to such animal source foods, fruits and vegetables. While high external input production systems, usually monocultures, may increase rice yield (compared to more diverse, yet lower external input systems), they bear the risk to lead to monotonous diets that are high in carbohydrates and low in animal source foods and micronutrient-rich fruits, fish and vegetables (trade-off). More diverse, yet less external input systems may lead to higher dietary diversity and better nutrition, with potential trade-offs in yields of rice. These hypotheses are built on many assumptions however as the choice of diet, as said above, is not only a question of availability and access to a more diversified choice of food produce, but also, more importantly, a question of customs and behaviour.

Rice production in flooded environments has the large potential to improve the dietary diversity of rural populations. Many traditional systems in Asia are based on concurrent cultivation of rice and fish, whereas other systems alternate between rice cultivation in one season and fish culturing in the other. Still other systems—especially those in more commercialized rural economies—rely on separate and permanent fish culturing systems (Dey et al 2012). Fish in rice-fish systems does not refer only to fin-fish; it includes the wide variety of aquatic animals living in rice fields: shrimp, crayfish, crabs, turtles, bivalves, frogs, and even insects. Farmers may also allow aquatic weeds, which they harvest for food (Datta & Banerjee 1978). Surveys in Cambodia, for example, have documented the harvest of over ninety different organisms from rice paddies and used daily by rural households (Balzer et al 2002; Halwart and Gupta 2004). These wild and gathered foods from the aquatic habitat provide important diversity, nutrition and food security, as food resources from rice-field environments which supply essential nutrients that are otherwise not adequately found in diets.

### b. Genetic variability

It is also important to acknowledge that rice agro-ecosystems not only differ in terms of species diversity leading to a higher dietary diversity, but also regarding genetic diversity of rice itself. Evidence has shown that individual cultivars, strains and breeds of the same (rice) species do have significantly different nutrient contents (Kennedy and Burlingame 2003). In fact, there are thousands of different rice varieties, some of which have been around for centuries while others are new hybrids bred to increase rice yields or reduce the susceptibility to rice pests.

With its long history of cultivation and selection under diverse environments, rice has acquired a wide adaptability enabling it to grow in a range of environments, from deep water to swamps, irrigated and wetland conditions, as well as on dry hill slopes. Probably far more than any other crop, rice can grow under diverse geographical, climatic and cultural conditions.

The quality preferences of rice consumers, over millennia, have resulted in a wide diversity of varieties specific to different localities. Although the exact diversity cannot be gauged, it is estimated to be around 140 000 different genotypes (Rai 2003). Paddy rice farmers in Asia generally grow a number of photoperiod-sensitive rice varieties adapted to differing environmental conditions. These farmers regularly exchange seed with their neighbours because they observe that any one variety begins to suffer from pest problems if grown continuously on the same land for several years. The temporal, spatial, and genetic diversity resulting from farm-to-farm variations in cropping systems confers at least partial resistance to pest attack.

### c. Water purification

The rice terraces that characterize upland rice farming systems throughout east, south and southeast Asia are ancient (and current) feats of farmers working with nature. In these often mountainous areas that could otherwise not sustainably produce crops, the act of growing rice remains labour intensive— built and maintained by generations of farmers sculpting the land, and preserving water and soil. Water supply, the most important aspect of rice terraces, comes from rivers and mountain streams; irrigation flows through the same complex canals and river ways that are centuries old. The different levels of rice terraces allow water to flow successively down each level.

Watershed management on the part of family farmers in Asian rice production systems extends well beyond the paddy fields themselves. In Ifugao, the Philippines, where the rice terraces have been named a UNESCO World Heritage Site, rice terraces are supported by indigenous knowledge management of muyong, a private forest capping each terrace cluster. The muyong is managed through a collective effort and under traditional tribal practices. The communally-managed forestry areas on top of the terraces are highly diverse, harbouring indigenous and endemic species. The terraces and forests above serve as a rainwater and filtration system and are saturated with irrigation water all year round. A biorhythm technology, in which cultural activities are harmonized with the rhythm of climate and hydrology management, has enabled farmers to grow rice at over 1000 meters (Concepcion 2003).

### d. Groundwater recharge

Groundwater recharge is a provisioning ecosystem services provided by rice paddies. When rice fields are flooded, standing water percolates through the soil and recharges the groundwater. Although a significant part of the water flows back into river or drainage channels, around seven percent recharges the underground aquifers (Abdullah, 2002).

### e. Moderation of extreme events

The moderation of extreme events such as flood prevention or mitigation is a regulating ecosystem service provided by rice paddies when they hold water during heavy rainfalls. Rice ecosystems may serve effectively as landscape elements that capture floodwaters during monsoon periods, and thus recharge groundwater that may be used during drier periods. This is the concept behind the "Ganges Water Machine" (Smakhtin 2013), a long debated concept to find a solution to water issues in the Ganges River Basin, where 80% of the monsoon-driven river flow occurs during a four month period. While rice systems do serve in this capacity already, the logistics of optimising such functions across an entire river basin – and around 30% of India's cultivable land- remain challenging.

### f. Cultural heritage

As the product of indigenous agricultural innovations and communal decisions and customs, smallholder rice production systems provide a living testament to the possibilities of a harmonious relationship between human and nature. The ancient Subak water management systems developed more than 1000 years ago for paddy rice cultivation on Bali Island, Indonesia are an excellent example of this (UNESCO, n.d) .  Subak is a traditional ecologically sensitive irrigation system that does not simply supply water to rice fields; it is a cultural service that considers the entire water needs of the community and provides a pulsed provisioning of water to that community.  Paddy fields in Bali were built around water temples and the allocation of water has traditionally been made by a priest, in accordance with teachings of the Hindu religion.

# 4. Conclusions

## 4.1 Reliable conclusions

### a. Typologies need to zoom in on management practices and systems to reflect the diversity of rice farming and its values

In order to draw broader conclusions on the possible invisible costs and benefits of rice production, it is important to define and identify a typology of the major rice growing systems, as each has differing attributes making them capable of generating different sets of ecosystem services. Figure 1 illustrates the key systems for rice production, and our subsequent analysis has made use of this basic typology.

Yet within the specific objectives of this study, to identify visible and invisible costs and benefits of rice agro-ecosystems; i.e. externalities, it has been necessary to set up contrasts, to compare generation of externalities in different instances. Farmers within each agroecological zone and corresponding farming system (for example lowland rice) have a wide range of production options, and it is the choice of these options that will determine the generation of negative and positive externalities. Thus the basic typology of farming systems needs a much finer refinement disaggregating into specific options and farming practices.

The results have confirmed the need for practice and location specific typologies to show the full range of externality benefits and costs. As our results clearly show, environmental impacts and ecosystem services linked to rice farming strongly respond to the type of agricultural management practised. Needless to say that water quality is affected by the type and the amount of pesticides and herbicides used as well as the type and amount of fertilizers that run-off to nearby waterways and lakes. Air quality is only affected when rice straw is burnt and ammonia emissions are released from fertilizers – if straw is managed in a different way and when fertilizers are not applied to the soil, there is no air pollution. Electricity or heat can be produced from rice husks when adequate energy facilities are available, and rice straw can enhance the organic matter content of soils when incorporated into the agricultural fields – yet when none of this is done, this ecosystem service will not materialize. While these facts are obvious, rice farming or agriculture in general, is too often categorized as one homogenous activity, when in reality farming is extremely diverse.

This study has therefore made an attempt to go beyond production systems and rice growing environments, zooming in on the different ways in which rice is produced. While evidence gets scarcer the more detail is added to the typology – a true challenge indeed - the authors of this study are convinced that there is no other way if one aims to do justice to the diversity of the farming sector.

### b. Results confirm that a trade-off, synergies, and "benefits without losses" analysis is mandatory if the study is to inform policy

In principal, there is no such thing as a free lunch. It is often assumed that if farmers adopt more environmentally-friendly practices, they inevitably will incur reduced yields. In fact, our results illustrate the diversity of possible outcomes. Indeed in some cases there are distinct trade-offs; for example, a reduction in herbicide use will lead to improved water quality, but in some cases there will a decrease in rice yields. One can reduce water consumption by practising water saving management practises, yet a specific service provided by permanent flooding, weed management, will most likely decrease. Certainly the trade-offs with respect to weeding options are amongst the most problematic in rice production (see Figure 2 and 13 in van Dis et al. 2015)

However, in rice production systems there may also be instances of win-win. Let's take the example of residue management. The studies by Eagle et al (2001) and Cintas and Webster (2011) have clearly shown that a change from rice straw burning to rice straw incorporation not only significantly reduces air pollution but also increase yields – a clear synergy (see section 3.3).

What is perhaps most striking in many of the biophysical results reported here in the vote-counting analysis is the number of instances where there were "no significant differences" –in terms of yield with benefits in other ecosystem services. This indicates that in a majority of cases, yields could be maintained while other ecosystem services could be generated through the application of holistic practices (see Figures 3, 4, 6, 7, 8 10 11, 12, 14 and 15 in van Dis et al. 2015). This is an instance where "no significant difference" is a significant finding; yields in these findings were not being sacrificed through more environmentally-friendly farming practices, in fact they were being maintained.

One also needs to keep in mind that trade-offs are the extreme cases. For instance comparing AWD with continuous flooding gives a clear trade-off between two extreme alternatives, but if AWD is only done partly, one does not have to compromise on yield, while water use reduces.

The study can bring these opportunities for synergies to the attention of decision makers and point out where trade-offs can be minimized, synergies can be maximised, and yields can be maintained while ecosystem services are being generated and enhanced.

Expressing these trade-offs, synergies or instances of "yield maintenance with benefits", i.e. maintaining yields while generating positive externalities" in monetary values will help to make them visible. This will inevitably facilitate the decision making process as it helps to comprehend both the upside and downside of a particular choice – expressed in the same currency and thereby easily comparable. Tables in the Annex give an overview of the trade-offs that were monetized. However, it should be recognised that there are still a number of critical limitations in the application of monetary values to the biophysical findings, as discussed further in 4.2.b below.

### c. Private benefits needs to be weighed against public benefits

A trade-off analysis is worthless unless the study specifies WHO is affected and to WHAT DEGREE. Our results have shown that where there is a private benefit to the farmer, there is often a public cost. By changing a farm management practise, the negative public benefit often becomes positive, yet the private benefit decreases at the same time. For example, if Senegal changed from herbicide use to hand weeding and biological control throughout all irrigated lowland systems in the entire country, herbicide damage costs would be reduced by US$ 95,000, a public benefit. The private benefit of all farmers together would be reduced considerably however, to a much higher degree than the public benefit. The revenue of rice would drop by US$1,324,000, almost 15 times the value of the public benefit. It is therefore very unlikely that farmers would adopt such change unless policy makers would compensate for the loss by providing incentives.

There are other cases however, where a change in management practise would generate benefits both for the farmer and society. For example, when upscaling the results from a study on SRI in rainfed lowland systems in Cambodia, farmers in the scenario analysis gained a total of US$ 801,040,000 in revenue. At the same time, the GHG emission costs dropped by US$ 257,920,000 – a public benefit. In this case, policy makers would not need to provide any economic incentives to encourage a change from conventional to SRI. The private benefit should be compelling enough, once the case is made and farmers have seen the advantage of the management change. Policy makers could promote the adoption of this practise through awareness raising, or education to accelerate the uptake by the farmers.

To be quite realistic, farmers are conservative and risk-averse, and most cannot shoulder any transition costs to new practices; thus, there remains a role for public policy and support to assist farmers in a transition, particularly through training to systems such as SRI that depend on farmer knowledge management.

## 4.2 Promising directions

**a. System integration – comparing different dependencies and impacts at the same time**

As this study has shown, rice production, or agriculture in general, has many different positive and negative impacts. Yet most of these environmental and socio-economic benefits and costs are often studied in isolation from each other, despite them being closely interconnected.

Agriculture, the science, art, or practice of cultivating the soil, producing crops, and raising livestock, is by definition geared towards production with a singular focus on commodity yields in most scientific studies. The focus on multiple goods and services for the sake of sustainability is rarely ever studied in agronomy. Benefits from agricultural production, in most analyses, have been traditionally measured through crop yields and financial returns, with little or no attention to overall resource efficiency, diversity of outputs, risk reduction and non-commodity outputs (Silici 2014).

By contrast, the focus of TEEBAgFood is on trade-offs between yields and services as water quality, GHG emissions and habitat provisioning. This provides a valuable context for bringing in an assessment of the negative externalities as "invisible" costs to production. While it is impossible to average different biophysical measurement parameters, a monetary valuation allows converting all impacts into the same currency. Each of these can be assigned a monetary value, thus identifying the "true costs" of a production system.

In order to arrive at this "true cost" one needs to bring all different benefits and costs into the equation. However, reliable data is most often only available for two different ecosystem services or environmental impacts. And even when this data is complemented with some modelled values as tried in this study, the overall framework is still incomplete. The study authors therefore opted to present results in separate figures, sometimes in monetary values, and sometimes in biophysical units, without trying to sum them up and arriving at one single – and incomplete – figure.

**b. Improving the collection of comprehensive and quality data from field studies**

The above raises an important point – while it seems paramount to create a valuation framework that can analyze the combined effects of entire production systems at the same time, there are no studies that provide such a comprehensive set of data. Even for rice, a supposedly well researched main staple crop, there are surprisingly few comparative studies that document impacts on multiple ecosystem services (even as simple as, for instance, crop yield and water quality) from different management practices. The majority of all studies focus on yield increases. Few studies focus on an environmental impact or ecosystem service at the same time as yields or economic returns. Hardly any take more than two variables into account.

Some ecosystem services and benefits are not covered at all in the research literature from the five case study countries: Flood control, groundwater recharge, dietary diversity, tourism and recreational activities, other cultural services from rice, ecological resilience against pests or extreme weather events, and carbon storage.

Also certain environmental impacts have not been researched in these countries: eutrophication from fertilizer run-off or land pollution. There is also lack of data when it comes to the amount of inputs such as pesticides or the use of labor.

An area of particular concern is the ongoing dispute of environmental and health costs related to pesticide use, incl. herbicides. As explained in section 3.1., pesticide, incl. herbicide damage costs depend largely on the distance of the field to the next waterway. Hence values are very context specific. Furthermore, irrespective of these context specific differences, there is increasing evidence in the global literature that there may be direct human health costs, and that there is the longer term development of resistance or reduction of ecological infrastructure to prevent or impede weed infestations when certain chemical are used. For example, a recent publication by the International Agency for Research on Cancer (IARC), caused a large uproar in the global media, as the study authors announced that glyphosate, a widely used herbicide, also in rice, is probably carcinogenic to humans (Guyton et al 2015). The results of this study are highly contested, however, as many different interests are at stake (Cressey 2015). The results presented in this report related to herbicide and pesticide use therefore need to be treated with a lot of caution, since the data that underpins the valuation methodologies might not adequately reflect the full range and magnitude of negative effects.

Related to this are issues of data quality; a considerable amount of data and observation on rice production and its effects, particular those with a focus on agroecology, has been published in the "grey literature": not formally published through a commercial process. Innovations in agroecology have generally come from farmer groups or practitioners, and have been shared through lateral, farmer-to-farmer routes rather than conventional agricultural research and extension channels. A good case in point is Systems of Rice Intensification, which was developed in 1983 by a French Jesuit Father priest in Madagascar. It has been shared and elaborated by many farmer groups, now around the world. The initial documentation of its success in the scientific literature was due to Dr. Norman Uphoff at Cornell University, although proponents and critics of the system continue to disagree (Uphoff 2003; Uphoff and Kassim 2011; Surridge 2004). Yet even critics of the system acknowledge its wide and impressive rate of uptake among farming communities. There is a growing recognition that the system delivers not just good yields, but also healthier soils and other additional benefits. Thus, the reality on the ground has only recently, and partially, been reflected in peer-reviewed scientific literature. The debate brings interesting light on how agroecological methods might better be evaluated in the literature (Glover 2011). One key point is that smallholder farming practices rarely if ever conform to an abstract norm as farmers adapt recommendations to their needs and conditions; thus criteria of "performance" (understood more widely than yields alone) should be considered.

### c. Enhancing models that mimic agro-ecological processes where specific data points are missing

As just mentioned, the results presented in the section above rely on primary research data where possible. Modelling of biophysical processes can help to fill data gaps where there is no primary data. Some models exist and have been found to be reliable. Others are still in need of development.

Usually, only two parameters are compared with each other such as rice yields versus water consumption. Sometimes it is possible to model or estimate other related variables and add them to the equation, as demonstrated in section 3.4 for instance where we assumed that rice husks were used to produce energy. These estimations are fairly solid as we know how much rice husk is produced for each kilogram of rice. Yet it becomes more complicated when one tries to model entire agro-ecological and biophysical processes that rely on many different variables, such as the release of greenhouse gas emissions or groundwater recharge for example.

The methodology used in this study, published by Koch & Salou (2015), models the release of GHG emissions from rice production systems is mainly based on global assumptions. While these allow for a rough calculation to get an idea of the order of magnitude of GHG emissions that rice production systems release, they are not appropriate to model the difference between different rice management practises.

This becomes obvious when one compares actual emission data with modelled data. Let's take the Cambodian study of Ly et al (2013) which compared SRI with conventional management (see section 3.5). For this system comparison, primary research data for GHG emissions from fertilizer volatilization was available. The monetary valuation for GHG emissions in Cambodia's RL paddies resulted in an average cost of US$690 for conventionally managed systems and US$586 for SRI per hectare of rice production – a reduction in costs of 15%.

When applying the GHG emission model to the same case ignoring the actual data given in the journal papers, the results look different. The GHG emission costs for both systems amount to US$192 per hectare. As this example shows the results are considerably lower than the measured data – about three times lower than the real costs. And second, they are the same for both treatments; hence, there is no difference at all between the two systems. This can be explained by the fact that the model is based on global assumptions, and is not able to distinguish between differences at practice level. We can therefore conclude that the modelled data has to be used with caution when trying to show the differences between different management practices and systems. However, in the absence of original GHG emission data for some of the comparisons, it can be used to show the order of magnitude of GHG emission costs when compared to other costs and benefits derived from rice production.

The study authors call for the development of more precise models that are able to mimic agroecological and biophysical processes. Without these, researcher will remain restricted to primary research data.

## 4.3 Outstanding areas for further analysis

#### a. From practices to holistic farm management

Within the TEEB framework a more holistic analysis is needed. The TEEB Foundation Study recognises that "ecosystem assessments should be set within the context of contrasting scenarios - recognising that both the values of ecosystem services and the costs of actions can be best measured as a function of changes between alternative options." Yet the economic modalities for comparing, in a genuinely holistic sense, between two systems of production, rather than the outcomes of two practices (such as use of pesticides versus reduction or non-use of pesticides) is a methodology in need of development.

One solution to addressing the practical limitations of a standard cost-benefit analysis would be the use of a dynamic systems approach. This could lend a number of strengths, including a much-needed time dimension, since practices and interventions might have their impacts not immediately, but may have crucial outcomes over time. In addition, a dynamic systems approach can handle non-linear relationships, given that the marginal impact of a given activity on ecosystem services are unlikely to be constant. Agricultural ecosystems can react non-linearly to interventions: they could for instance balance small interventions such as soil salinity in rice paddies, be gradually affected by a stronger interference up to a threshold point through for example a tsunami causing major saltwater intrusions in a coastal zone, and then collapse.

From an agroecological perspective, practices or sets of practices are not the operational focus; practices are measures that are adapted and modified in locally context-specific ways to optimize interactions in agroecological systems, comprising both biological and social aspects. Proponents of Agroecology state quite clearly that it is a set of principles that take technological forms depending on the socio-cultural, economic and environmental realities of each community, or situation. Thus, "diversity" as a principle may be actualized in many forms such as intercropping or agroforestry; each of these are ways to optimise interactions between crops, shade, pests, soil organisms, etc. Many practices may also entail social elements such as increasing social interaction, learning and empowerment as part of the system.

Yet it is evident in the literature that most research focuses on specific practices; in the scientific approach of introducing one intervention while holding all other factors constant. And even if studies report on a particular "system", the set of practices, and degree of their optimization, is quite variable. Where specific management systems have a number of key practices or principles (such as Systems of Rice Intensification, or Conservation Agriculture), research studies often focus on one or two of these, rather than comparing the implementation of all practices against the absence of all. Additionally, different definitions of practices and systems are used within specific contexts and cannot easily be simplified or homogenized in light of the peculiarities of each context.

### b. Lack of monetary valuation methodologies of agro-ecosystem benefits

While there is very little research undertaken on ecosystem benefits of sustainably managed rice production such as building soil fertility or, building resilience to pest invasions as well as weather and climate variability, there is also a lack of valuation methodologies that can capture these kind of services. Or put differently, there seems to be a clear lack of valuation methodologies *because* there is a lack of biophysical data related to agro-ecosystem benefits beyond rice production. More work tends to be undertaken on environmental impacts related to agriculture as opposed to the benefits that agro-ecosystem can generate if they are sustainably managed. As already stated in Section 2.4, methodologies can only be built once one knows the attribute that is being valued.

An effective TEEB analysis should be able to zero in on these "benefits" as much as the costs of agricultural production. Current methodologies include them as avoided costs (e.g Sandhu et al 2015), but they should also be displayed as a benefit contributing to the building of an ecological infrastructure over time. While there is a risk of double-counting, it would be a mistake to omit this function all together as it assures the sustainability of the agro-ecosystem on the long run.

This is perhaps best illustrated with the rice example: rice under conventional production may use high inputs of both pesticides and fertilizer. Use of pesticides over time will have negative impacts on the natural enemy community that can provide natural forms of pest control. In a TEEB analysis as currently constructed, this will appear only as a cost applied to water quality instead of cost incurred by the bio-physical "infrastructure" underpinning production. In order to value this cost, one would first need to find accurate indicators for the loss of the natural enemy community in order to measure the biophysical changes to the ecological infrastructure. In a second step these biophysical metrics would need to be valued in monetary terms with adequate valuation methodologies.

Under agroecological approaches to rice production, perhaps best described in the work of Settle (1995), ecosystem services come in to play in critical and nuanced ways, contributing to a mechanism that supports high levels of natural biological control. If organic matter is increased early in the growing season, abundant populations of detritus- feeding and plankton-feeding insects will be fostered, usually peaking and declining in the first third of the season. These insects have no impact on rice yields, either positive or negative- but their populations provide natural enemies of rice pests a "head start" – to build up their populations early in the season so as to be able to strongly suppress the pest populations which enter the paddy in mid-season. Pesticides early in the season will prevent the strong build-up of natural enemies, killing both them and their early-season food source. Minimal application of organic material into paddy soils, and a greater dependence on inorganic fertilizers will similarly impact the early season build-up of natural enemies. And clearly, the process of building a strong ecological community is a multi-year process that nonetheless could be reversed in one year of high applications of agricultural chemicals.

The process as described above - only part of the complex rice ecosystem - is precisely the kind of "dependence upon biodiversity" that TEEB seeks to make visible. Yet, for lack of tools and methods, it is not clear at this point how numbers and values can be assigned to such "internal" ecosystem functions as natural pest control and natural fertility maintenance, as their value goes well beyond the avoided cost of polluting water. It may, or may not, be accurately reflected in the benefit of yields, but the additional benefits, of building natural capital need also to be reflected.

#### c. Need to better adapt current models for monetary valuation to the realities of developing countries

The biophysical modelling of the impacts of air, land, and water pollution from pesticides, for example, depends on many factors. Besides the dispersion of pollutants, which can depend on wind speed and direction, precipitation, and temperature, factors that differ according to the level of development within countries or regions include the consumption of meat, vegetables, and/or drinking water. All of these factors have been taken into account on a continental level when calculating the impacts to local populations in this analysis, however, national or sub-national differences cannot currently be accounted for.

Models that have been developed to quantify the impact of pollutants emitted to air, land, and water, which include USES-LCA2.0 and USEtox2.0, require large capital and time investments to develop. Hence, these types of models either predominantly use data points based on developed regions from where the funding is provided, or calculate global averages which lack the required robustness or granularity to be utilised in very site-specific circumstances. This can lead to the averaging of impacts across regions or countries, which may not provide a true reflection of the impacts caused. This highlights the need to develop either sub-national, national, or regional models that can take into account the multitude of factors the can influence the dispersion and intake of harmful pollutants.

#### d. Need to link economic valuations to market costs, and avoided costs for the farmer

The valuation methodologies that have been employed to demonstrate the human health and ecosystem impact from the emission of air, land, and water pollutants reflect the costs borne by society, not by the farmer. Related to section 4.3(c) above, this can be attributed to a number of factors.

Firstly, all of the models that can be utilised to calculate the biophysical - human health and ecosystem - impact, calculate the effect on society. The effects on wider society are more broadly understood and have a greater amount of scientific research supporting the calculations developed in models such as USES-LCA2.0. Conversely, health impacts experienced by farmers, or farm workers, are poorly understood and are rarely quantified in a consistent and systematic way that is needed to compare the effects of the thousands of substances that could negatively impact human health. Similarly, the effect on farm ecosystems cannot be captured in these models as such site specific analysis requires the development of models that are able to take into account the nuances of farm level practices and understand farm biodiversity. Without detailed models, the intricacies present between similar farming systems will be lost, and the effect on farm ecosystems will not be truly represented, and could give a false sense of accuracy.

When valuing the impact of pesticides on human health in this study for instance, the valuation relates to the loss of income suffered from ill health, the loss in productivity and the willingness-to-pay (WTP) to avoid pain and discomfort. These are hidden costs, which in some ways exist in the market, but are not linked to the use and effect of pesticides. In order to make the costs of these impacts visible, the use of market costs may improve the visibility of these impacts, but this will alter the magnitude of the value calculated. For instance, valuing the time lost at work due to acute pesticide poisoning of farm workers may make these impacts more visible to the farmer, but the effect on the farm worker resulting from pain and discomfort, or the loss of income, will not be captured in this valuation. This highlights the need for valuations that are context specific: One needs to be clear on who is the value pertinent to and when.

### e. Opportunity to link to SEEA/Ecosystems Accounting

This project used ecosystem valuation techniques to reflect the values of natural capital in the rice farming sector. Another way to bring the values of natural capital to the attention of decision makers could be through ecosystem accounting techniques. While ecosystem valuations usually focus on the local level, ecosystem accounting methods aim to aggregate information to produce statistical results at the national level. The valuation can be expressed in physical or monetary terms as explained in the previous section.

Following the System of Environmental-Economic Accounting (SEEA) - Experimental Ecosystem Accounting, Carl Obst has developed a draft data model of how to link the ecosystem valuations to national accounting illustrated in the following figures. Each figure zooms in on a different aspect to better illustrate the complexity of the model.

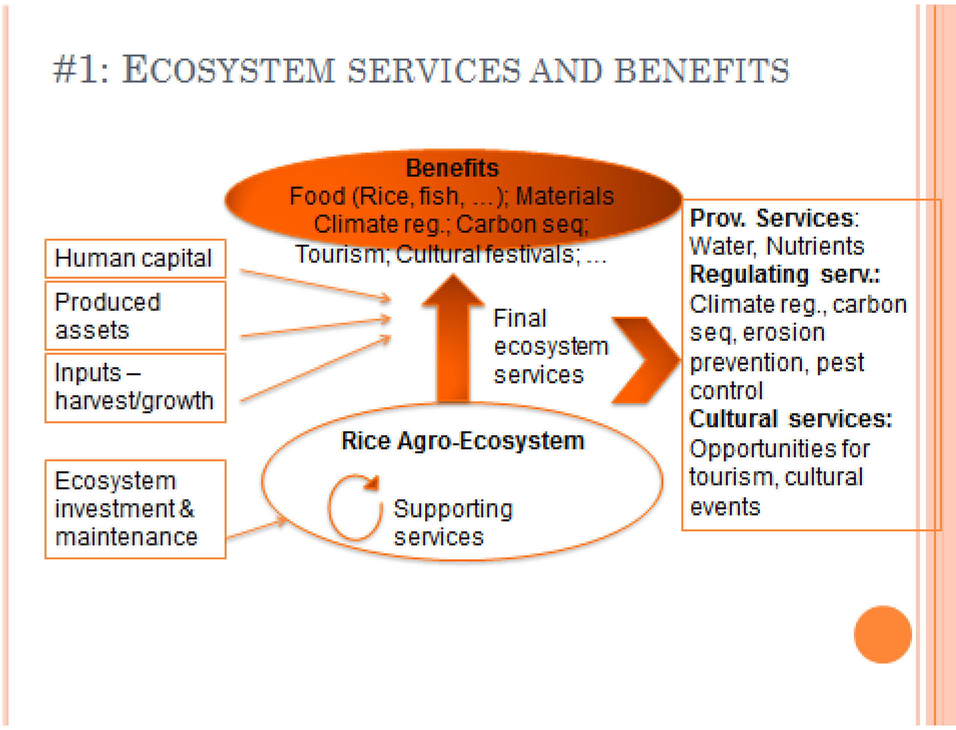


**Figure 4.3.1.** Ecosystems and benefits defined in ecosystem accounting

Figure 4.3.1 explains how ecosystem accounting defines ecosystem services and benefits:

1. Distinction is made between the Rice Agro-Ecosystem (defined as a specific spatial area); the Final Ecosystem services it generates; and the Benefits to which those Final ES contribute.

2. In the case of the Benefits “Food” and “Materials” the Final ES are combined with inputs of human capital / labour, produced assets (including working animals such as buffalo) and intermediate inputs such as fuel/energy, inorganic and organic fertiliser, pesticides, seeds/seedlings, etc.

3. Consequently, the Final ES are not equivalent to the tonnes of food or materials but rather are considered embodied in the benefits. The actual Final ES are the services provided by the rice agro-ecosystem itself – e.g. water taken up by the plant, role of soil nutrients, erosion prevention, pest/biological control. Commonly in farming systems some of these Final ES may be substituted by intermediate inputs and it is the trade-offs between the source of the inputs that is of interest.

4. For benefits such as “Climate regulation” and “Carbon sequestration” since there is little if any human contribution to the generation of these services the flow of benefits equals the flow of final ES. Note for these to be considered final there must be a direct beneficiary (individual, economic unit or society as a whole)

5. For tourism benefits the contribution of the Rice-agro Ecosystem is that it provides the basis for the tourism activity – depending on the type of tourism there may be a small or large amount of input from humans, produced assets, etc.

6. Depending on the use mix there will be varying levels of investment and maintenance of the rice-agro ecosystem – this should be seen as equivalent to investment and maintenance of produced assets rather than a direct input to the production of benefits. The result of the investment and maintenance will be reflect in the flows of final ecosystem services – generally one would expect that lower investment will lead to lower flows of Final ES.

7. Within the Rice-Agro Ecosystem there will be some supporting services that reflect the functioning of the ecosystem. Examples here include water purification and soil related functioning (e.g. soil formation).The quantity of supporting services will reflect the condition of the ecosystem.

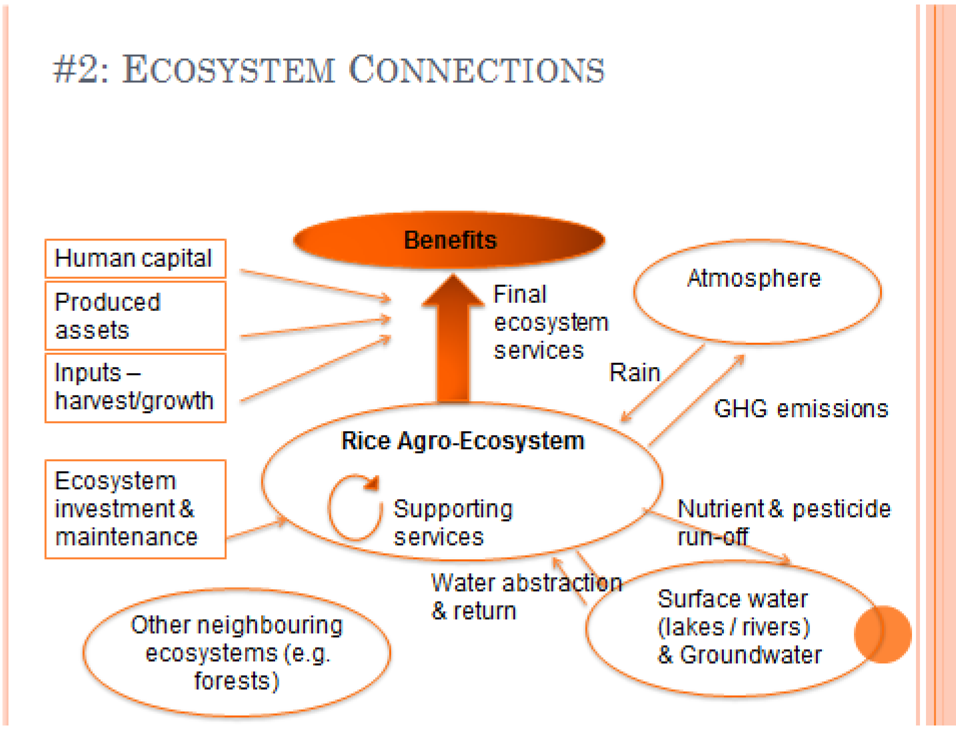


**Figure 4.3.2.** Ecosystem connections

Figure 4.3.2 focuses on ecosystem connections:

1. In this diagram additional ecosystems are added as well as associated flows. We see that water is abstracted and return to surface and groundwater and also is sourced from precipitation. GHG emissions are released to the atmosphere, and nutrient and pesticide run-off is shown flowing to surface and groundwater.

2. None of these flows are regarded as final ecosystem services. At the same time from a data and modelling perspective it will be possible to compare the levels of water use, GHG emissions and nutrient and pesticide run-off quite readily with the Final ES flows, other inputs and with the output of food, materials, and other benefits. The main thing is that by seeing these flows separately we don't just add and subtract flows whose role in the set of relationships is quite different.

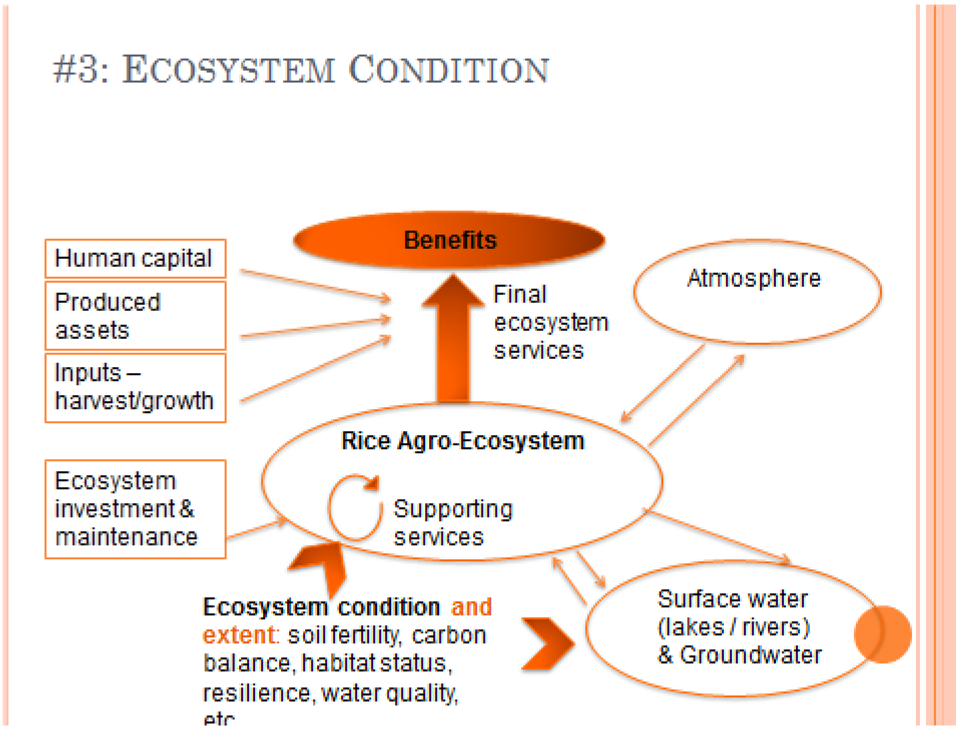


**Figure 4.3.3.** Ecosystem services and environmental impact influenced by ecosystem conditions

Figure 4.3.3 describes how the ecosystem condition influences ecosystem services and environmental impacts:

1. In this diagram the focus is on information relating to the condition of the ecosystems – i.e. the natural capital. Many measures of ecosystems are in fact measures of the condition or change in condition of the asset itself rather than being Final ES. In the case of the Rice-Agro Ecosystem relevant measures of ecosystem condition would relate to: soil fertility, carbon balances (perhaps a link to crop residues); resilience, habitat status, pest levels, health of fish, loss of habitat (i.e. changing the area of the ecosystem or its composition – e.g. by putting in a road).

2. Overall declines in ecosystem condition reflect degradation. This may be offset by sufficient ecosystem investment and maintenance.

3. For surface water and ground water the condition can also be measured. One fact to consider may be the level of pesticides and nutrient loads. In this it is relevant to note that the measurement of the flow of nutrients to water need not imply that the condition of the water is not acceptable. Various thresholds will apply. Equally there may be other sources of pollution.

4. The measurement of ecosystem condition is central to an accounting approach to assessing sustainability since the idea is that the use of the ecosystem should be such that the condition does not decline. Getting agreed indicators of ecosystem condition is therefore an important step.

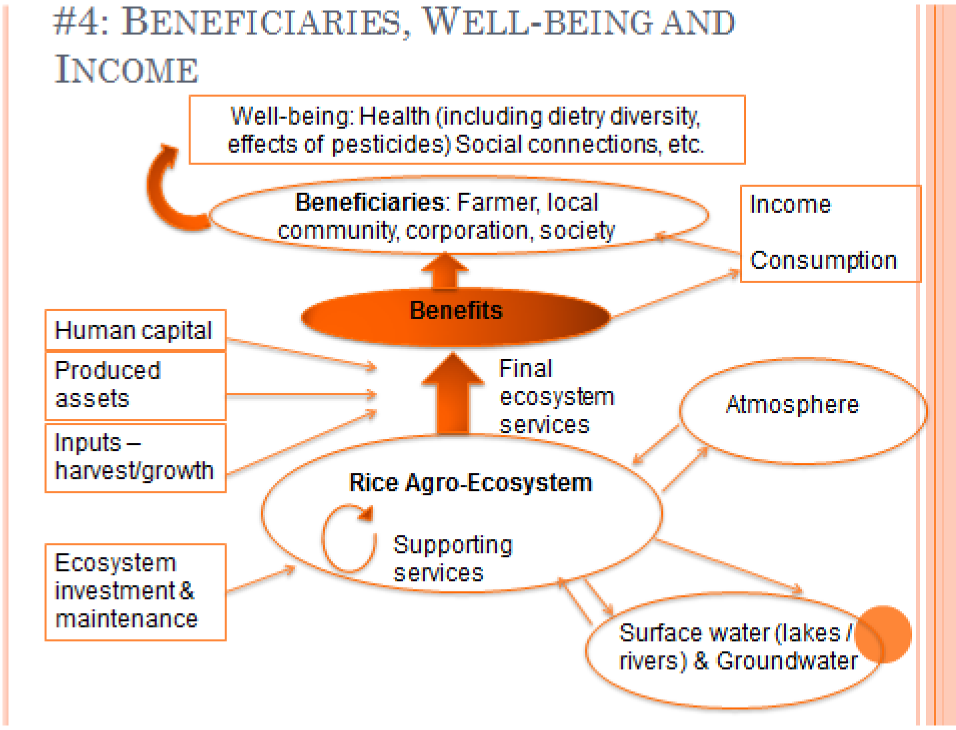


**Figure 4.3.4.** Beneficiaries, well-being and income

Figure 4.3.4 considers the recipients of the benefits, and how they relate to well-being and income:

1. The addition in this last diagram is the extension beyond benefits to consider the recipients of the benefits – the beneficiaries. These may be the farmer, the local community, a corporation or non-local business or society more generally. Understanding the connections between the benefits (food, materials, carbon sequestration, etc) and the beneficiaries in terms of consumption and income will be important in understanding the logic of trade-offs between different models of production. For example, the extent of own-account consumption of rice and fish may imply quite different trade-offs compared to the rice being grown for sale to other economic units. Even if the underlying production function is the same.

2. The final link in the chain is the connection between beneficiaries and well-being. It is here that information on health would come into the model – this might be measured using a range of indicators depending on the issues of focus. Note that well-being will be a function of far more than links to ecosystems.

The challenges to forge credible linkages between the farming-system specific TEEB –AF analysis and systems of national accounts for agriculture is large, and will take quite some work from both the TEEB-AF side and the SEEA-Ag side. But the opportunity to articulate a nuanced, concrete message to policy makers, about how public support for transitions to a more sustainable, holistic agriculture can build national assets, is a very substantial one.

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# Annex

**Table A.1** Pesticide and herbicide damage costs, revenues, pesticide input costs and labour costs in irrigated lowland systems (IL) in US dollars per hectare. Values refer to hypothetical scenarios. Field level results have been upscaled to the entire area of irrigated lowland systems in each country.

| **Water pollution (IL)** | **CALIFORNIA** | | | | **CAMBODIA** | | | | **COSTA RICA** | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Scenarios** | **Revenue, thousand US$** | **Pesticide input costs, thousand US$** | **Labour costs, thousand US$** | **Pesticide damage costs, thousand US$** | **Revenue, thousand US$** | **Pesticide input costs, thousand US$** | **Labour costs, thousand US$** | **Pesticide damage costs, thousand US$** | **Revenue, thousand US$** | **Pesticide input costs, thousand US$** | **Labour costs, thousand US$** | **Pesticide damage costs, thousand US$** |
| No herbicide use - herbicide use | 588,278 | 19,366 | n.d. | 114 | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. |
| No pesticide use - pesticide use | 136,019 | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. |
| Reduced herbicide use - standard herbicide use | 388,692 | 16,700 | n.d. | 1,293 | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. |
| Biological weed control + hand weeding - herbicide use | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. |

| | PHILIPPINES | | | | SENEGAL | | | |
|---|---|---|---|---|---|---|---|---|
| **Scenarios** | **revenue for rice grain, thousand US$** | **Pesticide input costs, thousand US$** | **Labour costs, thousand US$** | **Pesticide damage costs, thousand US$** | **revenue for rice grain, thousand US$** | **Pesticide input costs, thousand US$** | **Labour costs, thousand US$** | **Pesticide damage costs, thousand US$** |
| No herbicide use - herbicide use | 1,915,760 | 4,620 | n.d. | 616 | 243,948 | 33,106 | 498 | 95 |
| No pesticide use - pesticide use | 3,397,240 | n.d. | n.d. | 2,716,560 | n.d. | n.d. | n.d. | n.d. |
| Reduced herbicide use - standard herbicide use | 252,560 | 4,250 | n.d. | 708 | n.d. | n.d. | n.d. | n.d. |
| Biological weed control + hand weeding - herbicide use | n.d. | n.d. | n.d. | n.d. | 1,324 | 48,997 | 849 | 95 |

| | |
|---|---|
| **GREEN** | Increase |
| **RED** | Decrease |
| **BLACK** | No change |
| n.d. | no data |

**Table A.2.** Eutrophication costs, revenues, fertilizer input costs and labor costs in irrigated lowland systems (IL) in US dollars per hectare. Values refer to hypothetical scenarios. Field level results have been upscaled to the entire area of irrigated lowland systems in each country.

| | CALIFORNIA | | | | CAMBODIA | | | | COSTA RICA | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | **revenue for rice grain, thousand US$** | **Fertilizer input costs, thousand US$** | **Labour costs, thousand US$** | **Eutrophication costs, thousand US$** | **revenue for rice grain, thousand US$** | **Fertilizer input costs, thousand US$** | **Labour costs, thousand US$** | **Eutrophication costs, thousand US$** | **revenue for rice grain, thousand US$** | **Fertilizer input costs, thousand US$** | **Labour costs, thousand US$** | **Eutrophication costs, thousand US$** |
| No fertilizer use – mineral fertilizer application | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. |
| Reduced mineral fertilizer use - standard mineral fertilizer application | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. | 6,989 | 682 | n.d. | 0 |

| | PHILIPPINES | | | | SENEGAL | | | |
|---|---|---|---|---|---|---|---|---|
| | **revenue for rice grain, thousand US$** | **Fertilizer input costs, thousand US$** | **Labour costs, thousand US$** | **Eutrophication costs, thousand US$** | **revenue for rice grain, thousand US$** | **Fertilizer input costs, thousand US$** | **Labour costs, thousand US$** | **Eutrophication costs, thousand US$** |
| No fertilizer use – mineral fertilizer application | n.d. | n.d. | n.d. | n.d. | 70,185 | 4,351 | 393 | 0 |
| Reduced mineral fertilizer use - standard mineral fertilizer application | 338,800 | 64,680 | n.d. | 0 | 34,903 | 3,405 | n.d. | 0 |

| | |
|---|---|
| **GREEN** | Increase |
| **RED** | Decrease |
| **BLACK** | No change |
| n.d. | no data |

**Table A.3.** Water consumption costs versus food production in irrigated lowland systems (IL). Values refer to hypothetical scenarios. Field level results have been upscaled to the entire area of irrigated lowland systems in each country.

| | CAL | | CAM | | CR | | PHI | | SEN | |
|---|---|---|---|---|---|---|---|---|---|---|
| **Scenarios** | **revenue for rice grain, 2015 in thousand US$** | **water consumption cost, 2015 in thousand US$** | **revenue for rice grain, 2015 in thousand US$** | **water consumption cost, 2015 in thousand US$** | **revenue for rice grain, 2015 in thousand US$** | **water consumption cost, 2015 in thousand US$** | **revenue for rice grain, 2015 in thousand US$** | **water consumption cost, 2015 in thousand US$** | **revenue for rice grain, 2015 in thousand US$** | **water consumption cost, 2015 in thousand US$** |
| Improved irrigation water management - continuous flooding | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. | 1,124,200 | 425,040 | 25,255 | 21,755 |
| SRI – conventional agriculture | n.d. | n.d. | 801,040 | n/A | n.d. | n.d. | 749,760 | n/A | 16,553 | 11,350 |
| Direct seeding – transplanting | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. | 902,440 | 55,440 | n.d. | n.d. |
| Dry tillage – puddling | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. | 517,440 | 175,560 | n.d. | n.d. |

| | |
|---|---|
| **GREEN** | Increase |
| **RED** | Decrease |
| **BLACK** | No change |
| n.d. | no data |

**Table A.4.** Air pollution costs from fertilizer use and rice straw burning, revenues, nutrient value of rice straw in irrigated lowland systems (IL) in US dollars per hectare. Values refer to hypothetical scenarios. Field level results have been upscaled to the entire area of irrigated lowland systems in each country.

| | CALIFORNIA | | | | CAMBODIA | | | | COSTA RICA | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Scenarios | Revenue for rice grain, thousand US$ | Nutrient value of rice straw, thousand US$ | Air pollution costs from straw burning, thousand US$ | Air pollution costs from fertilizer ammonia emissions, thousand US$ | Revenue for rice grain, thousand US$ | Nutrient value of rice straw, thousand US$ | Air pollution costs from straw burning, thousand US$ | Air pollution costs from fertilizer ammonia emissions, thousand US$ | Revenue for rice grain, thousand US$ | Nutrient value of rice straw, thousand US$ | Air pollution costs from straw burning, thousand US$ | Air pollution costs from fertilizer ammonia emissions, thousand US$ |
| Straw incorporation - straw burning | 9,289 | 29,594 | 1,462,197 | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. |
| Straw rolling - straw burning | 47,091 | 29,593 | 1,462,197 | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. |
| Straw removal - straw burning | 1,944 | 0 | 1,462,197 | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. |
| No mineral fertilizer use - mineral fertilizer use | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. |
| Reduced mineral fertilizer use - standard mineral fertilizer use | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. | 7,746 | n.d. | n.d. | 170 |

| | PHILIPPINES | | | | SENEGAL | | | |
|---|---|---|---|---|---|---|---|---|
| **Scenarios** | **Revenue for rice grain, thousand US$** | **Nutrient value of rice straw, thousand US$** | **Air pollution costs from straw burning, thousand US$** | **Air pollution costs from fertilzer ammonia emissions, thousand US$** | **Revenue for rice grain, thousand US$** | **Nutrient value of rice straw, thousand US$** | **Air pollution costs from straw burning, thousand US$** | **Air pollution costs from fertilzer ammonia emissions, thousand US$** |
| Straw incorporation - straw burning | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. |
| Straw rolling - straw burning | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. |
| Straw removal - straw burning | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. |
| No mineral fertilizer use - mineral fertilizer use | 46,200 | n.d. | n.d. | n.d. | 70,185 | n.d. | n.d. | 2,837 |
| Reduced mineral fertilizer use - standard mineral fertilizer use | 400,400 | n.d. | n.d. | 391,160 | 34,903 | n.d. | n.d. | 1,135 |

| | |
|---|---|
| **GREEN** | Increase |
| **RED** | Decrease |
| **BLACK** | No change |
| n.d. | no data |

**Table A.5.** Energy values of rice husk, and **revenue** for rice grain in US dollars per hectare. Values refer to hypothetical scenarios. Average revenues per hectare and energy values have been upscaled to the entire area of rice production in each country.

| | **CALIFORNIA** | | **CAMBODIA** | | **COSTA RICA** | | **PHILIPPINES** | | **SENEGAL** | |
|---|---|---|---|---|---|---|---|---|---|---|
| **Scenarios** | **Revenue for rice grain, thousand US$** | **Enery value of rice husk, thousand US$** | **Revenue for rice grain, thousand US$** | **Enery value of rice husk, thousand US$** | **Revenue for rice grain, thousand US$** | **Enery value of rice husk, thousand US$** | **Revenue for rice grain, thousand US$** | **Enery value of rice husk, thousand US$** | **Revenue for rice grain, thousand US$** | **Enery value of rice husk, thousand US$** |
| Use of rice husk for electricity production - no use | 1,006,474 | 290, 047 | 1,142,300 | 347,820 | 136,663 | 4,022 | 2,948,400 | 343,200 | 102,337 | 12,437 |

| | |
|---|---|
| **GREEN** | Increase |
| **RED** | Decrease |
| **BLACK** | No change |
| n.d. | no data |

**Table A.6.** GHG emission costs and, revenues, in irrigated lowland systems (IL) and rainfed lowland systems (RL) in US dollars per hectare. Values refer to hypothetical scenarios. Field level results have been upscaled to the entire area of IL and RL systems in each country. * These reductions only occur initially. Straw incoporation will increase yields over the long term.

| GHG emissions (IL+RL) | | CAL | | CAM | | CR | | PHI | | SEN | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Scenarios | | revenue for rice grain, 2015 in thousandUS$ | GHG emission cost, 2015 in thousand US$ | revenue for rice grain, 2015 in thousandUS$ | GHG emission cost, 2015 in thousand US$ | revenue for rice grain, 2015 in thousand US$ | GHG emission cost, 2015 in thousand US$ | revenue for rice grain, 2015 in thousand US$ | GHG emission cost, 2015 in thousand US$ | revenue for rice grain, 2015 in thousand US$ | GHG emission cost, 2015 in thousand US$ |
| **Change from...** | **to....** | | | | | | | | | | |
| Conventional agriculture | SRI (RL) | n.d. | n.d. | 801,040 | 257,920 | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. |
| Straw burning | Straw incorporation (IL) | 9,797 | 126,677 | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. |
| Standard mineral fertilizer application | Reduced mineral fertilizer use (IL) | n.d. | n.d. | n.d. | n.d. | 7,746 | 76 | 400,400 | 24,640 | 34,903 | 756 |
| Mineral fertilzer application only | Organic and mineral fertilizer use (IL) | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. | 551320* | 385,000 | n.d. | n.d. |
| Local variety | New variety (IL) | n.d. | n.d. | n.d. | n.d. | n.d. | n.d. | 748,440 | 24,640 | n.d. | n.d. |

| | |
|---|---|
| **GREEN** | Increase |
| **RED** | Decrease |
| **BLACK** | No change |
| n.d. | no data |